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I. INTRODUCTION

1. Contexte de la recherche

1.1. Un système à bout de souffle

La croissance démographique et la prospérité industrielle que l'humanité a pu connaitre durant le XXème siècle sont en grande partie apportés par une amélioration considérable des rendements de production alimentaire. L'agriculture (du latin *agricultura*, composé à partir de *ager*, champ et *colere*, cultiver), peut être définie comme « le processus par lequel les humains aménagent leurs écosystèmes et contrôlent le cycle biologique d'espèces domestiquées, pour produire des aliments et des ressources utiles à leurs sociétés »[1]. L'agriculture est donc par définition une activité qui transforme les écosystèmes naturels en agroécosystèmes, dont la qualité et la productivité dépendent à la fois des conditions environnementales et de la gestion par l'humain. Les plantes en tant que producteurs primaires sont des organismes autotrophes qui transforment des molécules minérales en structures biologiques. Comme pour tout organisme vivant, leurs constituant majeurs sont l'eau, le carbone, l'azote, le phosphore et dans une moindre mesure le potassium. Historiquement l'agriculture s'est développée dans des régions où les apports d'eau sont assurés soit par la proximité d'une source en eau pour l'irrigation, soit par une pluviométrie suffisante. Le carbone est présent dans l'atmosphère, sous forme de CO_2, d'où il est anabolisé par photosynthèse. Les autres macroéléments, l'azote, le phosphore et le potassium, sont quant à eux prélevés sous forme de solutés dans la solution aqueuse du sol. Il est évident donc, que les rendements agricoles dépendent d'un ensemble de facteurs pédoclimatiques, spécifiques de la physiologie de la culture, dont la connaissance et la compréhension ont permis d'améliorer la gestion des systèmes agricoles. La fertilisation en N, P et K représentant un des piliers principaux pour l'amélioration des rendements, de grands efforts ont été consacrés pour assurer la disponibilité de ces ressources. Les gisements de phosphore et de potassium, découverts depuis le 19ème siècle ont permis d'assurer un apport constant en culture annuelle, sans périodes de jachère jusque-là nécessaires pour reconstituer le statut fertile du sol. L'azote a été apporté principalement par les déjections animales comme les

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[1] Marc Dufumier, ingénieur agronome, Institut national agronomique Paris-Grignon, Radio France Culture, émission du 30 mars 2019

fumiers obtenus des élevages et épandus dans les champs sous cette forme de fertilisant organique (FO).

Pourtant, au début du XXème siècle, l'agriculture a connu un tournant majeur lorsque Fritz Haber en 1909 réussit à condenser l'azote atmosphérique en ammoniac dans des conditions contrôlées. Cinq ans plus tard, l'équipe de Carl Bosch met au point le procédé industriel de synthèse d'ammoniac basé sur les travaux de recherche de Haber. Le procédé Haber-Bosch voit alors le jour et inspire largement la chimie industrielle moderne, qui à son tour révolutionne l'agriculture. La production d'engrais minéraux azotés alors possible, les limites naturelles des agrosystèmes sont en grande partie levées et les rendements des principales céréales cultivées en Europe (blé, maïs, orge) triplent en 10 ans après la fin de la deuxième guerre mondiale (Agreste 2008). De plus la mécanisation du travail agricole, la sélection variétale et l'amélioration génétique des cultures ont contribué à une capacité de production accrue sans précédent, ce qui a inscrit le 20ème siècle dans une dynamique de croissance effarante, grâce à cette « révolution verte ».

L'agriculture est aujourd'hui responsable d'environ ¼ des émissions de gaz à effet de serre (GES), directement dues à la mécanisation du travail et à l'utilisation de carburants fossiles, ou indirectement dues à la déforestation (IPCC 2014). L'augmentation de la population mondiale exerce une pression sur la sécurité alimentaire, et engendre l'intensification du changement d'usage des terres, en particulier par la déforestation. Ceci abouti à l'appauvrissement des sols et à la minéralisation du carbone stocké (et donc l'émission de CO_2) dans la biomasse et dans le sol (Fargione et al. 2008, Potma Gonçalves et al. 2018). L'impact anthropique est déterminant dans les cinétiques de transformation et de stockage de l'azote et du carbone. C'est pourquoi le devenir de ces éléments, régi désormais aussi par les activités humaines, est sujet d'étude depuis plusieurs décennies (Delwiche 1970). Ces études ont fait ressortir les problématiques de pollution des écosystèmes en dépit de la production alimentaire et du progrès industriel. En effet, si une plus grande concentration d'azote réactif dans le sol, c.à.d. non inerte comme le N_2, peut assurer la productivité des agroécosystèmes, les déséquilibres chimiques, et l'acidification des écosystèmes qui découlent d'une surfertilisation, peuvent au contraire, l'inhiber (Bolan et al. 2005, Sprogis et al. 2017, Ge et al. 2018). L'excès d'azote réactif impacte non seulement sur la biodiversité, mais présente des risques majeurs pour la santé humaine (EPA 1998).

Le cycle du carbone a été fortement perturbé depuis la révolution industrielle. L'exemple le plus courant est l'augmentation des concentrations de CO_2 dans l'atmosphère (Fig. 3) en parallèle avec d'autres gaz carbonés (méthane, monoxyde de carbone et des composés organiques volatiles). Les concentrations de CO_2 dépassent aujourd'hui les 400 ppm, ce qui représente plus de 30% d'augmentation par rapport aux ~280 ppm mesurées dans des glaciers observés avant que Watt ne mette au point la première machine à vapeur qui transforme en énergie mécanique la vapeur produite par l'eau chauffée au charbon en 1769 (Siegenthaler and Oeschger 1987, NOAA 2019). L'utilisation d'énergies fossiles et la déforestation semblent être les activités qui contribuent majoritairement à ce phénomène d'augmentation du CO_2 atmosphérique (Myrold 1999).

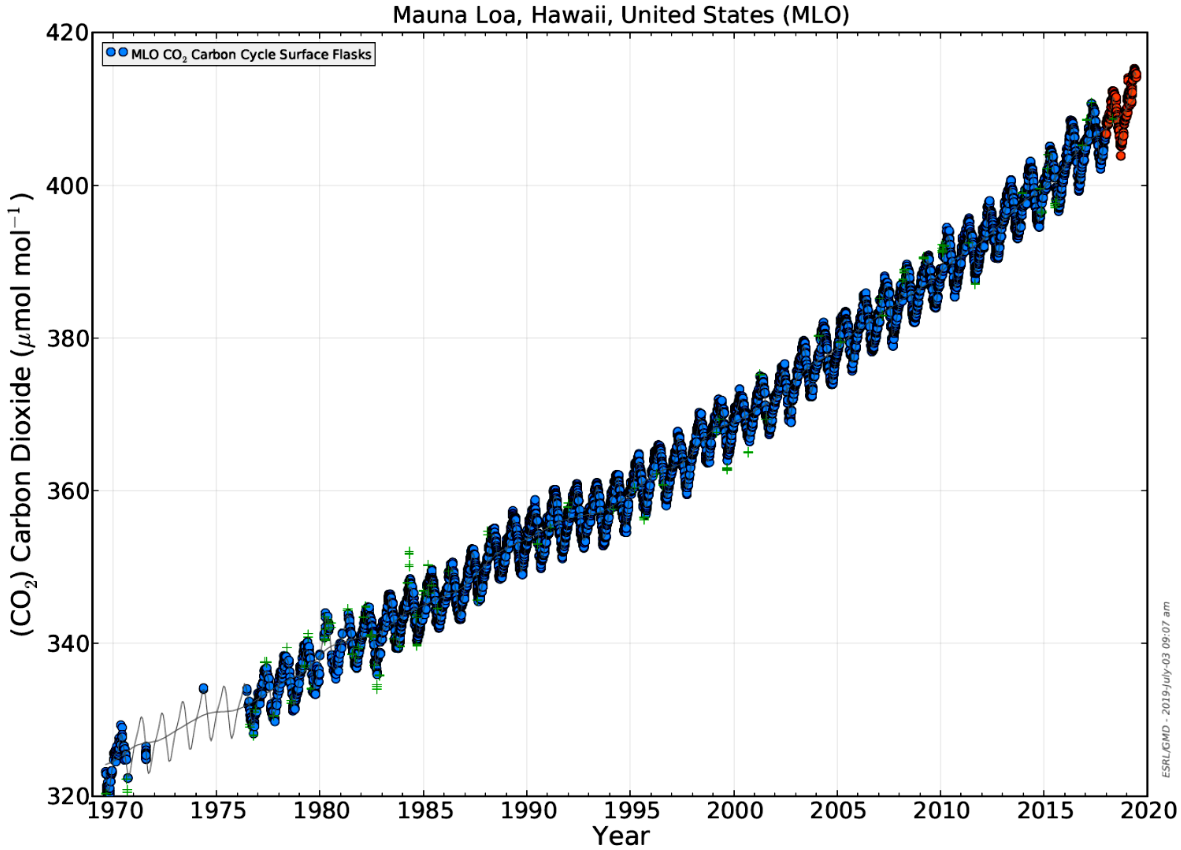

Figure 1 : Augmentations des concentrations de CO_2 atmosphérique depuis la révolution industrielle à la fin du 20ème siècle. Mesures sur l'air piégé dans de la glace ancienne de la Siple Station en Antarctique et les données annuelles moyennes de *Mauna Loa Observatory*

1.2. Vers un vent nouveau

Les progrès démographiques et économiques réalisés grâce à l'industrialisation de l'agriculture, se sont construits en négligeant les dynamiques de production et de consommation des ressources. Toutes ces avancées ont eu des conséquences adverses sur l'environnement, dont les risques aujourd'hui identifiés, pèsent dans la conscience collective et remettent en question tout le système économique et sociétal moderne. Force est de constater que ce même modèle de développement a été appliqué à tous les secteurs économiques, dont l'agriculture est juste un des exemples. Le modèle économique linéaire adopté depuis la révolution industrielle est basé sur le « produire, consommer, jeter » (A. Frosch and E. Gallopoulos 1989, Esposito et al. 2017, Korhonen et al. 2018, Millar et al. 2019). Ce paradigme, pourtant confortable a atteint ses limites dans de nombreux domaines et met en danger la capacité de production de denrées et d'énergie pour les populations futures (FAO 2019). Face à la nécessité d'optimiser l'utilisation des ressources, de limiter les déchets et de réduire les émissions de GES, et d'azote réactif dans l'environnement, un nouveau modèle s'impose, qui doit être diamétralement opposé à l'actuel, souvent qualifié de « modèle de gaspillage »[2] . La notion de recyclage des matières et de réutilisation des objets, et services entre personnes est au cœur d'un nouveau modèle, vers lequel les sociétés devraient tendre afin de limiter l'épuisement des ressources finies. Un modèle pour lequel le déchet est une ressource, mobilisable, transformable, réutilisable et par conséquent non limitée (Fig. 2). C'est notamment le concept de l'économie circulaire, formalisé dès 2002 par William McDonough et Michael Braungart (McDonough and Braungart 2011).

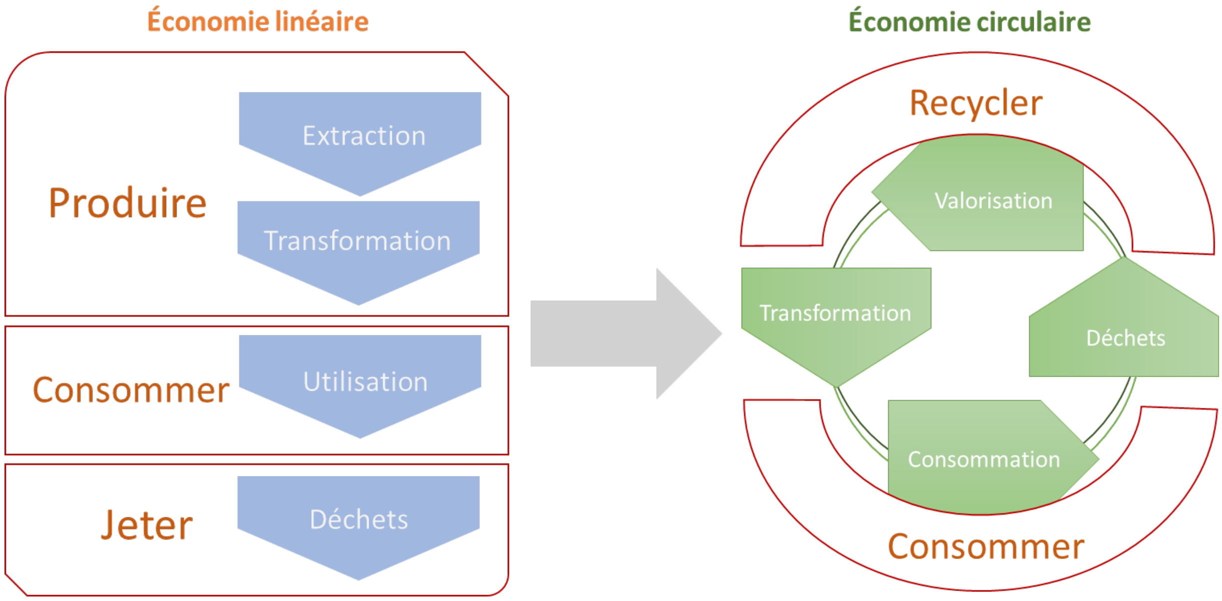

Figure 2 : Schéma conceptuel des deux modèles de consommation, linéaire ou circulaire

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[2] Ministère de la Transition écologique et solidaire. L'économie circulaire. 23 avril 2018

En agriculture, le cercle vertueux de l'économie circulaire peut être mis en place à travers l'utilisation de fertilisants organiques issus de gisement locaux, ainsi qu'en mobilisant une approche agroécologique optimisant les services écosystémiques pour la lutte contre les nuisibles, et en limitant le travail mécanisé par des machines à moteurs thermique (Tilman et al. 2002, Pretty 2008). Dans le contexte environnemental actuel il est crucial de proposer et d'évaluer les leviers d'action pour lutter contre le réchauffement climatique. La croissance démographique soulève de nombreuses questions sur la capacité du système actuel à maintenir cette dynamique socio-économique et impose une transition vers les approches de gestion durable dans tous les domaines. Face aux enjeux sociaux qui découlent de la dégradation de l'environnement, il est urgent de proposer des approches scientifiques pour accompagner les politiques de transition écologique et pour proposer des pistes pour une meilleure gestion des ressources par l'ensemble des acteurs publiques et privés. L'agriculture apparait comme un des piliers de la lutte contre le réchauffement climatique, car le sol représente le second puits de carbone après les océans et contient déjà 2 à 3 fois plus de carbone que l'atmosphère. L'initiative 4 pour 1000, lancée par la France pendant la COP 21 en 2015, incite à adopter des mesures concrètes pour séquestrer 0,4% de plus de carbone par an dans les 40 premiers centimètres du sol. Une telle démarche est possible seulement en adoptant des modes de gestion en phase avec les concepts agroécologiques. Il est évident, qu'on devrait rechercher le meilleur compromis entre les entrées et les sorties de matières organiques au sein des systèmes agricoles, tout en mobilisant les efforts pour trouver des alternatives aux énergies fossiles.

1.3. L'agriculture - un levier pour la régulation du climat

La qualité du sol cultivé est une préoccupation majeure afin d'assurer la production agricole. Dans un système de production artificialisé, tel qu'un agroécosystème, où le prélèvement systématique de biomasse est l'objectif, la bonne gestion qui assure la durabilité est d'importance capitale. Pourtant la gestion s'avère être complexe, à l'image du terme « qualité du sol », qui agrège des notions comme sa fertilité, sa santé et sa durabilité. Il est donc difficile de considérer une gestion simple, face à la complexité des processus physiques, chimiques et biologiques qui se déroulent dans un sol. Si on considérait uniquement la fertilité du sol, on se contenterait d'assurer la disponibilité en éléments chimiques (Cheng et al. 2007), en particulier ceux nécessaires à la croissance végétale, N, P, K principalement. Pourtant la santé du sol est une notion beaucoup plus complexe (Stevens 2018), qui détermine la durabilité de l'agroécosystème et la capacité de résilience face aux stress (Doran and Parkin 1994). Ainsi, l'évaluation de la santé du sol est systémique et s'articule autour de trois aspects qualitatifs - physique, chimique et biologique. La qualité physique du sol est liée à sa structure, qui est étroitement liée à la quantité de carbone organique qu'il contient (Johannes et al. 2017). Une qualité chimique assure la nutrition des plantes et donc leur croissance et développement. La qualité biologique quant à elle représente le moteur des processus au sein d'un sol (Van Der Heijden et al. 2008). La biologie du sol est au cœur des dynamiques fonctionnelles qui régissent la qualité physique et chimique. À titre d'exemple on peut mentionner la minéralisation et l'immobilisation de l'azote, du carbone, mais aussi la structuration, à travers l'agrégation et la désagrégation, les transferts d'eau, de gaz et d'éléments, tous gouvernés par l'activité biologique du sol. Face à cette complexité, il parait évident que le sol est un écosystème dont l'équilibre fragile doit être préservé afin d'assurer une production dans le temps. Un sol sain est un sol durable et donc résilient, capable d'assurer la croissance des plantes sans compromettre ou nuire à l'environnement (Gregorich et al. 1995).

Les apports exogènes (càd la fertilisation et l'amendement) sont nécessaires dès lors qu'on exporte de la matière, comme c'est le cas dans un système agricole. Les déchets organiques ont l'avantage de contenir du N, du P, du K, et d'autres (micro)éléments, le plus souvent aujourd'hui apportés par des fertilisants de synthèse produits à partir de ressources finies (carrières de phosphates, de potasse) ou par utilisation d'énergie (processus Haber-Bosch pour la production d'ammoniac). Les déchets organiques d'origine animale ou végétale, peuvent aussi contribuer à l'enrichissement du sol en carbone organique, car en tant que matières organiques ils contiennent une fraction de carbone non-négligeable. Or, un sol enrichi en carbone est un sol plus structuré, contenant les ressources pour maintenir une biodiversité fonctionnelle, qui recycle la matière organique exogène pour fournir les macro et microéléments nécessaires aux cultures. Au-delà de la possibilité de fermer la boucle des éléments par le recyclage, l'apport de fertilisants organiques permet la croissance végétale qui contribue à l'absorption d'une partie des GES à travers la photosynthèse. Ainsi une partie du CO_2 atmosphérique est transformé en carbone organique, dans des structures biochimiques plus ou moins complexes. Lors de la décomposition les fractions récalcitrantes (lignine et dans une moindre mesure les holocelluloses) peuvent être stockées dans le sol à moyen ou à long terme (Berg and McClaugherty 2008). Le sol, à travers l'agriculture peut constituer un véritable levier d'action contre le réchauffement climatique, ainsi qu'un moyen durable pour la valorisation des déchets organiques en tant que fertilisants pour la production agricole.

1.4. Les déchets organiques à la Réunion

L'ile de la Reunion est située dans le sud-ouest de l'océan Indien (20°54'12.2"S 55°31'46.6"E) et se caractérise par une croissance démographique forte (+ 3,6 % entre 2010 et 2015), comparé au reste du territoire national (+ 2,4 % entre 2010 et 2015)[3]. Le nombre d'habitants de cette région insulaire (842 767 habitants en 2014) augmente d'environ 5000 par an sur une superficie pourtant restreinte (2504 km^2). De plus l'ile de la Réunion accueille un flux touristique important (450 000 touristes en 2012). Le relief marqué de l'île de la Réunion et notamment les massifs hauts, le volcan, les ravines présentent de fortes pentes ce qui limite les territoires constructibles aux « basses terres ». La concentration de la population sur la périphérie de l'ile participe à la dynamique de production de **déchets organiques d'origine urbaine** à hauteur de 17% de la masse brute (MVAD-Réunion 2015).

D'autre part les « hautes terres » et surtout les « plaines » vertes réunissent les conditions pour la production bovine, grâce à un climat favorable à une végétation fourragère typique des prairies en régions tempérées. L'ile de la Réunion dispose de filières d'élevage dont on peut mentionner les filières bovines (viande et lait), porcine, avicole, cunicole, caprine, ovine ou encore cervidés, qui contribuent fortement à la production des **déchets organiques** à hauteur de 66% de la masse brute[4] (MVAD-Réunion 2015).

La production de canne à sucre quant à elle, est surtout concentrée dans les « basses terres » (en concurrence avec les zones aménageables) et génère des **déchets organiques d'origine agro-alimentaire** (écume, vinasse) à hauteur de 17% de la masse brute (MVAD-Réunion 2015).

Ainsi, les déchets organiques à la Réunion peuvent être classées en trois catégories en fonction de leurs origines. Une partie des déchets organiques est produites en zone urbaine, tels les composts, les déchets verts et les boues de stations d'épuration (boue de STEP). Une autre partie provient de l'élevage et de l'agriculture, comme les lisiers, les fumiers et les fientes en général. La filière cannière, quant à elle, produit des déchets d'origine agro-industrielle, générées lors de la production de sucre ou lors de la production spiritueuse (rhum). Parmi ces déchets il y a la bagasse, résidu fibreux issue de la première étape d'extraction du jus de canne

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[3] INSEE

[4] http://www.mvad-reunion.org/spip.php?rubrique10

et les écumes de sucrerie, issues de la décantation du jus : la mélasse, le sous-produit final de l'extraction du sucre, et la vinasse, le sous-produit de la mélasse après distillation de celle-ci.

La situation insulaire de la Réunion est source d'isolement ce qui accroit considérablement le coût du transport des déchets pour leur traitement et recyclage ailleurs sur le territoire national. À l'exception des déchets classés « dangereux », selon le Ministère de la Transition écologique et solidaire, impliquant l'exportation de certains déchets, principalement vers l'Asie et des pays de la zone de l'Océan Indien (Douanes 2018), la prise en charge locale des déchets organiques urbains consiste le plus souvent au stockage et à l'enfouissement (Agorah 2016), les déchets d'élevage et de l'agro-industrie étant déjà valorisés en agriculture à l'exception des vinasses rejetées en mer.

1.5. Valorisation des déchets organiques agricoles à la Réunion

La valorisation des déchets organiques n'est pas une pratique nouvelle, mais plutôt une pratique « oubliée » car suppléée au cours du XXème siècle par la production de fertilisants de synthèse (Houot et al. 2016). En outre, la valorisation en agriculture de matières organiques provenant de déchets domestiques et industriels, répond à des enjeux environnementaux et économiques tels que le recyclage des déchets, l'augmentation des coûts de l'énergie investie pour la fabrication des engrais minéraux de synthèse, et la raréfaction des ressources minières en phosphore, mais aussi la diminution des taux de matière organique des sols en général. Depuis 2010 à 2014, le CIRAD avec six autres entités, dont une communauté de communes et la chambre d'agriculture, anime le projet Girovar, « Gestion intégrée des résidus organiques par la valorisation agronomique », qui avait pour objectif de construire une filière de recyclage des déchets organiques en produits fertilisants agricoles normés. Les promesses écologiques et économiques de ce projet ont poussé l'ensemble des acteurs concernés (urbains et agricoles) du territoire de la côte ouest de la Réunion à promouvoir une logique de co-valorisation économique des ressources organiques quelles que soit leur origine (agricole, agro-industrielle, urbaine) plutôt que celle d'élimination de déchets polluants.[5] Ainsi, pour répondre aux exigences du gouvernement relatives à la transition énergétique pour la croissance verte, les filières de valorisation des déchets organiques se structurent de plus en plus. Actuellement une

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[5] Ile de la Réunion : Girovar, ou comment accompagner l'évolution de la filière de recyclage des déchets organiques en fertilisants. www.cirad.fr/nos-recherches/resultats-de-recherche/2017/ile-de-la-reunion-girovar-ou-comment-accompagner-l-evolution-de-la-filiere-de-recyclage-des-dechets-organiques-en-fertilisants

valorisation des déchets organiques est prévue via la filière énergétique, principalement par deux processus mis en œuvre. Le premier, encore peu développé, consiste dans la méthanisation, càd la production de biogaz par des voies biologiques en conditions contrôlées ; à noter que le digestat issu de cette transformation est valorisable en agriculture. Le deuxième est plus direct et consiste dans l'incinération, et la récupération de l'énergie thermique pour la production de vapeur utilisée par le process sucrier et pour la production d'électricité [6] ; les cendres de combustion sont recyclées en agriculture. La voie de valorisation directe en agriculture en tant que fertilisants est également possible dans le contexte local comme c'est le cas des déchets d'élevage ou de l'industrie sucrière.

1.6. La place de la canne à sucre dans la transition énergétique

La principale culture de la Réunion est la canne à sucre avec une surface agricole utile avoisinant les 60% des terres arables[7]. C'est une culture historique, dont la production est principalement destinée à l'exportation de sucre vers la France métropolitaine et la production de boissons spiritueuses (le rhum). Les dernières années, à l'échelle mondiale, un grand intérêt est porté à la production d'agrocarburant, il s'agit en effet de la production par fermentation du sucre en éthanol. L'éthanol est produit à partir de plantes sucrières en particulier à partir de canne à sucre dans les régions tropicales ou à partir de matières végétales riches en amidon comme le maïs dans les régions tempérées (Vohra et al. 2014). Le Brésil, premier producteur mondial de canne à sucre co-génère avec les Etats-Unis environ 90% de la production mondiale de bioéthanol[8]. Grâce aux subventions gouvernementales et la raréfaction des carburants fossiles, les agrocarburants sont devenus rentables à tel point que les projections à l'échelle de 2020 montrent qu'environ 70% des récoltes de canne à sucre au Brésil seront destinées à la production de bioéthanol (Leal et al. 2013b). Les inconvénients liés à la production d'agrocarburants sont liés aux besoins d'augmenter les surfaces cultivées. Or les changements d'usage des terres, mènent le plus souvent à une augmentation des émissions des gaz à effet de serre, comparé au biotope de départ (Fargione et al. 2008). Face à ce danger, les normes législatives sont de plus en plus strictes quant aux changements d'usage des terres. Une des

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[6] Valorisation des déchets à La Réunion. Question écrite n° 02233 de Mme Viviane Malet, publiée dans le JO Sénat du 30/11/2017 - page 3749

[7] AGRESTE - DAAF La Réunion - Recensement Agricole 2010

[8] RFA analysis of public and private data sources 2019

ouvertures possibles est la substitution des terres agricoles utilisées pour la production alimentaire. Ici surgit une autre réticence vis-à-vis de la concurrence d'occupation de terres arables par des cultures non-alimentaires, mettant ainsi en danger la sécurité alimentaire (Tilman et al. 2009).

Les résidus de coupe (paille) – qui consiste en les feuilles, les gaines et les parties sommitales qui restent après la récolte des tiges usinables, sont quant à eux moins exploités comparé aux coproduits fibreux issus de la transformation de la canne à sucre, qui sont déjà valorisés en agriculture ou en industrie. D'ailleurs, le paillis a longuement été éliminé par brulis pour faciliter la récolte manuelle. À la Réunion, la pratique du brulis est maintenant prohibée en raison des risques pour la santé humaine et des risques environnementaux, directement liés aux émissions de gaz de combustion toxiques et générateurs d'effet de serre. Pour faciliter cette transition, des programmes incitatifs ont été mis en place à travers les mesures agro-environnementales et climatiques (MAEC), qui consistent à subventionner les agriculteurs qui épaillent la canne et laissent le paillis dans les champs sous forme de mulch, pour lutter naturellement contre les adventices, et limiter le recours aux herbicides (Antoir et al. 2016). Pourtant cette ressource désormais utilisée pour ces effets positifs sur le sol cultivé, est aussi de plus en plus convoitée par les industriels en tant que matière première pour la production d'énergie, soit directement par combustion soit par l'intermédiaire de procédés chimiques qui aboutissent à la production de carburant. En effet le potentiel énergétique de la paille de canne à sucre représente environ 30% du potentiel énergétique de la culture et le paillis est considéré comme une matière première pour la production d'agrocarburant de seconde génération - « 2G ethanol » (Franco et al. 2013, Khatiwada et al. 2016). Cette nouvelle perspective de production de carburant solutionne en grande partie les controverses que lève l'utilisation des parties « nobles » de la culture, qui sont aussi utilisés pour la production alimentaire.

Ainsi, le paillage et la fertilisation organique font partie aujourd'hui des mesures agroécologiques concrètes (Minasny et al. 2017), mais les résidus de coupe des cultures (comme la paille de canne à sucre) représentent aussi aujourd'hui une matière première intéressante pour la production d'énergie (Vohra et al. 2014). Dans ce cadre il est nécessaire d'évaluer à la fois les avantages et les inconvénients agronomiques et environnementaux du recyclage conjoint de résidus de coupe (paillis) et des produits résiduaires organiques (PRO) en tant que fertilisants du sol.

2. Synthèse bibliographique

2.1. *Les cycles du C et du N*

Les cycles du carbone et de l'azote peuvent être présentés comme une série de processus biochimiques, impliquant les organismes vivants dans toutes les étapes de transformation. Le carbone atmosphérique, sous forme de CO_2 est fixé par les producteurs primaires par le processus de photosynthèse, qui le transforme dans un premier temps en sucres simples, comme le glucose. Les plantes transforment ensuite ces sucres en structures biochimiques fonctionnelles, structurelles ou de stockage, plus complexes, comme les sucres complexes, les lipides, les protéines etc. Les organismes hétérotrophes (producteurs secondaires) consomment ensuite la matière organique végétale, pour synthétiser leurs propres structures biochimiques, qui peuvent à leur tour être consommées par les organismes carnivores ou omnivores. La matière organique est ensuite déposée au sol, par les fèces (matière organique non-assimilé) ou bien après la mort des organismes. Cette matière organique est remobilisée par les décomposeurs du sol – les organismes coprophages et saprophytes. Tout au long de cette chaîne trophique les organismes vivants utilisent une partie de la matière organique assimilée pour le maintien ou le développement de leurs structures organiques, et une autre partie afin d'obtenir de l'énergie nécessaire pour leur fonctionnement. Ainsi la respiration aérobie qui fournit cette énergie consiste à dégrader les molécules biochimiques complexes, jusqu'à la minéralisation du carbone en formant ainsi du CO_2, qui repart dans l'atmosphère. Le stockage du carbone dans le sol est une autre possibilité. Le carbone stocké représente le carbone non-minéralisé par les organismes, qui se transforme en matière organique stable à long terme. La matière organique stable constitue le complexe argilo-humique du sol, ou bien se présente sous forme de carbone fossile (charbon, pétrole, gaz naturel) (Myrold 1999).

L'azote est aussi régi par les organismes vivants lors de sa transformation. L'azote a 7 degrés d'oxydation, ce qui le prédispose à des possibilités de transformation biochimiques, des spéciations de stockage et de processus de transport nombreux et complexes. L'azote dans sa forme simple (N_2) est le constituant majeur de l'atmosphère terrestre (78%). Il peut être fixé dans les sols par des microorganismes de la famille *Rhizobiaceae* en symbiose avec les plantes légumineuses, et des actinomycètes du genre *Frankia*, en symbiose avec les aulnes et les filaos (Girard 2005). L'azote atmosphérique peut aussi dans une moindre mesure, être fixé par les parties aériennes des plantes grâce à des associations entre les plantes et des bactéries endophytes telles que *Lysinibacillus sphaericus* (Shabanamol et al. 2018).

L'azote du sol sous forme organique ou minérale peut être métabolisé par la biomasse du sol, ce qui établit un équilibre entre immobilisation et minéralisation. L'azote sous forme d'ammonium par exemple, produit lors de la minéralisation ou apporté par des fertilisants peut être immobilisé par la biomasse microbienne, ou oxydé lors du processus de nitrification, ou bien encore volatilisé sous forme d'ammoniac (Kirchmann and Witter 1989). Le plus souvent sous sa forme de nitrate, l'azote est prélevé par les plantes et immobilisé dans des structures biologiques complexes (protéines, ADN etc.). Les nitrates non-prélevés peuvent soit être lixiviés dans les couches profondes et les eaux souterraines, soit dénitrifiés et renvoyés dans l'atmosphère sous forme de diazote (N_2), mono (NO) ou dioxyde d'azote (NO_2) ou bien de protoxyde d'azote (N_2O) (Stevenson 1986). Toutes ces transformations sont contrôlées par les conditions du sol, comme le pH (Kirchmann and Witter 1989), la présence d'oxygène (Hellmann et al. 1997), la biodégradabilité du carbone (Andersson 1996), la disponibilité de l'azote (Martins and Dewes 1992), la température (Schlegel et al. 1993), la structure de la communauté microbienne du sol (Hellmann et al. 1997) et du C:N du substrat (Paillat et al. 2005).

Le carbone et l'azote, comme l'oxygène et l'hydrogène sont des éléments constitutifs du monde du vivant. D'autres éléments sont indispensables, mais sont présents à des quantités relativement faibles, c'est pourquoi on les classifie en tant qu'oligo-éléments. De par le fait que ces transformations naturelles sont effectuées par des processus biologiques, assurées par des organismes vivants, les cycles du carbone et de l'azote sont indissociables et interdépendants (Recous et al. 2017) (Fig. 3). Les processus biologiques de transformations des deux éléments dépendent de leur biodisponibilité respective, qui peut être limitante ou excédante, et qui peut restreindre ou favoriser leurs transformations concomitantes dans le temps (Recous et al. 1995, Lerch et al. 2013).

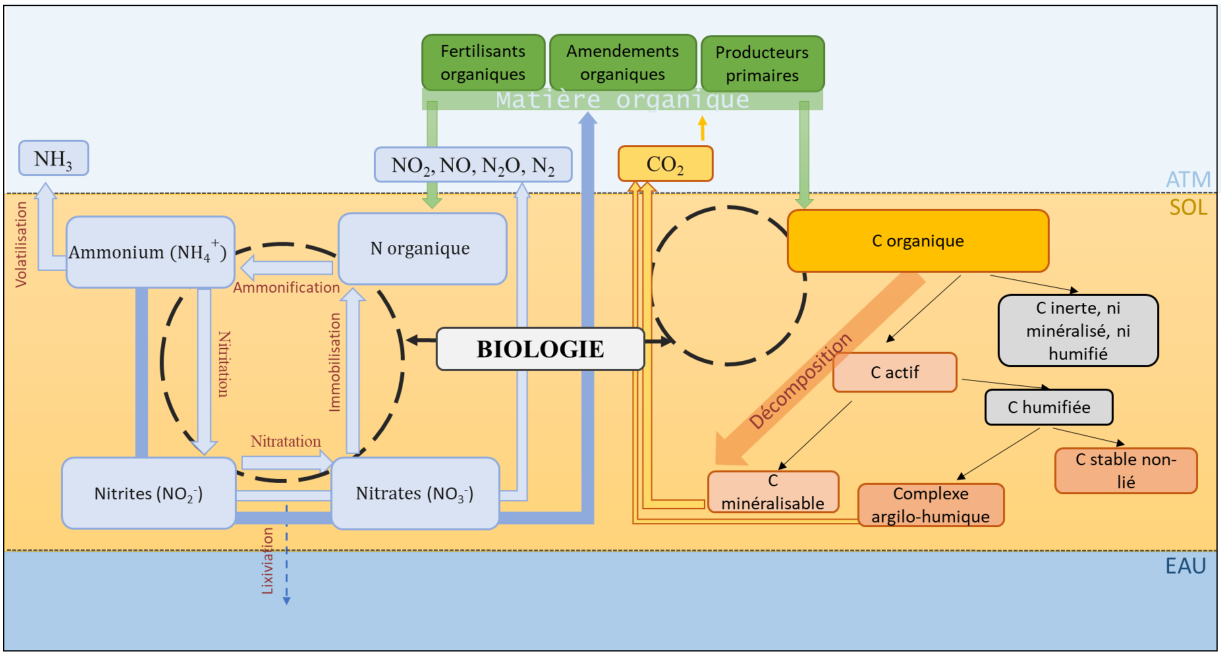

Figure 3 : Schéma conceptuel des cycles du carbone et de l’azote. Leurs biodisponibilités correspondantes exercent une rétroaction sur leurs cinétiques de transformation respectives. Les deux cycles sont interdépendants car les dynamiques de transformation sont contrôlées par la biologie du sol

2.2. La fertilisation

La fertilisation des systèmes agricoles permet d’assurer la croissance des cultures dans des conditions optimales en évitant l’appauvrissement du sol. On peut qualifier la fertilisation comme l’apport à un milieu de culture, tel que le sol, des minéraux nécessaires à la croissance et au développement des producteurs primaires – les plantes. Dans le cas de l’agriculture, on apporte le plus souvent les éléments majeurs nécessaires à la croissance des plantes – l’azote (N), le phosphore (P) et le potassium (K). L’origine de ces éléments peut être synthétique, extraite ou synthétisés à partir de réserves minérales, ou bien d’origine organique renouvelable, comme les PRO. Les fertilisants azotés organiques par exemple, peuvent être d’origine végétale, non-transformés comme les résidus de culture (paille), les engrais verts, des cultures intercalaires, fixatrices ou non-d’azote atmosphérique ; d’origine animale, non-transformés comme les déchets carnés d’abattoir, ou transformées, les farines de plumes hydrolysées, les farines de viande, les farines de cuir ; des résidus bruts d’origine mixte végétale et animale sans litière (lisiers, fientes, guanos), et avec litière (fumiers), les matières stercoraires ou transformés (par méthanisation et/ou compostage). Parmi les fertilisants azotés synthétiques on peut citer l’urée, les bicarbonates d’ammonium, les nitrates d’ammonium, les sulfates d’ammonium, l’hydroxyde d’ammonium et d’autres (Agehara and Warncke 2005, Snyder et al. 2009, Das and Adhya 2014, Abalos et al. 2016, Wang et al. 2017).

Pour la plupart des plantes, les formes biodisponibles de l'azote, l'ammonium (NH_4^+) et les nitrates (NO_3^-), sont absorbées par les systèmes racinaires en tant que solutés (Britto and Kronzucker 2013). Les difficultés de l'usage des FO sont liées à leur hétérogénéité qualitative des PRO. En effet, les cinétiques de minéralisation de l'azote, comme démontré plus haut, peuvent varier en fonction de la qualité physicochimique des FO (Lashermes et al. 2010). Ceci explique le succès des fertilisants synthétiques, qui ont une réponse immédiate aux besoins des cultures car leur biodisponibilité est importante. Pourtant l'usage de FO présente de nombreux avantages. La fertilisation organique permet le recyclage des PRO, autrement considérés comme des déchets. Leur disponibilité locale et leur implication dans le stockage du carbone dans le sol sont des atouts non-négligeables (Cole et al. 1997, Maltas et al. 2012, Minasny et al. 2017).

2.3. Gestion des résidus végétaux

Le paillage ou encore le « mulching » est en effet une technique agricole, qui consiste à couvrir le sol, lorsque celui-ci se retrouve à nu. Le terme « paillage », fait clairement référence à la paille, ou aux résidus de coupe, laissés au sol après la récolte des parties utiles des cultures. Pourtant le paillage est un terme générique car le paillis (mulch) comprend des matériaux différents, d'origine biologique ou non. Le mulch peut donc être des films en plastique (encore appelé paillage plastique), du gravier, du sable, des textiles d'origine biologique, des résidus végétaux, ou de la paille (Chakraborty et al. 2010, Gan et al. 2013, Mo et al. 2016, Prosdocimi et al. 2016).

Couvrir le sol par un mulch a de nombreux avantages sur les propriétés physicochimiques du sol. Parmi les avantages sur la physique du sol on peut citer la préservation de l'humidité (Lal 1974, Chakraborty et al. 2010, Awe et al. 2015), l'effet tampon sur la température du sol superficiel (Moody et al. 1963, Lal 1974, Siczek et al. 2015) et la protection de la structure du sol (Are et al. 2018). D'autres auteurs ont rapporté des effets bénéfiques sur la biologie et la chimie du sol, comme la biomasse du sol et la fertilité du sol (Wardle et al. 1999, Bationo and Buerkert 2001, Muñoz et al. 2017), la limitation de la croissance de plantes adventices (Campiglia et al. 2015, Splawski et al. 2016). Lorsque le mulch est d'origine organique, on constate une augmentation de la qualité d'infiltration d'eau lors des précipitations, ainsi cela prévient le ruissellement et l'érosion (Smets et al. 2008, Wang et al. 2016a). En effet, le recouvrement de paillis crée une barrière rugueuse et poreuse, qui réduit le débit d'arrivée d'eau au sol, en protégeant sa structure en surface et assurant une meilleure infiltration (Gilley and R. Kottwitz 1995, Cassol et al. 2004).

Ces effets positifs du mulch sont cependant dépendants des modalités d'application. Parmi les modalités on peut citer, l'emplacement du mulch, couvert homogène ou en interligne de culture ; les temps de résidence au sol et le moment d'application ; le type de mulch, ou encore l'enfouissement suite au labour ou pas (Lal 1978, Edwards et al. 2000, Li et al. 2004). En ce qui concerne la productivité de l'agroécosystème, généralement le constat est que les meilleurs rendements sont obtenus lors de l'application couplée de mulch et de fertilisants (Rahman et al. 2005, Xiukang et al. 2015, Mo et al. 2017).

Dans le cas de la canne à sucre, les techniques culturales ont longuement été déterminés par la facilité de récolte des tiges usinables. Ainsi avant la récolte pour faciliter l'accès aux rang de canne à sucre lors de la coupe manuelle, les feuilles, les gaines et les parties sommitales ont été brulées (Sandhu et al. 2017). Ces techniques ont aussi été justifiées par la diminution de la proportion de biomasse transportée en usine de production et la concentration en sucre plus importante de cette biomasse dépourvue de résidus qui diluent le taux de saccharose (degrés Brix plus bas) (De Beer et al. 1995, Núñez and Spaans 2008). Certains auteurs ont rapporté des effets négatifs de la conservation de paillis sur les rendements de canne à sucre, suite à la diminution de la température du sol dans le stade végétatif (Kingston et al. 2002, Sandhu et al. 2013). Des pertes ammoniacales plus importantes ont aussi été constatées lors de la fertilisation, lorsque les résidus de coupe ont été laissés au sol après récolte (Meyer et al. 1961, Bacon and Freney 1989). Toutefois, l'effet du paillage sur les rendements semble dépendant des conditions pédoclimatiques, avec notamment un effet positif de la rétention d'eau pour des sols drainants (Wood 1991, Antwerpen et al. 2001). Il semble important de souligner que ces effets négatifs n'ont pas été quantifiés lors d'applications de quantités différentes de paillis. Lisboa et al. (2019) par exemple ont constaté que la quantité de paille de canne à sucre laissée au sol ne semble pas affecter les rendements pendant la période d'un cycle cultural. Pourtant la quantité de paille de canne à sucre peut varier considérablement et se situe entre 10 à 30 t MS ha^{-1} (Hassuani et al. 2005, Cantarella et al. 2013). Cette biomasse est 10 fois supérieure comparée à la paille d'une culture comme le blé par exemple (Talebnia et al. 2010).

2.4. Les effets des fertilisants sur la décomposition de la paille

La fertilisation azotée influe aussi sur les dynamiques de décomposition de la litière (et donc de la paille) et par analogie, le devenir du carbone. En effet, il existe un consensus aujourd'hui sur le fait que la qualité de la matière organique, notamment les concentrations en P, S, K et N, sont positivement corrélés à la vitesse de décomposition dans l'étape initiale. Un C:N bas et donc une concentration en N plus grande, est considéré comme indicateur de décomposition plus rapide dans cette étape initiale (Berg and Ekbohm 1983). Dans le cas de la litière, il a été démontré que l'azote n'est pas entièrement biodisponible, car souvent lié à des structures biochimiques plus ou moins complexes comme la cellulose, les hémicelluloses et la lignine (Stevenson 1995). L'ajout de FO contenant de l'azote facilement accessible peut alors stimuler la décomposition lorsqu'il est apporté dans un système de paillage. Melillo et al. (1982) avaient proposé un autre indicateur de décomposition basé sur les effets opposés des concentrations en N (accélérant la décomposition) et de la lignine (retardant la décomposition). Cependant le rapport lignine : N bas peut présager une vitesse accrue en début de décomposition, mais pas en phase avancée de la décomposition. Cette phase avancée de la décomposition dépend de la vitesse de décomposition de la lignine et N semble avoir un effet retardateur sur la décomposition (Berg and Matzner 2011). Ceci peut être expliqué par le changement de la structure de la population microbienne de décomposeurs du sol qui est modifiée lorsque la disponibilité de l'azote est grande (Banerjee et al. 2016). Selon Entry (2000), la présence d'azote peut stimuler l'activité bactérienne en dépit de l'activité fongique, par inhibition de la synthèse de ligninase par les champignons (Berg and McClaugherty 2008). Toutefois, l'effet de la qualité chimique et physique de la MO sur la vitesse de décomposition est un facteur majeur à l'échelle locale. L'effet de la biologie du sol peut aussi être spécifique dans les régions arides, par exemple la présence de termites xylophages peut contribuer grandement à la décomposition de la MO ligneuse. Le sol aussi par sa structure et composition chimique peut moduler la cinétique de décomposition (Adu and Oades 1978). Mais à l'échelle globale, le climat, et donc la température et l'humidité semblent être des facteurs qui expliquent la vitesse de décomposition des MO dans les écosystèmes (Meentemeyer 1984).

Lorsqu'on mélange des FO et de la paille on peut ainsi constater des interactions entre les cinétiques de transformations du C et du N. Les interactions entre les matières organiques riches en N, tels que les FO ou les résidus de légumineuses, et les matières organiques riches en C, telles que la plupart des résidus de culture, comme la paille, peuvent être synergiques ou antagonistes. Dans le cas où les deux sources de C et de N n'interagissent pas on constate une additivité, où le taux de minéralisation est la moyenne des taux de minéralisation de chacune des matières organiques prises individuellement (Meier and Bowman 2010). Le cas d'additivité dans la littérature reste pourtant rare, la plupart des auteurs ont trouvé une synergie de la décomposition des résidus riches en C, stimulée par l'apport d'azote (Chantigny et al. 2001, Pérez-Harguindeguy et al. 2008, Berglund and Ågren 2012, Redin et al. 2014). D'autres auteurs ont constaté un effet nul sur la minéralisation du carbone, mais un effet synergique sur la minéralisation de l'azote, plus importante dans le cas où la paille était absente (Aita et al. 2012, Giacomini et al. 2015). Ces résultats montrent la relation très proche entre le C et le N, mais ne permettent pas de trancher sur l'effet des fertilisants sur les cinétiques de décomposition au champ.

Ainsi de nombreux modèles ont été développés pour décrire la cinétique et pour mieux prédire les dynamiques du C et du N dans le sol. La minéralisation nette des résidus de récolte et / ou des matières organiques fertilisantes a été modélisé à l'aide de modèles comme CANTIS (Garnier et al. 2003, Iqbal et al. 2014), APSIM (Luo et al. 2014), TAO (Pansu et al. 2003), NCSOIL (Noirot-Cosson et al. 2017), CropSyst (Rinaldi et al. 2017), CENTURY (Dimassi et al. 2018) etc. Si la plupart de ces modèles sont suffisants pour prédire le devenir du C et du N à partir d'apports organiques individuels, l'application d'un ensemble universel de paramètres capables de prédire la dynamique de tous les types de matière organique (Lashermes et al. 2010) ou de mélanges de différents types de matières organiques, est toujours difficile.

2.5. Émissions de gaz à effet de serre

La nécessité de réduire les émissions de gaz à effet de serre (GES) a été pointée en tant que mesure pour limiter le réchauffement climatique précipité au cours du 20ème siècle. L'agriculture contribue directement aux émissions de GES à travers les émissions de gaz comme le CO_2, le N_2O et le CH_4. L'agriculture à elle seule contribue à environ 90% des émissions de N_2O (IPCC 2013). Or le protoxyde d'azote a un potentiel de réchauffement global à 100 ans de 298 fois celui du CO_2. Les apports d'azote par les fertilisants peuvent être directement responsables des émissions de N_2O par le processus de dénitrification ou nitrification dénitrifiante (Fig. 4). Brièvement, l'oxydation partielle de NH_4^+ en NO_2^- en conditions limitantes en oxygène, est suivie d'une réduction en N_2O par dénitrification. Poth and Focht (1985) ont conclu que le processus de réduction de NO_2^- en N_2O par *N. europaea* correspond à la définition de la nitrification dénitrifiante. La production de N_2O lors de la nitrification est corrélée avec l'addition d'ammonium et non pas avec l'addition de nitrate, cela peut s'expliquer par l'utilisation de l'ammonium comme source d'électrons et par l'absence de l'activité de la nitrate réductase dans *N. europaea* (Poth and Focht 1985).

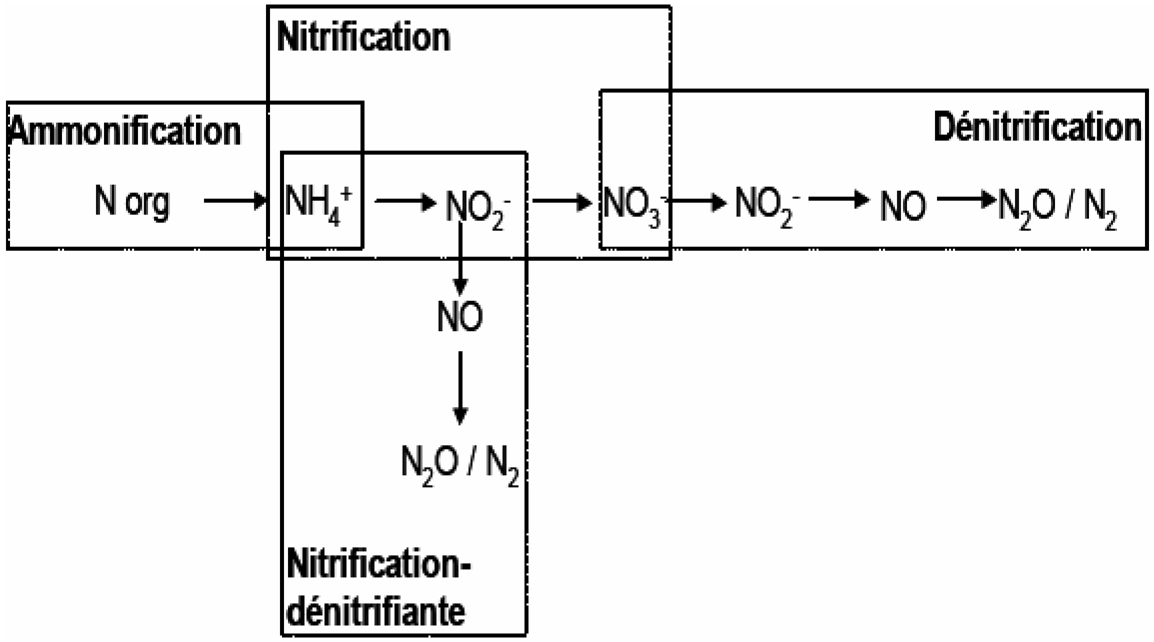

Figure 4 : Processus biologiques de transformation de l'azote conduisant à une production de N_2O (nitrification autotrophe, la nitrification dénitrifiante autotrophe et la dénitrification hétérotrophe). Processus intervenant aussi bien lors de l'élimination de l'ammonium que du nitrate *(Tallec et al. 2007)*

Les critères de classification des FO se réfèrent souvent à leur capacité à fournir de l'azote minéral pour les plantes. Ce critère reflète la volonté de représenter leur efficience agronomique (fertilisant azoté), tout en cherchant à déterminer une base qualitative et prédictive de minéralisation de l'azote. Les chercheurs ont ainsi tâché de comprendre les dynamiques de minéralisation de l'azote en les liant au début aux rapport C:N des apports (Aerts 1997, Nicolardot et al. 2001, Hadas et al. 2004). Pourtant, comme le souligne Recous et al. (1995), cette approche s'est avérée insuffisante pour la prédiction des dynamiques de C et de N et ainsi des capacités de stockage de carbone et d'émissions de GES. Le fractionnement biochimique successif, décrit par Van Soest et al. (1991), qui détermine des pools biochimiques plus ou moins décomposables a ainsi été utilisé avec un certain succès dans des modèles prédictifs différents (Rodrigo et al. 1997, Garnier et al. 2001, Pansu et al. 2003, Gabrielle et al. 2004). Différents indices de stabilité de la matière organique, basés sur les différentes fractions biochimiques ont aussi été mis en place, parmi lesquels on peut mentionner l'ISB (indice de stabilité biologique), le Tr (taux résiduel), ou encore l'ISMO (indice de stabilité de la matière organique), qui intègre de plus un potentiel de minéralisation mesuré en laboratoire (incubations en conditions contrôlées)[9].

La stabilité des FO est souvent interprétée comme un potentiel de stockage de carbone. Ceci veut aussi dire que les émissions de CO_2 sont d'autant plus réduites, que la matière organique apportée est stable. Il est plus difficile d'en déduire les émissions de N_2O car les conditions de sa production restent multifactorielles et ce gaz ne représente souvent qu'une faible part des formes minérales volatiles de l'azote, parmi d'autres (N_2, NO, NH_3). On peut néanmoins évoquer l'effet de la quantité de N apporté, qui a été bien documenté dans la littérature. Généralement, on considère que l'apport d'azote au sol accroit la biomasse microbienne, ce qui accroit la respiration microbienne. Une biomasse plus grande contribue à une consommation de O_2 plus élevée. Ceci crée des conditions favorables à la dénitrification (Paul et al. 1993, Kim and Dale 2008, Thangarajan et al. 2013, Millar et al. 2018). Les caractéristiques chimiques ou la spéciation de l'azote et du carbone apportés semblent avoir aussi une influence déterminante. Il est communément accepté que des apports en matières organiques riches en carbone et en azote biodisponibles favorisent aussi la croissance microbienne et par conséquence les émissions de GES (Snyder et al. 2009, García-González 2012, Senbayram et al. 2012). Ainsi, par exemple l'apport de fertilisants comme le lisier de porc, riche en ammonium et en carbone organique facilement dégradable est susceptible d'augmenter les émissions de GES (Velthof et al. 2003, Chantigny et al. 2010). La dénitrification étant un processus de réduction des nitrates ou des nitrites, un milieu enrichi en eau et donc appauvrit en oxygène est favorable aux émissions de N_2O. Généralement, les conditions d'humidité comprises entre 50% et 90% de « *water filled pore space* » (WFPS) sont les plus favorables pour les émissions de N_2O du sol (Zhu et al. 2013). Bateman and Baggs (2005) et R. Shelton et al. (2000) ont trouvé que les émissions de N_2O résultaient de la nitrification en dessous de 60% de WFPS, et de la dénitrification au-dessus de ce pourcentage de saturation des pores du sol. Toutefois, un milieu complétement anoxique et engorgé, avec une ressource en C suffisante, semble favoriser la dénitrification complète et ainsi décroit le rapport N_2O/N_2 (Rudaz et al. 1999). La qualité physique des apports peut donc influencer les dynamiques des GES après l'épandage, avec un risque accru d'émission corrélé au taux d'humidité des fertilisants, d'après la méta-analyse réalisée par Charles et al. (2017).

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[9] Caractéristiques des PRO épandus, INRA (2018)

Les résidus de coupe influent aussi sur les émissions de GES, avec des effets adverses rapportés dans la littérature, probablement liés à la préservation d'eau dans la surface du sol. La paille de canne à sucre et sa fraction de C organique soluble contribuent à l'augmentation des émissions de N_2O (Cerri et al. 2009, Carmo et al. 2013, Fracetto et al. 2017). Pour autant ce rôle du mulch dans l'augmentation des émissions de GES reste un sujet à controverse car d'autres études ne montrent aucun effet (Malhi and Lemke 2007, Chen et al. 2017). Certains auteurs ont même constaté une diminution des émissions de GES lorsque la quantité de paille laissée au champ augmente (Congreves et al. 2017, Pitombo et al. 2017). Ils ont justifié leurs résultats par le fait que les émissions de N_2O ne sont pas uniquement contrôlées par des processus simples comme le taux d'apport azoté, mais par des processus écologiques complexes. Par exemple, Pitombo et al. (2016) ont constaté que l'apport de paille de canne à sucre augmente la diversité fonctionnelle du sol, avec l'accroissement de l'abondance de taxons réducteurs de N_2O en N_2.

La complexité devient encore plus grande dès lors qu'on mélange des matières organiques différentes. En effet, il n'existe toujours pas de consensus au sein de la communauté scientifique concernant l'effet des mélanges de litières (Makkonen et al. 2012). Par exemple, la seule qualité chimique ne semble pas suffisante pour prédire si la dynamique de décomposition serait la somme des dynamiques de décomposition des deux litières seules, ce qui serait une additivité, ou s'il y aurait des interactions synergétiques ou antagonistes (Hoorens et al. 2003). Pourtant, beaucoup d'auteurs se sont concentrés sur les transferts de nutriments depuis les litières riches en azote vers des litières pauvres (Thomas 1968, Klemmedson 1992, Salamanca et al. 1998, Hoorens et al. 2003). D'autres auteurs ont souligné l'effet des composés spécifiques secondaires (composés phénoliques, tanins, etc.) capables d'intervenir en inhibant ou en catalysant la décomposition (Schimel et al. 1998). Les interactions constatées sont aussi contrôlées par la biologie du sol, qui à son tour peut être modifiée par la variabilité et la diversité des mélanges apportés. Cette interdépendance a été constatée car le taux de prélèvement des nutriments varierait avec l'apport des mélanges de litière. Ceci implique probablement des modifications des relations trophiques au sein des communautés de décomposeurs, lorsque des mélanges de litières servent de substrat (Hättenschwiler and Bretscher 2001, Hättenschwiler et al. 2011, Vos et al. 2011). Enfin la qualité physique de la litière a aussi été pointée en tant que facteur de décomposition. Wardle et al. (2003) ont trouvé que la capacité de rétention d'eau importante des mousses en forêt boréale, favorisait la décomposition de litières plus riches en azote, et la libération de celui-ci. Ceci a abouti à une interaction synergétique de décomposition du mélange des deux litières.

Dans un agroécosystème, tel que la culture de canne à sucre, le mulch est laissé au sol et la culture est fertilisée par des apports azotés d'origine synthétique ou d'origine organique, dont les qualités physicochimique et biologique varient. Il existe peu de références qui adressent la question du mélange entre FO et ce qui s'apparente à une litière végétale – la paille de canne à sucre. Par exemple, Huang et al. (2017) ont réalisé une étude dans un système de rotation maïs-blé, avec des quantités croissantes de fertilisant-N de synthèse, avec ou sans l'application de paille. Ils ont constaté systématiquement que les émissions de N_2O augmentaient avec l'augmentation de N apporté et l'application de paille. Ces résultats ont été confirmé dans une autre étude avec l'application d'urée (Wang et al. 2016b). Carmo et al. (2013) ont testé un gradient de quantité de paille de canne à sucre en combinaison cette fois avec de l'urée et un FO liquide, la vinasse. Ils ont constaté une relation presque linéaire entre la quantité de paille et les émissions de N_2O et de CO_2. Toutefois, dès 2001 (Chantigny et al.) il a été mis en évidence, dans les interactions entre la paille (orge) et les FO (lisier de porc), une synergie pour la minéralisation de C et un antagonisme pour les émissions de N_2O, probablement dû à une immobilisation de l'azote par la biomasse du sol croissante. Il est important de mentionner que les possibles interactions ont été peu discutées dans ces études et n'ont pas permis de tirer des conclusions générales sur les effets de la combinaison des résidus de culture et des FO de natures physico-chimiques contrastées.

3. Projet de recherche

3.1. Objectifs

Face à l'intérêt croissant porté sur les paillis de canne à sucre pour la valorisation dans des filières de l'énergie, mais aussi aux besoins de porter un regard critique sur les préconisations des pratiques agricoles qui encouragent le paillage et le recyclage des déchets organiques en agriculture, nous nous proposons d'étudier les devenirs de C et de N au sein d'un système couplé d'apport en paille et FO. Le manque d'information dû au nombre limité de références scientifiques sur les interactions possibles liées à l'usage de FO, de différents types, en mélange avec de la paille, notamment de canne à sucre, nous a amené à considérer l'impact potentiel de ce type de systèmes sur l'environnement, notamment en termes d'émissions de GES et de stockage de carbone. Les objectifs de ce travail sont donc 1) de comparer l'effet des fertilisants organiques de nature contrastée, comme une boue de STEP sèche riche en azote organique et un lisier de porc liquide riche en ammonium et en carbone organique facilement disponible, sur la décomposition d'un mulch de paille de canne à sucre, et 2) d'évaluer les potentialités d'émission de GES des mélanges mulch-FO en fonction de la quantité de paille laissée après la récolte des tiges usinables de canne à sucre.

3.2. Question scientifique et hypothèses de recherche

La question générale de la thèse peut être formulée de la manière suivante :

Est-ce que la substitution de l'engrais N minéral par des fertilisants organique recyclés, dans un système de conservation complet ou partiel de la paille de canne à sucre laissée au sol, a un impact sur la disponibilité de l'N pour la décomposition de la paille et la culture de canne à sucre, sur les émissions de GES et sur le stockage de C à court terme dans le sol ?

Cette question peut se décliner en trois questions spécifiques :

- Quel est l'effet de la quantité de paille sur la cinétique et l'amplitude des émissions de GES ?
- Quel est l'effet du type de PRO sur les émissions de GES ?
- Quel est l'effet des mélanges paillis-PRO sur les émissions de GES et la décomposition du C, contribuant à un stockage net ou des émissions nettes de C ?

Ces questions visent à comprendre les différences entre :

- les additions de N organique ou N de synthèse ;
- les types de fertilisants organiques apportant peu ou beaucoup d'azote disponible ;
- les quantités de résidus de culture laissées au sol ;
- les rapports C:N des mélanges obtenus par la combinaison de la paille de canne à sucre et des fertilisants organiques testés ;

constituant ainsi les quatre hypothèses qui sont formulées dans le tableau 1 ci-dessous.

***Tableau 1** : Hypothèses testées dans la thèse*

<table>
<tr><td rowspan="3">Hypothèse 1</td><td colspan="4"><i>N fertilisant organique vs N fertilisant de synthèse</i></td></tr>
<tr><td>CO_2 ↗</td><td>Stockage C ↗</td><td>N_2O ↘</td><td>N disponible ↘</td></tr>
<tr><td colspan="4">Explications avec la littérature

Les émissions de CO_2 seraient plus importantes avec les fertilisants organiques, en raison d’une quantité plus grande de C apporté par ces derniers (Carmo et al. 2013)
.
Les stocks de carbone dans le sol seraient plus importants avec les fertilisants organiques, car une plus grande quantité de C est apportée par ces derniers et la disponibilité de l’azote des fertilisants organiques est plus faible (Wardle 1992, Diacono and Montemurro 2010).

Les émissions de N_2O seraient plus importantes avec le fertilisant de synthèse, car le dispositif expérimental est un système irrigué avec une pluviométrie importante, ce qui favorise la dissolution rapide et une disponibilité de l’azote plus grande, plus rapidement avec l’urée (Ball et al. 2004, Meijide et al. 2007).

Le N disponible, ou N minéral (NH_4^+ et NO_3^-) serait moins abondant dans le sol avec les fertilisants organiques, en raison de la stabilité chimique et de la protection physique de l’azote présent sous forme organique (Clark et al. 1998, Poudel et al. 2002, Herencia et al. 2007).</td></tr>
<tr><td rowspan="3">Hypothèse 2</td><td colspan="4"><i>Boue de STEP séchée vs Lisier de porc</i></td></tr>
<tr><td>CO_2 ↘</td><td>Stockage C ↗</td><td>N_2O ↘</td><td>N disponible ↘</td></tr>
<tr><td colspan="4">Explications avec la littérature

Les émissions de CO_2 seraient plus importantes avec le lisier de porc, car c’est un fertilisant liquide, contenant une concentration de carbone dissout plus importante (Tejada and Gonzalez 2005, Carmo et al. 2013).

Par conséquent, le stock de carbone serait plus important avec la boue de STEP séchée, car les pertes de CO_2 pour ce fertilisant seraient moins importantes que pour le lisier de porc.

Les émissions de N_2O seraient plus importantes avec le lisier de porc, car c’est un fertilisant organique riche en $NH_4^{+,}$ qui est rapidement transformable en NO_3^- dans le sol et représente le précurseur majeur de la dénitrification (Allen et al. 2010, Nguyen et al. 2014).

Le N disponible dans le sol , ou N minéral (NH_4^+ et NO_3^-) serait plus abondant avec l’application du lisier de porc, riche en N minéral (Aguilera et al. 2013).</td></tr>
</table>

<table>
<tr><td rowspan="3">Hypothèse 3</td><td colspan="4">Grande quantité de mulch vs Petite quantité de mulch</td></tr>
<tr><td>CO_2 ↗</td><td>Stockage C ↗</td><td>N_2O ↗</td><td>N disponible↘</td></tr>
<tr><td colspan="4">Explications avec la littérature

Les émissions de CO_2 seraient plus importantes avec une quantité de paille plus grande. Ceci serait dû à la plus grande quantité de C soluble apporté par la quantité plus importante de paille (Carmo et al. 2013).

Le stock de carbone serait plus grand lorsque la quantité de paille est plus grande, car la proportion de C stabilisé serait proportionnelle à la quantité de C récalcitrant apportée (Lou et al. 2011).

Les émissions de N_2O seraient plus importantes avec une quantité de paille plus grande. Ceci serait dû au maintien de l'humidité du sol par la paille et à la quantité de C biodégradable apporté plus élevée favorisant la dénitrification (de Oliveira et al. 2013, Wang et al. 2016b).

Le N disponible, ou N minéral (NH_4^+ et NO_3^-), serait plus élevé lorsque la quantité de paille est réduite, car l'immobilisation de N par la biomasse microbienne est plus importante lorsque la quantité de C est plus importante.</td></tr>
<tr><td rowspan="3">Hypothèse 4</td><td colspan="4">Rapport C:N bas vs rapport C:N élevé du mélange paillis-PRO</td></tr>
<tr><td>CO_2 ↗</td><td>Stockage C ↘</td><td>N_2O ↗</td><td>N disponible ↗</td></tr>
<tr><td colspan="4">Explications avec la littérature

Les émissions de CO_2 seraient plus importantes lorsque le C:N des mélanges est plus bas. Ceci a été démontré pour des résidus de coupe de C:N plus bas (Gentile et al. 2011).

Le stock de carbone serait plus grand lorsque le C:N des mélanges est plus grand, car la cinétique de décomposition est plus lente (Aerts 1997, Chen et al. 2014a).

Les émissions de N_2O seraient plus importantes lorsque le C:N du mélange est plus petit, car, pour une même quantité de résidus de canne à sucre apportés, la quantité de N apportée est plus grande dans ce cas. Les émissions sont corrélées positivement avec la quantité de N (Snyder et al. 2009).

Le N disponible, ou N minéral (NH_4^+ et NO_3^-), serait plus grand lorsque le C:N des mélanges est plus bas. Ceci a été démontré pour des résidus de coupe de C:N plus bas (Trinsoutrot et al. 2000).</td></tr>
</table>

3.3. Stratégie de recherche

Compte tenu des contradictions constatées dans la littérature vis-à-vis des effets des mélanges sur la dynamique de minéralisation du C et du N, nous avons étudié dans un premier temps les interactions en mélangeant de la paille de canne à sucre et des fertilisants organiques. Nous avons cherché à déterminer si ces interactions étaient positives (synergie) ou négatives (antagonisme). Ce travail a été fait grâce à des incubations en conditions contrôlées, dans des conditions de N non limitantes, avec des matières organiques incubées seules ou en mélange.

Dans un deuxième temps nous avons cherché à prédire, par la modélisation, les cinétiques de minéralisation des mélanges à partir des données obtenues avec les matières organiques incubées seules. Nous avons ainsi testé un modèle additif simple et un modèle mécaniste de transformation du carbone et de l'azote dans le sol – CANTIS (Garnier et al. 2003). Les relations interactives déterminées grâce à ces modèles, nous ont ensuite permis de proposer un mécanisme potentiel, dont la prise en compte dans les modèles augmente la précision de prédiction des dynamiques C et N.

En parallèle, nous avons effectué des essais au champ, pour suivre à la fois la décomposition de la paille de canne à sucre en mélange avec des fertilisants organiques et les émissions de GES. Dans le but de récolter le maximum de données, deux campagnes expérimentales ont été menées : la première s'est étalée sur une période de 120 jours après la récolte d'une canne à sucre en premier cycle cultural, et la seconde sur une canne à sucre en repousse, qui a duré 350 jours. Ce choix a permis de mesurer la décomposition de deux pailles de canne à sucre de qualité différentes (plus ou moins riches en N), mais aussi de quantités de paille représentatives différentes. La seconde campagne de mesures nous a permis d'étendre aussi la gamme de N apporté par les fertilisants organiques afin d'avoir des rapports C:N des mélanges plus contrastés et de compléter les mesures *in situ* par des mesures physico-chimiques, comme la température sous la paille, la quantité d'azote minéral du sol et l'humidité du sol au cours du temps.

Toutes les matières organiques utilisées ont été prélevées dans des gisements locaux. La paille de canne à sucre était celle obtenue après la récolte de la canne à sucre du site expérimental du SOERE PRO de la Réunion, sur la station expérimentale du CIRAD à La Mare. Les boues de STEP ont été fournies par la station d'épuration du Grand Prado, qui traite les eaux usées de la ville de Saint-Denis, de Sainte-Marie et Ste Suzanne à la Réunion. Le lisier de porc épandu provient des élevages locaux de producteurs porcins (Fig. 5).

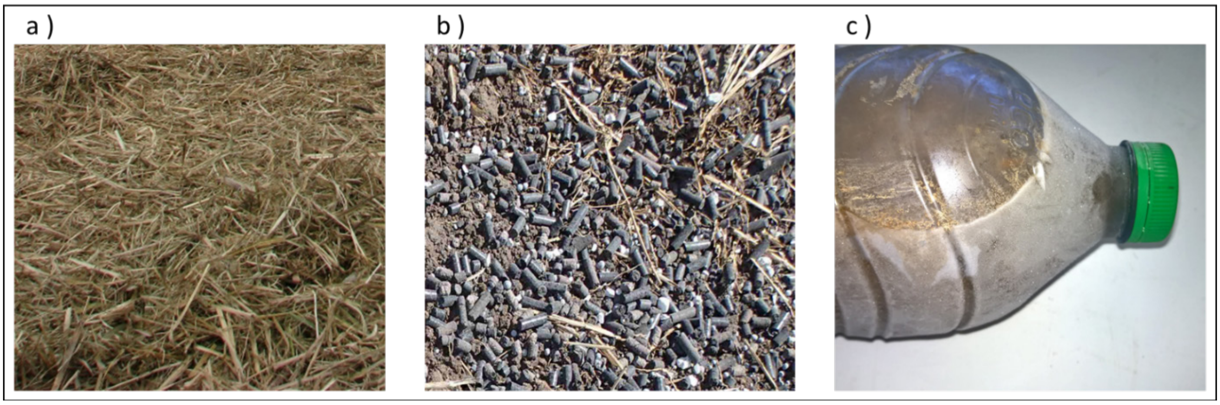

Figure 5 : Matières organiques utilisées pour évaluer l'impact des mélanges de paillis de canne à sucre (a) et de fertilisants organiques de qualités différentes, comme les boues de STEP séchées et chaulées (b) et le lisier de porc liquide (c)

4. Organisation du manuscrit

Ce manuscrit ne rend pas compte du déroulement chronologique des expérimentations et est structuré en 4 chapitres, dont 3 ont fait ou feront l'objet de publications dans des revues à comité de lecture internationales. Ces 3 chapitres sont rédigés en anglais. Compte tenu de ce choix de structure sous forme d'articles et afin d'assurer la compréhension de l'ensemble de la thèse, des résumés courts sont présentés au début de chaque chapitre.

Le chapitre 1 concerne les recherches d'interaction entre les matières organiques en mélange, réalisé grâce à des expérimentations en conditions contrôlées. Ce chapitre intègre l'approche par modélisation qui permet de discuter des processus impliqués dans les relations d'interaction éventuelles. Il a été publié dans la revue European Journal of Soil Science sous le titre « Investigating interactions between sugarcane straw and organic fertilizers recycled together in a soil using modelling of C and N mineralization ».

Le chapitre 2 est entièrement consacré aux expérimentations au champ sur une période de 4 mois après la récolte d'une canne à sucre après plantation. Ce chapitre traite des conséquences de l'application d'une quantité de paille différente et de fertilisants de qualités différentes. Il a été publié dans la revue BioEnergy Research sous le titre « Application of N fertilizer to sugarcane mulches : consequences for the dynamics of mulch decomposition and CO_2 and N_2O fluxes ».

Le chapitre 3 est aussi consacré aux expérimentations au champ sur une période de 1 an après la récolte d'une canne à sucre après repousse cette fois-ci. Ce chapitre traite des conséquences de l'application d'une quantité de paille différente et de fertilisants de qualités différentes, de quantités différentes et met en relation les dynamiques de minéralisation du carbone et de l'azote avec des variables physicochimiques du sol, telles que l'azote minéral, la température et l'humidité il sera soumis dans une revue à comité de lecture prochainement. Ce chapitre contient des figures annexes, situées juste après.

Le dernier chapitre (4) propose une synthèse qui conclut ce travail. Il fait un rappel des objectifs et des hypothèses et agrège les résultats, en fournissant une discussion générale. Il a aussi pour vocation de formaliser les connaissances acquises, en essayant de fournir des indications pour une meilleure gestion des systèmes d'apport couplé paille/fertilisant organique. Ce chapitre pointe aussi sur les perspectives de recherche qui s'ouvrent à partir des connaissances apportées.

Une liste des figures est présentée au début de chaque chapitre.

Une liste des références bibliographiques citées dans le texte est présente à la fin de chaque chapitre.

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CHAPITRE 1

II. Étude des interactions entre la paille de canne à sucre et les fertilisants organiques recyclés ensemble dans le sol. Modélisation de la minéralisation C et N

Résumé

L'apport de fertilisants organiques dans les sols est une alternative intéressante pour substituer la fertilisation minérale, mais on ignore encore comment leur recyclage conjoint avec les résidus de culture influe sur le devenir du carbone (C) et de l'azote (N) apportés. Par conséquent, nous avons analysé l’effet de l’ajout de fertilisants organiques et de paille de canne à sucre sur la minéralisation du C et du N. Nous avons incubé de la paille de canne à sucre (S), du lisier de porc (PS) et de la boue sèche de station d'épuration (DS), séparément et en mélange (PS-S et DS-S) à 28°C, pendant 182 jours. Pour discuter des interactions, nous avons utilisé un modèle d'additivité simple basé sur les mesures effectuées et un modèle mécaniste pour les transformations C et N dans le sol (CANTIS). Les deux modèles ont surestimé la minéralisation du C et n’ont pas prédit correctement la minéralisation du N des deux mélanges. Les différences entre les valeurs observées et celles qui étaient attendues, calculées avec les modèles, étaient négatives pour la minéralisation de C, indiquant une interaction antagoniste pour les mélanges. Les limitations imposées à la décomposition du C pourraient résulter de nombreux facteurs, tels que le « priming effect » (PE) ou la limitation de l'accessibilité du N, qui ne sont pas pris en compte par CANTIS. Sous l’hypothèse que le PE induit par la minéralisation d'un mélange n'était pas significativement différent du PE induit par la minéralisation des matières organiques incubées seules, nous avons appliqué un facteur de contact dans CANTIS, qui a permis d'ajuster correctement les cinétiques C et N pour les mélanges aux données mesurées. Le facteur de contact reflète l'effet des hétérogénéités de la distribution du C et du N à une échelle fine dans la matrice du sol, sur l'intensité de la décomposition microbienne. Une meilleure intégration des interactions entre les différentes sources de N et de C devrait être envisagée, afin de développer la modélisation en tant qu'outil précis pour la gestion d'un agro-écosystème.

Investigating interactions between sugarcane straw and organic fertilizers recycled together in a soil using modelling of C and N mineralization

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Abstract

The input of organic fertilizers into soils is an interesting option as a substitution for mineral fertilization, but how their interaction with crop residues affects the fate of added carbon (C) and nitrogen (N) in the soil is still poorly known. Therefore, we analyzed the effect of adding together organic fertilizer and straw on subsequent C and N mineralization. We incubated sugarcane straw (S), pig slurry (PS) and solid sewage sludge (DS) separately and in mixtures (PS-S and DS-S) at 28°C during 182 days. To discuss interactions, we used a simple additivity model based on measurements and a mechanistic model for C and N transformations in soil (CANTIS). Both models overestimated the C mineralization and did not correctly predict N mineralization of the two mixtures. The differences between observed and expected values calculated with the models were negative for C mineralization, indicating an antagonistic interaction in mixtures. The limitations for C decomposition might be the result of many factors, such as negative priming effect or limitation in N accessibility, which are not considered by CANTIS. We assumed that the priming effect induced by the mineralization of a mixture was not significantly different from the priming effect induced by the mineralization of the organic matters incubated alone. The use of a contact factor in CANTIS allowed the predicted C and N kinetics for the mixtures to be correctly fitted to measured data. It reflects the effect of fine-scale C and N distribution heterogeneities on the intensity of microbial decomposition. A better integration of the interactions between different N and C sources should be addressed to develop modelling as an accurate tool for agroecosystem management.

Highlights

- Experimental data and modelling allowed assessment of effects of mixtures on C and N mineralization.

- Mineralization of the organic material mixture was not the sum of the mineralization of each material.
- A contact factor reflecting the N-rich substrate accessibility was needed to predict C and N data.
- Pig slurry and sewage sludge mixed with straw showed antagonistic effects on mineralization.

KEYWORDS

antagonistic interaction, mixture, nitrogen accessibility, pig slurry, sewage sludge

List of figures

1. Introduction

One of the management options for the mitigation of greenhouse gases from agriculture is the use of organic fertilizers, which contributes to the recycling of organic wastes while minimizing synthetic fertilizer inputs (Saj, Torquebiau, Hainzelin, Pages, & Maraux, 2017). The benefits of organic fertilizers for C sequestration have long been recognized (Bordovsky, Choudhary, & Gerard, 1999; Bremer, Janzen, & Johnston, 1994; Campbell & Zentner, 1993; Ogle, Breidt, & Paustian, 2005). Thus, over the last two decades, considerable efforts have been made to correlate the biodegradability potential with the physicochemical indicators that characterize the various organic fertilizers (Jimenez, Lei, Steyer, Houot, & Patureau, 2017). There has been keen interest in N mineralization with organic fertilizer to better adapt inputs to the crop N requirements (Lashermes et al., 2010).

The effect of interactive relations between N-rich organic matter, such as organic fertilizer or legume crop residues, and C-rich organic matter, such as most crop residues, on the C budget and N mineralization can be synergistic, antagonistic or additive (Meier & Bowman, 2010). Nitrogen immobilization is generally observed when a complex organic C source with high C:N is added because of increasing heterotrophic microbial N demand during C assimilation, to fulfil stoichiometric requirements. In the past, most authors have studied these interactions occurring in mixtures of crop residues of various plant species or various plant organs, and very few studies have investigated the relation in mixtures composed of crop residues and organic fertilizers. Redin et al. (2014), Berglund and Ågren (2012) and Pérez-Harguindeguy, Blundo, Gurvich, Diaz, and Cuevas (2008) found that in most situations of N-rich and C-rich plant residue mixtures, a synergistic effect of the mixture on C mineralization occurred compared with the decomposition of a residue alone, due to the supply of N from one of the elements of the mixture to the other, which was N-limited. For fertilizer inputs, Chantigny, Rochette, Angers (2001) found a strong interaction between C-rich barley straw residues and N-rich pig slurry when simultaneously incorporated into the soil, with the pig slurry stimulating the mineralization of straw-C. Conversely, Aita, Recous, Cargnin, da Luz, and Giacomini (2012) found the same proportion of straw-C mineralized when the straw was incorporated into the soil alone or with pig slurry. Giacomini et al. (2015) also found that C mineralization was similar whether oat straw was incorporated into the soil alone or with sewage sludge. Therefore, the pattern of interaction between organic fertilizers and crop residues added together to a soil is unclear. Heterogeneity in biochemical and physical characteristics of the organic matters added together could, therefore, explain the inconsistency in the interactive effects of mixtures on C and N mineralization.

To better understand and predict C and N turnover in soils, many models were developed to describe kinetics and predict net mineralization of crop residues and/or organic fertilizer (Garnier et al., 2003; Iqbal, Garnier, Lashermes, & Recous, 2014; Lashermes et al., 2010; Luo et al., 2014; Nicolardot, Recous, & Mary, 2001; Pansu, Thuriès, Larré-Larrouy, & Bottner, 2003; Rinaldi, Scalcione, Perniola, Maddaluno, & Garofalo, 2017). Most of those models can predict the fate of C and N from single organic inputs, whereas identifying a universal set of parameters able to predict the dynamics of all types of organic matter (Lashermes et al., 2010) or of mixtures of different types of organic matter, is still difficult. Chalhoub et al. (2013) simulated C and N dynamics after compost application in the field using the PASTIS model and included a “contact factor” to simulate the local N limitation induced by N-poor composts. The model uses a multiplicative function of decomposition to describe the limitation of C decomposition rate that was calculated from the potential contact area of the residue with the soil.

Thus, the main objective of our study was to quantify the effect of the interactions between sugarcane straw and organic fertilizer applied jointly to the soil, on C and N mineralization. To achieve this objective, we incubated, under controlled conditions, sugarcane straw, pig slurry and sewage sludge alone or in mixtures and studied the additive, antagonistic or synergistic response of the mixtures. Then we used modelling with a soil C and N transformation model (i.e. CANTIS by Garnier et al. (2003) and Chalhoub et al. (2013)) to interpret and quantify the processes involved in the interactions observed between these organic products.

Table 1 : *Explanation of the abbreviations*

Abbreviation	Explanation
OF	Organic fertilizer
Treatments	
S	Sugarcane straw
PS	Pig slurry
DS	Solid sewage sludge
PS-S	Pig slurry and straw mixture
DS-S	Solid sewage sludge and straw mixture
Model pools / Parameters	
HOM	Humified organic matter
SOL_{SOIL}	Soluble organic matter in soil
FOM	Fresh organic matter
AUB	Autochthonous biomass
ZYB	Zymogenous biomass
LIC	Lignine-like fraction of fresh organic matter
HEM	Hemicellulose-like fraction of fresh organic matter
CEL	Cellulose-like fraction of fresh organic matter
SOL_S	Soluble fraction of fresh organic matter with slow rate of decomposition
SOL_F	Soluble fraction of fresh organic matter with fast rate of decomposition
K_{MZ}	Contact factor

2. Materials and Methods

2.1. Characterization of Soil and Organic Materials

The soil used for the incubation experiments was a nitisol (Working Group Wrb, 2015) collected in the surface layer (0-20 cm) from a field of long-term experiments at the experimental station of La Mare at Sainte-Marie, Reunion Island (20°54’12.2” S, 55°31’46.6” E). The nitisol is characterized by 40% clay, 34% fine silt, 11% coarse silt and 15% sand, CEC of 108.6 mmol kg^{-1}, soil organic carbon (SOC) of 16 g C kg^{-1}, 1.8 g N kg^{-1}, of which 10.2 mg was NO_3^--N kg^{-1} and 3.6 mg was NH_4^+-N kg^{-1}, 0.09 g P kg^{-1} and pH_{H2O} of 6.2. The soil was prepared according to the French standard FD-U-44-163 (AFNOR, 2016a). Briefly, the soil was airdried until reaching stable moisture content (half of the water-holding capacity) and then sieved to 2 mm, removing any visible particulate organic matter. Soil microbial biomass of 184.2 g C kg^{-1} was determined by the fumigation-extraction method described by Vance, Brookes, and Jenkinson (1987).

Three organic materials were incubated: sugarcane straw (mixture of leaves and tops), pig slurry and solid sewage sludge. The straw was collected from the mulch after harvesting the second ratoon of sugarcane (R579 variety) on November 2015. Straw was oven-dried at 60°C for 72 hr, until reaching stable weight, cut to 5-mm size particles and stored in an air-cooled room at 22°C before incubation. Pig slurry was collected directly from the anaerobic storage facility of a hog producer and was stored at -20°C before incubation. The pig slurry was applied in the liquid state after thawing. Digested and limed pellets of sewage sludge were oven-dried (60°C, 72 hr) and crushed to 1-2 mm granules before incubation. All these pretreatments were chosen in order to mimic a more realistic contact with the soil, as in the field.

Total C of all organic materials was determined by dry combustion elemental analysis with the Dumas method (NF ISO 10694:1995). Total N for sugarcane straw and sewage sludge was determined by dry combustion elemental analysis using the Dumas method (NF ISO 13878:1998), and by the Kjeldahl method for pig slurry. Ammonium content was measured by alkalization followed by distillation and then titration for all organic materials. Biochemical fractioning (see Table 1 for Abbreviations) was performed on organic materials for the soluble fraction, neutral detergent fibre (NDF), acid detergent fibre (ADF) and acid detergent lignin (ADL). The Van Soest soluble fraction (SOL) and hemicellulose-like (HEM), cellulose-like (CEL) and lignin-like (LIC) fractions were determined using the Van Soest methods described in the French standard FD-U-44-162 (AFNOR, 2016b). Ash was determined by calcination (550°C) followed by a gravimetric measurement. The characteristics of the organic materials are shown in Table 2.

Table 2 : *Concentrations on a wet basis of dry matter (DM), total organic matter (total OM), total organic carbon (TOC), total N (total N) and NH_4^+-N and C content of the biochemical fractions on a dry basis for the organic materials used in the incubations*

Variables	**Unit**	**Sewage sludge**	**Pig slurry**	**Sugarcane straw**
Concentration on a wet basis				
DM	g kg^{-1}	907	69	891
Total OM	g kg^{-1}	543	48	820
TOC	g kg^{-1}	329	25	465
Total N	g kg^{-1}	51	4.1	5
NH_4^+-N	g kg^{-1}	4	2.3	n.d.
C:Ntotal	unitless	6.5	6	95
pH	unitless	12.3	7.4	n.d.
C content of the biochemical fractions on a dry basis				
Soluble fraction	g 100 g^{-1}	49.1	31.9	16.7
Hemicellulose-like	g 100 g^{-1}	6.4	8.8	32.3
Cellulose-like	g 100 g^{-1}	1.6	16.2	38.4
Lignin and cutin-like	g 100 g^{-1}	2.8	13.3	4.7
Ash	g 100 g^{-1}	40.1	29.8	7.9

n.d. = not determined.

2.2. Experiment

2.2.1. Incubation Conditions and Treatments

The experiment was conducted during 182 days at 28°C in a dark room. The incubation was performed on the mixtures of soil and each of the five treatments and on control soil (without organic material) according to the experimental design, complying with the French standard FD-U-44-163 (AFNOR, 2016a). The soil moisture was adjusted to 0.33 g H_2O g^{-1} dry soil during the incubation. This water content corresponded to the water-holding capacity (-31.6 kPa) as recommended by the FDU 44163 standard (AFNOR, 2016a). The treatments included the three organic materials incubated alone, pig slurry (PS), digested solid sewage sludge (DS) and sugarcane straw (S), two mixtures, namely, pig slurry with sugarcane straw (PS-S) and digested solid sewage sludge with sugarcane straw (DS-S), and a nonamended soil as the control. Each treatment had four replicates. The doses of organic materials were calculated to obtain non-limiting N in each treatment according to Recous, Robin, Darwis, and Mary (1995). The final C:N$_{organic}$ for the mixture treatments, DS-S and PS-S, were similar to one another and also to the C:N$_{total}$ of the S treatment in which mineral N was added in the form of KNO_3 to ensure non-limiting N conditions for straw decomposition. The final composition of the treatments is shown in Table 3.

The experimental design consisted of measuring CO_2-C, NO_3^--N and NH_4^+-N for the five amended treatments and for the non-amended soil (control). The decomposition of the organic materials was evaluated by measuring the respiration of CO_2-C on day 1, 2, 3, 7, 14, 21, 28, 49, 70, 90, 112, 135, 155 and 182. The apparent C mineralization from the organic products was calculated as the difference between CO_2 evolved from amended soils and CO_2 evolved from the control soil at each date. The priming effect (PE) was not measured directly but included in the "apparent mineralization". Cumulative C mineralization was obtained by summing the amount of CO_2-C evolved over the different time intervals.

The net mineralization of N was calculated by measuring the changes in NH_4^+-N and NO_3^--N contents in the soil from day 0 (1 hr after mixing soil and organic materials) to 1, 3, 7, 14, 28, 49, 90, 135 and 182 days of incubation. The net N mineralization from an organic product was calculated by the difference between the amended soil and the control soil at each date.

Table 3 : *Physicochemical characteristics of the treatments used for the incubations*

Treatment	**Organic material**	**C**	**Total N**	**NH_4^+-N**	**NO_3^--N**	**Organic N**	**C:N$_{total}$**	**C:N$_{organic}$**
		g kg^{-1} dry soil					**unitless**	**unitless**
S	Sugarcane straw + KNO_3	2.1	0.02	n.d.	0.06 (added)	0.02	24.9	94.6
PS	Pig slurry	4.4	0.73	0.42	n.d.	0.32	6.0	13.8
DS	Sewage sludge	2.5	0.38	0.03	n.d.	0.35	6.5	7.1
PS-S	Pig slurry	3.6	0.59	0.34	n.d.	0.26	11.8	25.9
	Sugarcane straw	3.8	0.03	n.d.	n.d.	0.03		
DS-S	Sewage sludge	1.8	0.28	0.02	n.d.	0.26	21.8	23.4
	Sugarcane straw	5.6	0.06	n.d.	n.d.	0.06		

n.d. = not determined.

2.3. Statistical Analyses

The normality of the distribution and the homoscedasticity were verified using Shapiro and Wilk normality and Bartlett's tests, respectively. When those conditions were met, a one-way ANOVA test was applied. For some treatments, a logarithmic or square root data transformation was performed to meet those conditions before applying an ANOVA test. All statistical analyses were performed with R software (v3.2.2, R Core Team, 2015, R Foundation for Statistical Computing, Vienna, Austria), powered by RStudio (v1.0.143–2009-2016, RStudio Team, 2015, Integrated Development for R, RStudio Inc., Boston, MA).

2.4. Simple Additivity Model

First, a simple additivity model based on experimental data was created to test whether the mineralization kinetics of the mixture treatments were the weighted sum of the mineralization kinetics of the two organic materials incubated alone. This model was based on the kinetics of the apparent mineralization measured from the three organic materials incubated alone (i.e. pig slurry, sewage sludge and sugarcane straw). The mineralization was expressed as a net fraction of initially added C or N (g g^{-1} added). Therefore, the corresponding C or N mineralization for the mixture treatment (AB) was the sum of C or N mineralization measured on day i for each organic material incubated alone (A or B) multiplied by their corresponding C or N contributions in the mixture treatment (α and β) (Equation 1):

$$AB_i = \alpha A_i + \beta B_i \qquad (1)$$

The values of the coefficients for the respective contributions of C and N in the mixtures are in Table 4. The additive model implicitly recognizes that the PE is included in the mineralization terms. We assumed that the PE induced by the incubation of the mixture was not significantly different from the weighted contribution of PE induced by the two organic matters incubated alone.

Table 4 : *Coefficients of C and N respective contributions used to calculate the expected mineralization for the organic materials (S) sugarcane straw, (DS) sewage sludge, and (PS) pig slurry in the mixture treatments (DS-S) sewage sludge with sugarcane straw and (PS-S) pig slurry with sugarcane straw.*

Treatment	Contribution coefficient	C	N
DS-S	α (S)	0,76	0,18
	β (DS)	0,24	0,82
PS-S	α (S)	0,51	0,11
	β (PS)	0,49	0,89

2.5. The CANTIS Model

CANTIS (see Fig. 1) is a mechanistic model simulating C and N transformations in soils (Garnier et al., 2003). The model has been used to simulate C and N mineralization for crop residues (Iqbal et al., 2014) and for manures and composts (Chalhoub et al., 2013). Soil organic matter is divided into five compartments: humified organic matter (HOM), soluble organic matter of the soil (SOL_{soil}), fresh organic matter (FOM) and two microbial biomasses, namely, autochthonous biomass (AUB) and zymogenous biomass (ZYB). Each compartment is expressed as mg C kg^{-1} dry soil and as mg organic N kg^{-1} dry soil using its N:C. The FOM is the input from organic materials that are to be incubated and is divided into three fractions: SOL, HEM+CEL and LIC. SOL is further divided into two fractions, namely, a fast decomposable soluble fraction, SOL_f, and a slow decomposable soluble fraction, SOL_s, which was proposed by Chalhoub et al. (2013) to consider the recalcitrant fraction produced by microbial degradation occurring during the composting process (Peltre, Dignac, Derenne, & Houot, 2010). The three pools of FOM, namely, SOL, HEM+CEL and LIC, feed the SOL_{soil} pool by enzymatic hydrolysis. The SOL_{soil} fraction is used for the growth of ZYB, whereas the HOM fraction leads to AUB growth. HOM is further fed by both ZYB and AUB recycling.

Microbial growth of AUB and ZYB is from the decomposition of HOM and SOL, respectively, using assimilation yields (Y_H and Y_S, respectively). Microbial death is a function of the AUB or ZYB size using the decomposition rates of microbial biomasses (K_A and K_Z, respectively), and a part of the dead biomass is recycled into HOM, by considering the humification coefficients H_A and H_Z, respectively. The other part of the dead biomass ($1-H_A$ and $1-H_Z$) is recycled into living biomass AUB and ZYB by considering assimilation yields Y_A and Y_Z, respectively. To account for the role of lignin in protecting cellulose and hemicellulose molecules (Corbeels, Hofman, & Van Cleemput, 1999; De Neve & Hofman, 1996; Iqbal et al., 2014;Moorhead, Lashermes, Sinsabaugh, & Weintraub, 2013), a retardation factor for HEM+CEL decomposition as a function of the lignocellulosic index was used (Iqbal et al., 2014).

Degradation of HOM by AUB and each FOM fraction (C_i) and SOL_{soil} by ZYB follows the first-order kinetics function of the decomposition rate of each fraction (k_i), moderated by the moisture limitation (f_W), the temperature limitation (f_T) and the N limitation (f_N) (Equation 2), as described in Garnier, Néel, Mary, and Lafolie (2001) :

$$\frac{dC_i}{dt} = -k_i . C_i . f_T . f_W . f_N . f_B \qquad (2)$$

The contact limitation function f_B depends on the “contact factor” K_{MZ} and ZYB (Equation 3):

$$f_B = \frac{B_Z}{K_{MZ}+B_Z} \qquad (3)$$

The C decomposition might follow strictly a first order ($K_{MZ} = 0$, $f_B = 1$) or might express limitation ($K_{MZ} > 0$, $f_B < 1$).

The K_{MZ} factor was first introduced by Hadas, Parkin, and Stahl (1998) using a reverse Monod equation to consider that microbial biomass can be the limiting factor for decomposition. This limitation may be due to microscopic heterogeneities such as the distribution of organic C and N that affect the accessibility of substrates (C of organic matter and N of soil) for microorganisms. Angers and Recous (1997) and Iqbal et al. (2014) showed that dispersed added organic matter distribution in soil due to small particle size of residues leads to faster decomposition than a poor distribution due to large particle size, in non-limiting mineral N conditions. Iqbal et al. (2014) found that K_{MZ} was correlated with the specific surface area of maize residue particles. Moreover, using a three-dimensional model, Garnier et al. (2008) showed that the limitation of decomposition was also a due to a limitation of the N diffusion towards the sites of degradation when the residue organic matter was concentrated, rather than only a limitation of residue colonization by microorganisms.

Nitrogen kinetics are linked to C kinetics when considering the C:N, as described by Garnier et al. (2001). The nitrification rate (V_N) depends on the maximal nitrification rate (V_{max}), the amount of NH_4^+-N (N_A), and the affinity constant between NH_4^+ and nitrifying bacteria (K_N) and is moderated by the moisture limitation (f_W) and the temperature limitation (f_T) (Equation 4):

$$V_N = \frac{V_{max}\,N_A}{K_N + N_A} f_T' f_W \qquad (4)$$

CANTIS does not simulate the PE. This is one of its limitations. Some models simulate the PE but the results often differ vastly from experimental data (Guenet, Moyano, Peylin, Ciais, & Janssens, 2016) because the influence of the quality of added organic matter on the PE is very complex (Kuzyakov, Friedel, & Stahr, 2000). CANTIS hypothesizes that the “apparent mineralization” comes only from the mineralization of the FOM. Consequently, if there is a PE, then it is implicitly taken into account in the mineralization of the FOM.

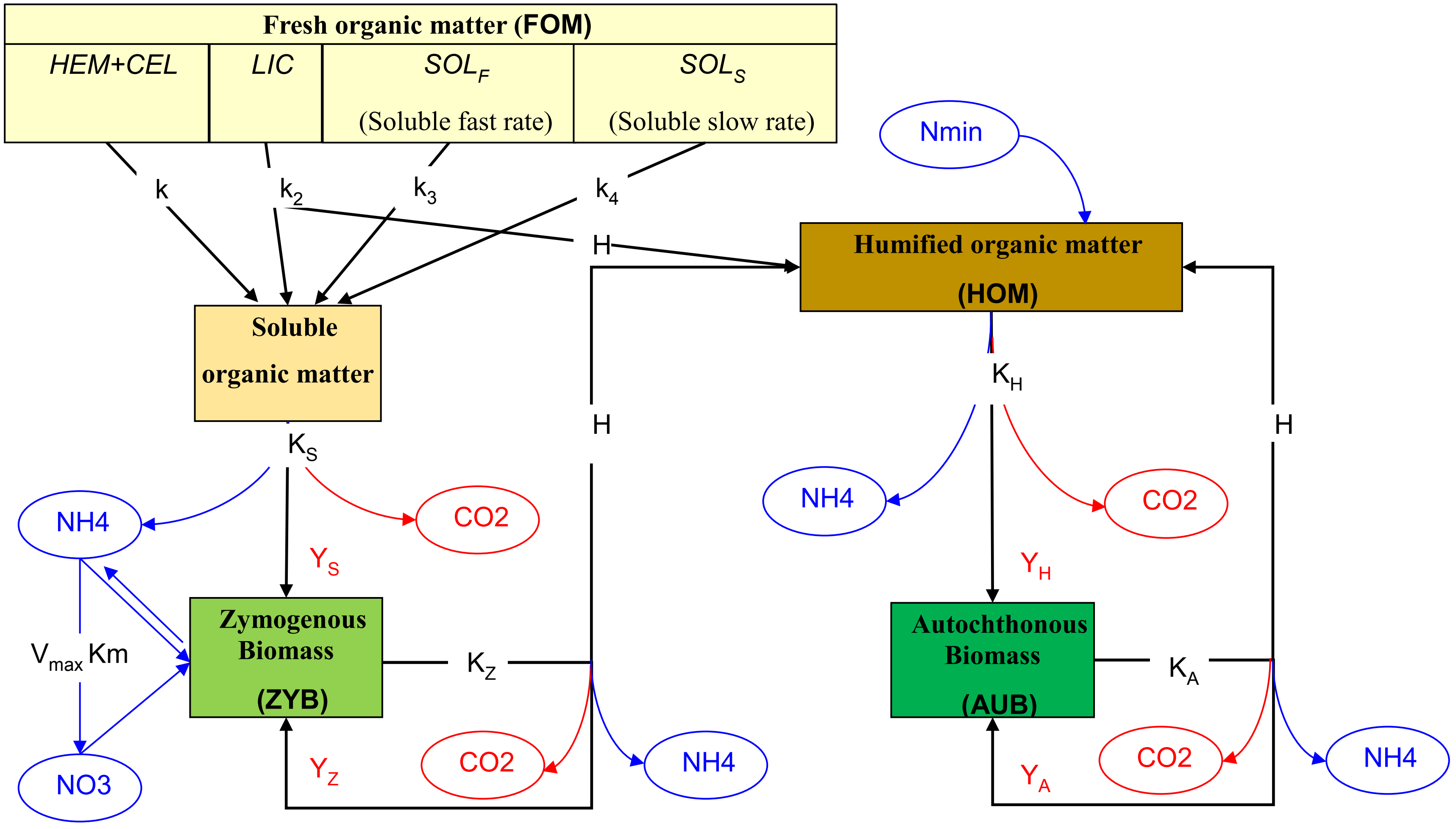

Figure 1 : General scheme of the CANTIS model (adapted from Chalhoub et al., 2013). The parameters are presented in Table 5

2.6. Modelling Strategy

2.6.1. Calibration of the CANTIS Model

The CANTIS model was calibrated with the CO_2-C, NO_3^--N and NH_4^+-N for S, PS, DS and soil control using a unique set of parameters. Parameters for biological transformations were optimized in two steps (Table 5). In the first step, we calibrated the death rate constant of autochthonous biomass, K_A, and the decomposition rate constant of HOM, K_H, using the experimental mineralization curve of C and N from the control soil.

In the second step, six parameters were optimized with the model using the experimental mineralization curve of the organic materials incubated alone. First, the death rate of zymogenous biomass (K_Z), the decomposition rates of HEM+CEL and LIC compartments (k_1 and k_2) and the humification coefficient (H_Z) were optimized from CO_2 mineralization curves of the three treatments (Fig. 2). The sizes of the soluble organic matter compartments SOL_F and SOL_S with their C:N were also optimized with CANTIS from the measurement of the total SOL fraction (Table 6). Because the CANTIS model was not used with soil or organic materials that initially contained significant quantities of ammonium, we optimized the maximal nitrification rate (V_{max}) from NH_4^+ and NO_3^- curves. The other parameters used for our simulations were previously optimized for a large number of crop residues (Iqbal et al., 2014).

CANTIS was linked to inverse methods based on the Levenberg-Marquadt algorithm. All the fittings were carried out using this method.

Table 5 : *Values used for the biological transformation parameters in the CANTIS model*

Symbol	Description	Unit	Value retained	Source
Decomposition of fresh organic matter (FOM), calibrated from organic materials incubated alone				
k_1	Decomposition potential rate of holocellulose-like compartment (HEM+CEL)	day^{-1}	0.015	Optimized
k_2	Decomposition potential rate of lignin-like compartment (LIC)	day^{-1}	0.0001	Optimized
k_3	Decomposition potential rate of soluble, fast decomposable fraction (SOL_f)	day^{-1}	0.25	Iqbal, 2013
k_4	Decomposition potential rate of soluble, slow decomposable fraction (SOL_S)	day^{-1}	0.0001	Chalhoub, 2013
K_S	Decomposition potential rate of SOL_{soil}	day^{-1}	1.49	Iqbal, 2013
K_Z	Death rate constant of zymogenous biomass (ZYB)	day^{-1}	0.15	Optimized
H_Z	Humification coefficient of zymogenous biomass (ZYB)	–	0.4	Optimized
V_{max}	Maximum nitrification rate	$mg\ N\ kg^{-1}\ day^{-1}$	4	Optimized
Y_S, Y_Z,	C assimilation yields of SOL and ZYB	$gC\ g^{-1}C$	0.55	Iqbal, 2013
H_L	Humification coefficient for LIC		0	Iqbal, 2013
K_{MZ}[a]	Contact factor for organic matter alone		0	Iqbal, 2013
Decomposition of humified organic matter (HOM), calibrated from control soil				
K_H	Decomposition potential rate of HOM	day^{-1}	0.00013	Optimized
K_A	Death rate constant of autochthonous biomass (AUB)	day^{-1}	0.006	Optimized
Y_H, Y_A,	C assimilation yields of HOM and AUB	$gC\ g^{-1}C$	0.55	Iqbal, 2013
H_A	Humification coefficient for AUB		0	Iqbal, 2013

The optimized values were obtained from incubations of organic materials alone, except for the contact parameter K_{MZ}, which was re-estimated for incubations of organic materials in mixtures.

[a] The contact factor K_{MZ} optimized for the PS-S mixture and DS-S mixture is equal to 60 and 130, respectively.

2.6.2. Prediction of the Mineralization from Mixture Treatments

The calibrated model from the three organic materials incubated alone and the control soil was tested for predicting the C and N mineralization from mixture treatments (i.e. DS-S and PS-S).

2.6.3. Statistical Criteria for Model Calibration and Evaluation

The optimization procedure aimed to minimize the relative root mean square error (RRMSE) between measured and predicted values for CO_2-C, NO_3^--N and NH_4^+-N per treatment, according to Equation 5:

$$RRMSE(\%) = \frac{100}{\bar{m}} \sqrt{\frac{\sum_{i=1}^{N}(s_i - m_i)^2}{n}} \qquad (5)$$

where m_i is the measured value for CO_2-C, NO_3^--N or NH_4^+-N, s_i is the simulated value, and n is the number of observations per variable.

Evaluation of the model's correctness included calculation of the Nash-Sutcliffe model efficiency index (Equation 6). Perfect agreement between the simulated and the observed data is considered when $Ef = 1$. An $Ef \leq 0$ indicates that the model provides less accurate predictions than the mean value of the observed data:

$$Ef = 1 - \frac{\sum_{i=1}^{n}(y_{obs,i} - y_{calc,i})^2}{\sum_{i=1}^{n}(\bar{y}_{obs} - y_{obs,i})^2} \qquad (6)$$

where $y_{obs,i}$ and $y_{calc,i}$ are, respectively, the observed and predicted values at the i^{th} observed point, and $\bar{y}_{obs}$ is the mean of all observed values for each variable.

Finally, the mean difference, $\bar{D}$, was calculated to provide an indication of the bias in the simulations (Equation 7):

$$\bar{D} = \frac{\sum_{i=1}^{n}(y_{calc,i} - y_{obs,i})}{n} \qquad (7)$$

where $y_{calc,i}$ and $y_{obs,i}$ are, respectively, the predicted and observed values at the i^{th} observed point, and n is the number of observations per variable.

2.6.4. Sensitivity Analysis

A sensitivity analysis was conducted to identify the most sensitive parameters calibrated with CANTIS. The original set of parameters calibrated in this study was considered as reference values. The effect of the individual changes of six parameters was investigated on the simulated value of cumulative C-CO_2 mineralized and NO_3^- concentration for both DS-S and PS-S treatments. The sensitivity coefficient, σ_P, for the parameter, P, defined following Myint, Nirmalakhandan, & Speece (2007), was calculated for each of the parameters:

$$\sigma_P = \frac{1}{t_f}\int_0^{t_f} \frac{S_{P+\Delta P} - S_P}{S_P} dt \tag{8}$$

where t_f is the incubation time, $S_{P+\Delta P}$ is the simulated value predicted by the model when the value of the P parameter changed from the reference value by an amount ΔP, and S_P is the simulated value predicted by the model with the reference value for the P parameter. Incubation duration of 182 days and variation of 20% in the value of the P parameter were tested.

3. Results

3.1. Organic Materials Incubated Alone

3.1.1. Experimental Data

The cumulative apparent mineralization of sugarcane residue (S) was 65% of added C after 182 days (Fig. 2a). During the same period, only 27% of added C was mineralized in the PS treatment (Fig. 2d), which was not significantly different from that in the DS treatment (32% of added C) (Fig. 2g). The straw had a SOL fraction with high degradability and a large percentage of HEM+CEL with intermediate degradability (Table 2). Its C decomposition was in the range observed for other crop residues incubated in similar conditions (Redin et al., 2014). The organic fertilizers had much more of the SOL fractions, with low degradability (Table 6) and very few HEM+CEL (Table 2).

***Table 6 :** Proportion of C and N contents in fast decomposable soluble fraction (SOLf) and in slowly decomposable soluble fraction (SOLs) optimized for the CANTIS model, using measured mineralization for the organic materials incubated alone: sugarcane straw (S), sewage sludge (DS) and pig slurry (PS)*

	SOL_{total}		SOL_f		SOL_S	
Treatment	**%C**	**C:N**	**%C**	**C:N**	**%C**	**C:N**
	Measured		Optimized from SOL_{total}			
S	18	30	100	30	0	–
DS	82	7	45	7	55	7
PS	45	17	35	7	65	22
DS-S	34	15	87	17	13	4
PS-S	31	15	68	15	32	15

Consequently, their decomposition was lower than that of sugarcane residue. Most of the NH_4^+-N added to pig slurry and sewage sludge was entirely nitrified after 49 days (Fig. 2f and i). For these treatments, the final NO_3^--N concentration (Fig. 2e and h) exceeded the initial NH_4^+-N content, indicating that some organic N was mineralized during decomposition. For the S treatment, NH_4^+-N concentration remained stable during the entire incubation period, and the NO_3^- added as KNO_3 was immobilized (Fig. 2b and 2c) but was not depleted, indicating that mineral N availability was not limiting during decomposition. At the end of the incubation, NO_3^- concentrations were higher than those measured initially, particularly for PS and DS treatments.

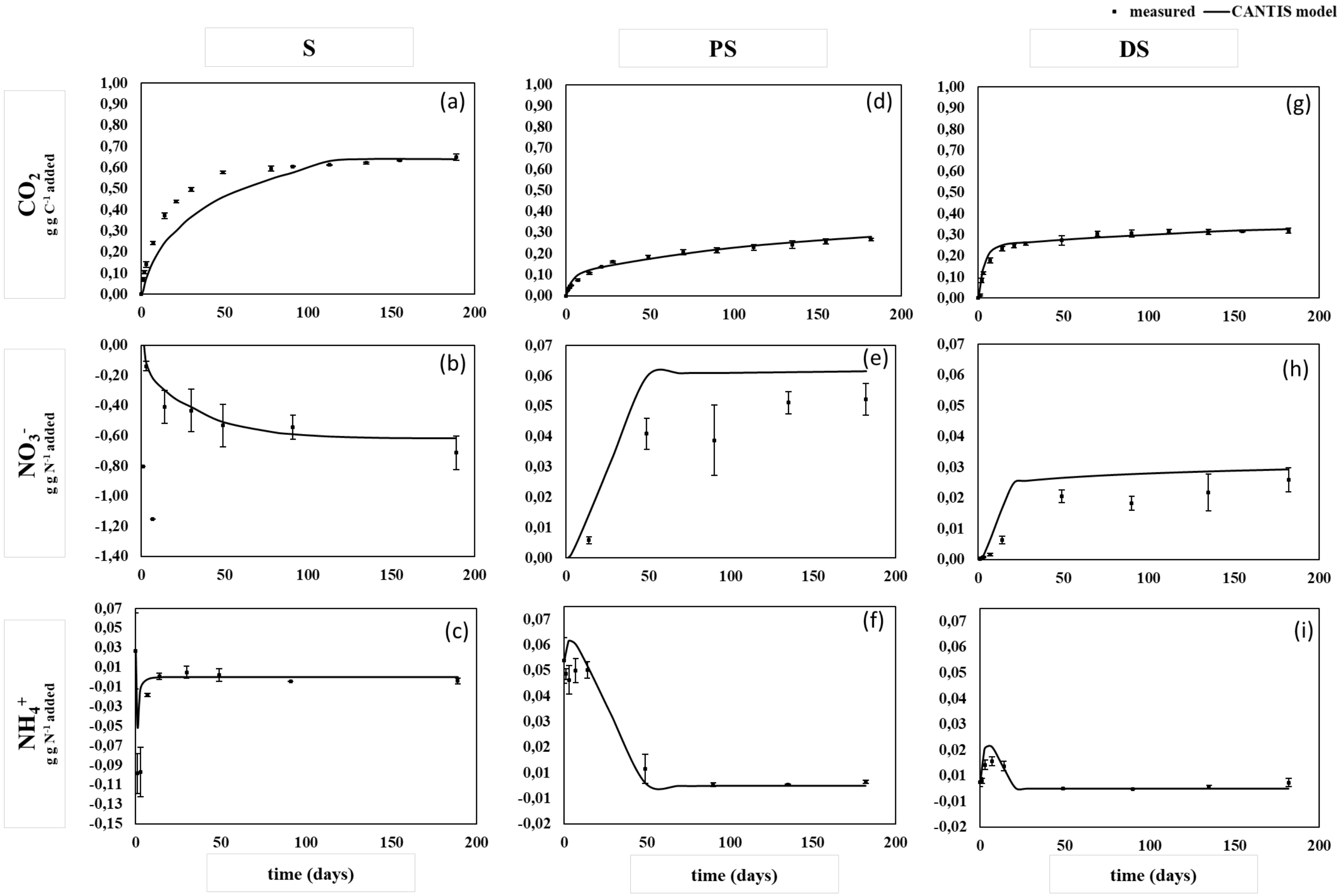

Figure 2 : Cumulative apparent mineralization of C and changes in NO_3^--N and NH_4^+-N concentrations in soil during incubation of organic materials alone: S, sugarcane straw (a, b, c); PS, pig slurry (d, e, f); DS, sewage sludge (g, h, i). The experimental data are the mean of four replicates and are the black circles (bars, the standard deviation), and the predictions with CANTIS are the black lines.

3.2. Calibration of the CANTIS Model: Single Set of Parameters

Most of the parameters of Table 5 were taken from Iqbal et al. (2014) and Iqbal (2013), who found a parameter set able to simulate a large amount of experimental data (with different soils and different crop residues). The decomposition rates of HEM+CEL and LIC, the death rate constant of zymogenous biomass K_Z, and the humification coefficient of zymogenous biomass H_Z were at the same order of magnitude as those found by Iqbal et al. (2014). Better prediction of NO_3^--N and NH_4^+-N contents was obtained by decreasing V_{max} for PS and DS treatments (Fig. 2f and i), compared with previous studies using the CANTIS model (Chalhoub et al., 2013; Garnier et al., 2001; Garnier et al., 2003; Iqbal et al., 2014). This modification was made because of the specificity of the added organic materials rich in NH_4^+-N compared to the organic materials of previous studies poor in ammonium.

Calibration of the CANTIS model provided a good prediction of all measured variables (CO_2, NO_3^- and NH_4^+) for the treatments with organic materials alone and the control soil, as can be seen from the Nash-Sutcliffe model efficiency indexes, which were all positive (Table 7). The Nash- Sutcliffe model efficiency index for CO_2-C was very close to 1 for all organic materials incubated alone. The model efficiency for the prediction of mineral N contents was good for PS and DS treatments. However, the prediction of NO_3^--N for PS (Fig. 2e) did not perfectly fit measured values. This discrepancy between the predicted values and the measured values was due to the use of a single set of parameters provided by a compromise that allowed the best possible prediction for all treatments, but which for some treatments and variables resulted in less good prediction. The low value of the efficiency index for NH_4^+-N of the S treatment (0.37 in Table 7) was due to a small variation over time of experimental data very close to the average.

Table 7 : *Statistical criteria used to evaluate the calibration procedure for control (CONT), treatments with organic materials incubated alone (sugarcane straw (S), pig slurry (PS) and sewage sludge (DS)), and treatments with organic materials incubated as mixtures with the single set of parameters (PS-S and DS-S) or with supplementary adjustment of the K_{MZ} parameter (PS-S_{corr} and DS-S_{corr})*

Organic inputs		**CONT**	**S**	**PS**	**DS**
Ef	CO_2	0.98	0.89	0.98	0.97
	NO_3^--N	0.99	0.92	0.75	0.65
	NH_4^+-N	0.50	0.37	0.89	0.24
RRMSE (%)	CO_2	10	19	8	9
	NO_3^--N	13	-16	55	57
	NH_4^+-N	71	-162	27	86
	RRMSE (%) mean	31	-53	30	51
$\bar{D}$	CO_2	-33	-87	-11	-8
	NO_3^--N	-2	-8	50	21
	NH_4^+-N	-2	-2	17	6

Mixture of organic inputs		**PS-S**	**PS-S_{corr}**	**DS-S**	**DS-S_{corr}**
Ef	CO_2	0.72	0.97	0.57	0.98
	NO_3^--N	0.80	0.87	-2.77	-2.28
	NH_4^+-N	0.98	0.98	-0.56	-0.52
RRMSE (%)	CO_2	28	8	39	8
	NO_3^--N	44	35	-213	-113
	NH_4^+-N	12	12	144	143
	RRMSE (%) mean	28	19	-10	13
$\bar{D}$	CO_2	241	-98	608	14
	NO_3^--N	-24	-18	-9	2
	NH_4^+-N	4	4	-6	-6

Note. RRMSE %: relative root mean square error per variable. RRMSE% mean is for all variables. Ef is the Nash-Sutcliffe model efficiency coefficient. $\bar{D}$ is the mean difference (in mg kg^{-1} of dry soil).

3.3. Organic Materials Incubated in Mixtures

3.3.1. Experimental Data

Mixing organic materials in PS-S and DS-S treatments with the same amount of added C led to C mineralization of 34 and 41% of added C, respectively (Fig. 3a and g). The amount of NO_3^- in the DS-S treatment was much lower than that in the PS-S treatment (Fig. 3b and h), which was due to the lower initial content of NH_4^+-N in sewage sludge than in pig slurry (Table 3). From Fig. 3c and i, NH_4^+-N was probably not adsorbed on the soil clay-humic complex, because most of the NH_4^+-N was available to be converted into NO_3^--N in the DS-S and PS-S treatments. From the statistics of Table 8, we see that the mineralization of the organic materials incubated alone was significantly different from the mineralization of both mixtures (with $p < 0.05$), except for the C mineralization of DS, which was not different from the C mineralization of DS-S ($p = 0.08$).

Table 8 : *Summary results from ANOVA for percentage of CO_2-C and NO_3^- on day 182 after incubation*

Treatments		**Sum of squares**	***F* ratio**	***p*-value**
PS-S/PS	CO_2	0.013	7.53	0.03
	NO_3^-	0.0003	11.1	0.02
DS-S/DS	CO_2	0.013	4.41	0.08
	NO_3^-	0.002	187.4	<<0.01
S/PS-S	CO_2	0.187	113	<<0.01
	NO_3^-	0.178	58.2	<<0.01
S/DS-S	CO_2	0.116	37.9	<<0.01
	NO_3^-	0.331	59.36	<<0.01

N = 8, J = 4, K = 2. S: sugarcane straw; PS: pig slurry; DS: sewage sludge.

3.4. Additivity Modelling

Calculated C mineralization for PS-S and DS-S was overestimated, indicating that the simple additivity model was not able to predict C mineralization of the mixtures correctly (Fig. 3a and g). Predicted NO_3^--N contents were higher than experimental values, particularly for the DS-S treatment.

3.5. CANTIS Modelling

Considering C mineralization of the mixtures, the CANTIS model overestimated systematically the CO_2 evolved from both mixture treatments (Fig. 3a and g). Considering mineral N content, the CANTIS model underestimated systematically the NO_3^--N measured in both mixture treatments (Fig. 3b and h). However, the prediction of NO_3^--N for the PS-S treatment was better than that for the DS-S treatment with model efficiency index EF values of 0.80 and -2.77, respectively (Table 7). The high mineral N content of pig slurry resulted in an immediate N availability in the PS-S treatment from the nitrification process, which led to the better prediction of soil mineral N kinetics in the PS-S treatment than in the DS-S treatment (Fig. 3b and h). These results indicated a limitation of C mineralization that occurred when mixing organic C and N sources that the CANTIS model did not consider using the parameters calibrated from the single organic source inputs.

3.6. Contact Factor Optimization in the CANTIS Model

In the CANTIS model, the K_{MZ} factor was used by Garnier et al. (2008) to simulate a slow microbial decomposition of a C source associated with low soil-residue contact. The factor is an empirical function proposed initially by Hadas et al. (1998) to account for modifications that affect particularly microbial colonization of the C source and N availability. In this study, the discrepancy between measured and predicted values for C and for N of the mixtures was corrected by increasing the value of the contact factor K_{MZ}. Considerable improvement in prediction was obtained by optimizing K_{MZ} to 60 and 130 for PS-S and DS-S, respectively (Fig. 3d, j, e, k, f, l). The K_{MZ} value of 130 was found for the DS-S treatment where 1-2 mm organic fertilizer granules were added. It was an intermediate value compared to those provided by Garnier et al. (2008), who found 116 and 287 for 1 and 5 mm sizes of organic matter, respectively. When K_{MZ} was increased, C degradation decreased, N immobilization was lower and consequently net mineral N was higher. For PS-S_{corr} and DS-S_{corr}, respectively (Table 7), model efficiency for CO_2-C was higher than that for PS-S and DS-S, increasing from 0.72 to 0.97 for PS-S and from 0.57 to 0.98 for DS-S. The increase in K_{MZ} values led also to an improvement of the prediction of NO_3^--N kinetics (Fig. 3e and k), compared with the simulations when K_{MZ} was equal to 0 (Fig. 3b and h). However, the improvement was higher for DS-S in which mineral N was less accessible for degradation than for PS-S, which contained a higher initial amount of NH_4^+.

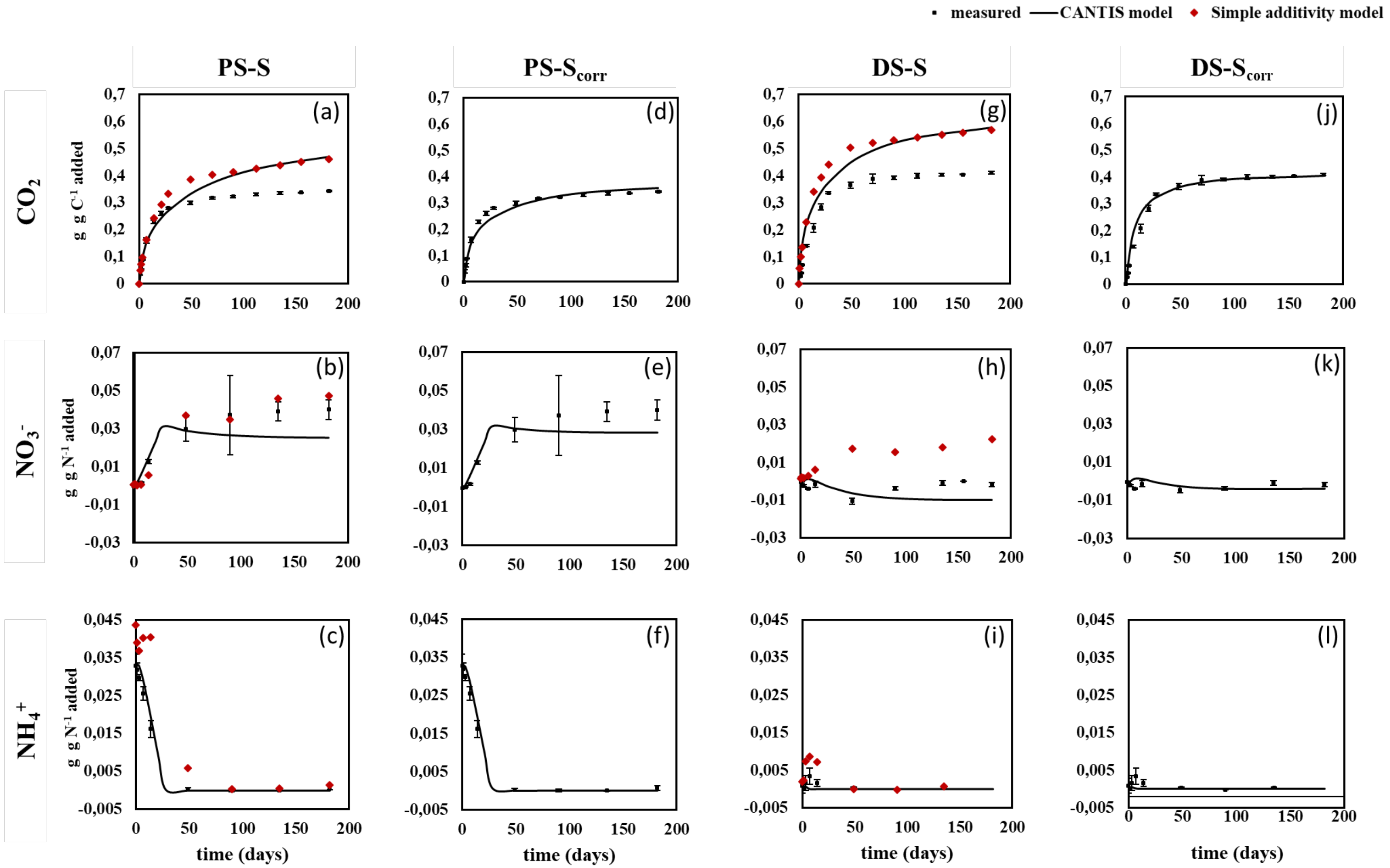

Figure 3 : C and N net mineralization kinetics during the incubation of mixtures of organic materials in control soil with pig slurry and sugarcane straw (PS-S) (a, b, c, d, e, f) and with sewage sludge and straw (DS-S) (g, h, i, j, k, l). The measurements are the black circles. The predictions with the simple additive model are the red diamonds. The predictions with CANTIS are the black lines. PS-S$_{corr}$, pig slurry and sugarcane straw (d, e, f), and DS-S$_{corr}$, sewage sludge and straw (j, k, i), display the simulation with CANTIS using a modified contact factor K$_{MZ}$

3.7. Sensitivity Analysis

A sensitivity analysis was performed on the input parameters proposed for the calibration of CANTIS. The effects on the mineralized carbon and nitrogen are displayed in Fig. 4. The model was much more sensitive to NO_3^- concentration for DS-S than for PS-S because NO_3^- was much less in the DS-S treatment. The most sensitive parameters were the humification of zymogenous biomass (H_Z). The parameters K_{MZ}, K_Z and k_1 were equally sensitive, as found in Zhang et al. (2012). Only the nitrate concentration of PS-S was sensitive to the maximum nitrification rate V_{max} because it is a parameter that only concerns nitrogen and there is a lot more nitrogen in the PS-S treatment.

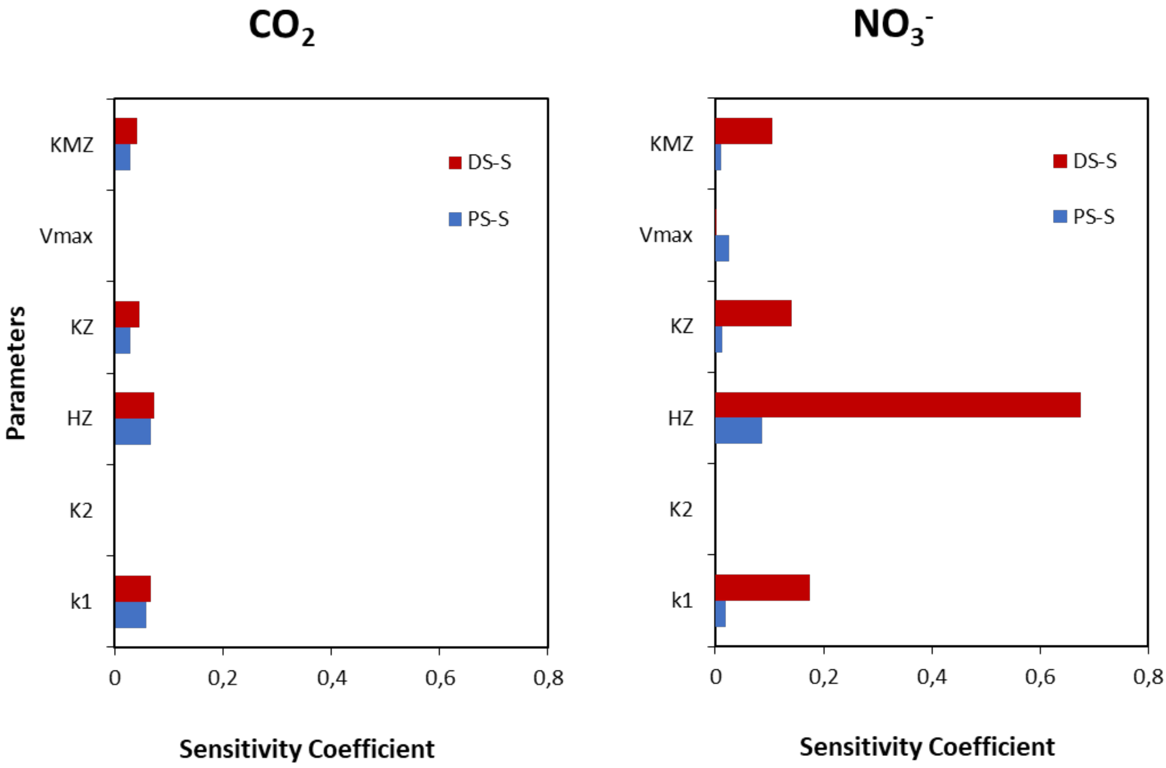

Figure 4 : Analysis of model sensitivity to the CO_2 and NO_3^- for six model parameters (K_{MZ}, V_{max}, K_Z, H_Z, k_1 and k_2)

4. Discussion

The differences between observed and expected values calculated with the additive model were negative for C mineralization, indicating an antagonist effect of mixing organic fertilizer and crop residue compared to what could have been inferred from the incubations of the products alone. This suggests that the decomposition of sugarcane straw incubated in the presence of pig slurry or sewage sludge has been reduced compared to the same straw incubated in the presence of non-limiting mineral N. The limitations for C decomposition might be the result of many factors. The fact that the mixture could have induced a negative PE compared to the PEs obtained from the weighted sum of the PEs of each organic matter incubated alone cannot be excluded. However, the contradictory results in the literature (e.g., Kuzyakov et al., 2000) do not help our understanding of this. We assumed that the PE induced by the incubation of mixtures was not significantly different from the weighted contribution of PE induced by the two organic matters incubated alone.

The limitations for C decomposition may be also due to poor contact between the organic matter and soil and/or N availability (Iqbal et al., 2014). In our study, sugarcane residue had a similar particle size in all treatments, but the source of the N changed. We conclude that the reduced accessibility to N might be due to the N speciation of the organic fertilizer. The initial distribution of N among mineral and organic compartments in organic fertilizer may have affected the N delivery over time towards the straw decomposition location.

The best fitting of the CANTIS model was obtained with contact factor K_{MZ} values of 0, 60 and 130 for the straw with mineral N in solution, with pig slurry and with solid sewage sludge, respectively. We hypothesize that the accessibility of N to decomposers might be increasingly limited in this order, due to the limitation of N diffusion. The addition of mineral N in solution (S) was the situation where N was the most available. The pig slurry treatment (PS-S) had high ammonium concentration that rapidly produced a high amount of available nitrate after nitrification. Several authors, such as Hiss and Cussler (1973), showed that the diffusion coefficients are inversely proportional to the viscosity of the liquid. The diffusion of nitrate in a viscous medium such as slurry is lower than that in a liquid medium (Olesen, Moldrup, & Henriksen, 1997), leading to reduced accessibility of N in the PS-S treatment. Most of the N was initially in organic form in the DS treatment, and before being taken up by microorganisms the organic N must be mineralized first. The N released by DS, which was quite slow in this experiment, may not necessarily coincide with the needs of N during the decomposition of the straw, which was fast in the beginning. Even if the CANTIS model was able to take into account the N mineralization/immobilization from the chemically different fractions of DS and S treatments, the model still overestimated (like the additive model) the C mineralization of the DS-S treatment. Therefore, the limitation of the decomposition of the DS-S mixture was not due to a lack of synchronization between N mineralization from DS and requirements of microbial N immobilization from S decomposition.

Some authors showed that the heterogeneity of distribution of organic matter in soil can lead to a decrease in the decomposition rate compared to a more homogeneous distribution (Angers & Recous, 1997; Magid, De Neergaard, & Brandt, 2006). The solid particles of DS from which the N originated were not necessarily located right next to straw particles, and therefore the N source was at a greater distance from the C source than when it was N in solution around the straw particles. Therefore, addition of mineral N in solution led to a more homogenous distribution of N in soil compared to solid particles such as DS, which for the latter, led to a more heterogeneous distribution of DS-derived N mineralization sites (Zhu, Christel, Bruun, & Jensen, 2014). This heterogeneity induced an N-availability limitation at the level of the active microsites of decomposition of sugarcane straw. Babey et al. (2017) showed for example that the characteristic time of diffusion was 6 days for a distance of 6 mm in loamy soils at the same water potential of 31.6 KPa as that of our study.

To take into account N limitation in models, two approaches are possible: spatial modelling that explicitly represents the heterogeneous location, or a homogeneous approach using “macroscopic” parameters that implicitly considers heterogeneity. The first approach has been used by Magid et al. (2006) using a 1-D model, to simulate C mineralization in incubation experiments where crop residues were placed in a band between two soil layers and by considering N diffusion limitation. In their work, they used a nitrate diffusion of 0.1 mm day^{-1}. Using a 3-D modelling approach, Garnier et al. (2008) found also that N diffusion limitation was the main explanation for the lower C mineralization observed in heterogeneous versus homogeneous distribution of crop residue patches.

The second approach has been used by Manzoni, Porporato, and Schimel (2008), who proposed a compartment model using two effective parameters function of microscale features. They simulated incubation experiments where patches of different crop residues were distributed into soil (Luxhøi, Bruun, Stenberg, Breland, & Jensen, 2006). The parameters were obtained from incubation experiments using ^{15}N dilution techniques in order to quantify gross mineralization-immobilization fluxes. Garnier et al. (2008) and Iqbal et al. (2014) simulated also the heterogeneous localization of crop residues and organic and inorganic soil N with the compartmental model CANTIS, but using a limitation parameter calculated from the geometry of the system (i.e. the contact surface between the residues and the soil). In the present study, we still used this limitation parameter, but it reflects the quality of the matrix containing N (viscous liquid as in slurry; solid as in DS) and not the contact surface of “rich” C residues with soil, as previously.

5. Conclusions

Replacing chemical N fertilizers with organic fertilizers to recycle wastes has an important effect on C and N cycles in soils. The results of this study showed antagonistic effects of mixing organic fertilizer and sugarcane residues compared with the expected effect (additive) from each source applied alone. These results might be the consequences of many factors, such as a negative priming effect or N diffusion limitation. We showed that the simulations of the experiments were improved by considering a contact factor between the N-rich fertilizer and the C-rich crop residues, suggesting that the limitation of decomposition could be due to poor contact and a low N accessibility for the C-rich crop residues. Therefore, applying organic fertilizers on sugarcane straw can reduce straw C mineralization, and in the longer term can lead to an increase in straw C accumulation. However, a study with a longer time scale would be needed to reach this conclusion. The observed reduction in microbial N immobilization associated with decreased C mineralization could lead to antagonistic outcomes, such as increasing N leaching or gaseous N emissions, but also increased plant N uptake because N is less immobilized by soil microbial biomass. In future works, it seems important to better characterize and quantify the macroscopic contact factor used by taking into account the chemical and physical properties of organic fertilizers and of the straw substrate. The N accessibility should be a priority that is addressed in models to drive the mineralization rate and to improve the accuracy of model tools for agroecosystem management.

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CHAPITRE 2

III. Application de fertilisants N sur le paillis de canne à sucre : conséquences sur la dynamique de décomposition du paillis et sur les flux de CO_2 et de N_2O

Résumé

L’apport de fertilisants organiques sur des paillis de canne à sucre est un potentiel substitut à la fertilisation minérale, qui peut être compétitif économiquement et en termes d'impacts environnementaux. Cependant, il n’existe pas un consensus général concernant les effets de la combinaison du paillage et de la fertilisation organique sur les émissions de gaz à effet de serre (GES) à court terme. Nous avons donc étudié différents mélanges sur le terrain en combinant deux quantités de paillis de canne à sucre (5 et 10 Mg ha^{-1}) avec différents fertilisants azotés (urée, lisier de porc et boues de station d’épuration, digérées, sèches). Nous avons mesuré les émissions de CO_2 et de N_2O de 0 à 14 jours, et la dynamique de décomposition du paillis de 0 à 120 jours après l’épandage. Nous avons émis l’hypothèse que la quantité relative de N par rapport au C modifie la dynamique de décomposition et les flux de GES. Le N_2O-N et le CO_2-C émis ont été mesurés à l'aide de cloches statiques de mesure de gaz. La décomposition du paillis-C a été mesurée à l'aide de sacs à litière (*litterbags*). Nos résultats ont montré que la proportion de paillis-C restant dans le sol au jour 120 n'a pas été modifiée ni par le type de fertilisant azoté, ni par la quantité de paillis. Sur une période plus courte (0 à 49 jours), les différents traitements ont affecté les pertes de mulch-C et de mulch-N, et les rapports C:N, indiquant une interaction transitoire entre la dynamique du paillis et celle du N apporté. Les émissions de N_2O-N ont été classées comme suit : lisier de porc > urée > boues de station d’épuration, soulignant l’effet de la forme physique des fertilisants appliqués. Ce travail marque la nécessité d'étudier conjointement les dynamiques du carbone et de l'azote et de prendre en compte le carbone du sol et les gaz émis du sol dans une approche de bilan de GES, selon les pratiques de culture de la canne à sucre.

Application of N Fertilizer to Sugarcane Mulches: Consequences for the Dynamics of Mulch Decomposition and CO_2 and N_2O Fluxes

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Abstract

Using organic fertilizers on sugarcane mulches is a potentially interesting substitute for mineral fertilization in terms of economic and environmental impacts. However, no general agreement exists regarding the short-term effect of combining mulching and organic fertilization on greenhouse gas (GHG) emissions. Therefore, we studied different mixtures in the field by combining two amounts of sugarcane mulch (5 and 10 Mg ha^{-1}) with different N fertilizers (urea, pig slurry, and digested sewage sludge). We measured CO_2 and N_2O emissions shortly after application of the mixtures (0–14 days) and the mulch decomposition dynamics from 0 to 120 days after application. We hypothesized that the relative amount of N to C modifies the decomposition dynamics and GHG fluxes. The emitted N_2O-N and CO_2-C were measured using static chambers. Mulch-C decomposition was measured using litterbags. Our results showed that the proportion of mulch-C remaining on the soil on day 120 was not altered by either the type of N fertilizer or the mulch amount. On a shorter timescale (0–49 days), the different N treatments affected the mulch-C and mulch-N losses and the C:N ratios, indicating a transient interaction between the dynamics of the mulch and the added N. The intensity of N_2O-N emission was ranked as pig slurry > urea > digested sewage sludge, underscoring the effect of the physical form of N fertilizer. This work highlights the need to jointly study carbon and nitrogen dynamics and consider both soil carbon and gas emissions to assess the GHG balances of sugarcane farming practices.

KEYWORDS

Sugarcane trash, Organic waste, Pig slurry, Sewage sludge, GHG emissions, Nitrogen

List of figures

1. Introduction

In sugarcane cropping systems, mulch represents 10 to 30 t dry matter (DM) ha^{-1} of the aboveground biomass depending on the sugarcane variety, pedoclimatic conditions, ratoon number, and harvesting technique (Hassuani et al., 2005; Franco et al., 2013; Leal et al., 2013). There is growing interest in utilizing plant biomass as a resource to meet energy and raw material needs (Cantarella et al., 2013; Leal et al., 2013; Sordi and Manechini, 2013) because of its potential to be used as a green C substitute for fossil resources (Latif et al., 2014; Kircher, 2015; Gaurav et al., 2017). However, the recycling of crop residues in soils is also crucial due to the important services provided to the agrosystem, e.g., the buffer effect of mulches on soil temperature and moisture, weed and erosion control, and C and nutrient recycling and subsequent storage (Lal, 1974; Cerri et al., 2011; Comte et al., 2012; Azevedo et al., 2014; Mthimkhulu et al., 2016). Negative effects of mulch have also been reported, mostly associated with the enhancement of greenhouse gas (GHG) emissions especially because the layer of crop residues present at the soil surface may favor denitrification processes due to high N and C concentrations at the soil-mulch interface (Cerri et al., 2009; Carmo et al., 2013; Fracetto et al., 2017). However, there is no general agreement on the impact that mulching has on GHG emissions (Chen et al., 2017; Malhi and Lemke, 2007), and few studies have investigated this question in sugarcane conditions. It is important to assess this question in the growing context of alternative uses and tradeoffs for the use of plant biomasses (Fowles, 2007; Duncan et al., 2016; Song et al., 2016).

The combination of recycled mulches and organic wastes brought to the sugarcane crop via organic fertilization is a developing practice (Gilbert et al., 2008; Carmo et al., 2013; Franco et al., 2013). This practice is particularly common in areas with high livestock production due to the necessity to enlarge manure spreading areas. Adding organic wastes to sugarcane mulch may also combine the interests of substituting mineral fertilization with organic fertilization and increasing soil organic matter, i.e., the availability of soil nutrients and C sequestration. The joint management of manure and plant biomass as straw on agricultural soils may, therefore, be a viable option for reducing the environmental impacts associated with livestock production and recycling nutrients efficiently. Aita et al. (2012) investigated the effects of joint wheat straw and pig slurry applications in a laboratory incubation. The results showed that the C mineralization of the straw applied to the soil surface was higher when pig slurry and straw were added simultaneously than when straw was applied alone, suggesting that the synergistic effects of the concomitant application of straw and pig slurry resulted from the suppression of the N limitation of decomposers by pig slurry-N application. Giacomini et al. (2015) did not observe any effect on C straw mineralization when oat straw and sewage sludge were incorporated together into a soil, suggesting that the decomposers were not N-limited, but the N from sewage sludge was more immobilized in the soil than with sewage sludge applied alone due to the high microbial demand during straw decomposition. These results indicate a close interaction between the C and N sources that is linked to the heterotrophic microbial activities during decomposition (in the soil and/or in the mulch), and the outputs of these interactions depend on the relative quantities of C and N, the chemical forms, and the respective location of these sources in or at the surface of the soils. Still, little is known about how mixtures of mulch residues and N fertilizers influence the C and N fluxes in the field. To the best of our knowledge, no study has addressed this question in sugarcane cropping systems.

Therefore, the objective of this work was to assess the effects of joint inputs of sugarcane trash (i.e., the sugarcane tops and leaves) left as mulch at the soil surface and organic N applied as fertilizers in a sugarcane cropping system on the dynamics of mulch-C decomposition and the emissions of N_2O just after N fertilizer application. We compared two organic wastes (pig slurry and solid sewage sludge) to synthetic fertilizer (urea) as a control treatment and combined them with two initial quantities of sugarcane trash left as mulch on the soil, mimicking actual farmers' practices. We hypothesized that the C:N ratios of the mixtures that were generated by these treatments significantly modified the decomposition dynamics of the sugarcane trash and the N_2O fluxes.

2. Materials and Methods

2.1. Site

The experiment was set up at the SOERE PRO[10] (i.e., long-term observation and experimentation systems for environmental research on organic waste products) experimental station at La Mare (Sainte-Marie), Reunion Island (20° 54′ 12.2″ S 55° 31′ 46.6″ E). This zone is characterized by a tropical climate with an average annual temperature of 24.6°C. The lowest

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[10] https://ur-recyclage-risque.cirad.fr/en/main-projects/soere-pro-la-reunion

mean monthly temperature is 21.9°C in August, and the highest temperature 27.2°C in February. The annual average rainfall (over the 2006-2016 period) was 1600 mm, 80% of which falls between December and June. The soil is a Nitisol according to the WRB-FAO soil classification (WRB-FAO, 2014), and the top 0-20 cm soil layer contains high clay and fine silt (40 and 34 g 100 g^{-1} soil, respectively), 11 g coarse silt 100 g^{-1} soil and 15 g sand 100 g^{-1} soil, with Ca 140, Mg 37.5, K 24.8, P 100 mg kg^{-1} soil, CEC 109 mmol kg^{-1} soil, N 1.82, and soil organic carbon (SOC) 20.6 g kg^{-1} soil.

The sugarcane trial was planted in March 2014 from viable buds of the R579 sugarcane variety and placed in 1.5 m row spacing with 6 rows per plot. The experiment was conducted during the first ratoon of sugarcane (from November 2014 to March 2015) on 9 plots (10.5 m × 28 m) over 3 separated blocks (replicates). Each block that was cultivated with sugarcane included 3 sub-plots that received the 3 fertilizer types: pig slurry, digested solid sewage sludge, and urea.

2.2. Experimental Design

In October 2014, 8 months after viable buds were planted, the sugarcane biomass was measured and sugarcane residues, consisting of leaves (blades and sheaths) and tops, were collected immediately after sugarcane harvesting by randomized spatial sampling over the 3 blocks (n = 27). The sugarcane residue samples were oven dried at 60 °C until reaching a stable weight (72 h).

The treatments studied were (i) two initial quantities of sugarcane trash: 10 t DM ha^{-1} (corresponding to the amount of sugarcane trash measured at harvest) and half this amount, i.e., 5 t DM ha^{-1}; and (ii) three forms of N fertilizer applied to the sugarcane trash: pig slurry, solid sewage sludge, and urea as a control for synthetic fertilizer. For an expected yield of 140 Mg ha^{-1} of sugarcane stalks, the local optimum recommendation for fertilization was 170/60/230 kg ha^{-1} of N/P/K. To comply with national regulations on the amounts of N and P input with fertilizers, to limit nitrates and phosphates in waters, and because the organic fertilizers had different N, P, and K contents and different mineral fertilizer equivalency values, the N, P, or K fertilizer application rates varied. Pig slurry, solid sewage sludge, and urea were thus applied at 139, 119, and 170 kg N ha^{-1}, respectively. Following the practices of actual cane growers, N fertilizers were applied over the mulch in their specific physical forms: liquid for pig slurry, digested and limed pellets for solid sewage sludge, and solid granules for urea.

Based on these treatments, five mixtures were studied: 5 t DM ha^{-1} mulch + urea (U0.5), 5 t DM ha^{-1} mulch + pig slurry (PS0.5), 10 t DM ha^{-1} mulch + urea (U), 10 t DM ha^{-1} mulch + pig slurry (PS), and 10 t DM ha^{-1} mulch + digested solid sewage sludge (DS). Urea is the N fertilizer reference used as a control in this study. Regarding the two trash treatments, even if some sugarcane mulch can be exported from the field for energy purposes (Franco et al., 2013), cattle litter or cattle feed (Hassoun et al., 2002; Andrade et al., 2003), local cane growers always keep some mulch on their fields to prevent weed development, limit erosion, and promote nutrient recycling. Therefore, depending on the sugarcane yield, the sugarcane trash left in the field is usually 10-12 t DM ha^{-1}, while the minimum quantity of sugarcane trash that remains on the field is 5 t DM ha^{-1} depending on the use options available to the cane growers. Among others, urea is the most common form of N fertilizer utilized by local cane growers. Therefore, in this study, U0.5 and U were both considered "control" treatments representing cane growers' actual practices. The physicochemical characteristics of the organic materials and their humus potential depicting the biodegradability of the organic matter (indicator of potential residual carbon, IROC), according to Lashermes et al. (2009) are shown in Table 1.

Table 1 : *Physical and chemical characteristics of the organic materials (digested sewage sludge, pig slurry) applied to sugarcane mulch*

Organic material	DM	TOC	Total N	N-NH$_4^+$	IROC	C:N	pH$_{H2O}$
Sewage Sludge	907	299	43.4	4.1	80.6	6.9	12.0
Pig Slurry	8.8	1.8	1.8	1.4	47.4	1.0	8.0
Mulch	928	453.8	6.7	*n.d.*	83.7	67.8	*n.d.*

DM, dry matter; TOC, total organic carbon; total N, N-NH$_4^+$ are expressed in g kg^{-1} fresh weight; IROC, indicator of residual organic carbon is expressed in g 100 g^{-1} of organic matter according to Lashermes et al. (2009); n.d., not measured

The final compositions (after mixing) of the treatments are given in Table 2.

Table 2 ***:****Initial total C and N inputs of the different treatments and global C:N ratio of the mixtures (PS 0.5 = pig slurry +5 t ha^{-1} sugarcane mulch, PS = pig slurry +10 t ha^{-1} sugarcane mulch, U0.5 = urea +5 t ha^{-1} sugarcane mulch, U = urea +10 t ha^{-1} sugarcane mulch, DS = digested sewage sludge +10 t ha^{-1} sugarcane mulch)*

Treatment		**Total C**	**Total N**	**Global C:N**
		(kg ha^{-1})		-
PS0.5	pig slurry mulch	141 2269	139 33.5	14
U0.5	urea mulch	73 2269	170 33.5	21
PS	pig slurry mulch	141 4538	139 67	23
U	urea mulch	73.1 4538	170 67	20
DS	sewage sludge mulch	816 4538	119 67	29

Rainfall, temperature, and atmospheric pressure were measured continuously at the nearest meteorological station (20° 88′ S 55° 52′ E).

2.3. Mulch-C Decomposition Using Litterbags

The mulch-C dynamics during decomposition were assessed using a litterbag approach. Litterbags (30 cm × 30 cm × 15 cm) of white polyvinylchloride garden grid with a mesh of 1 cm were prepared. The soil areas beneath the litterbags were free of any visible particulate organic matter before litterbag deposition. The mixtures were prepared directly in the field and placed into opened litterbags. The required amounts of sugarcane residues and N fertilizer were individually weighed for each litterbag, with 3 replicates per treatment and per sampling date. Pig slurry is known to have solid parts that rapidly deposit. Therefore, particular care was taken to obtain a homogeneous suspension (a helical paint mixer was used in the PS stock container) to be weighed rapidly and thus ensure that the PS on litterbags was distributed in equal quantities and in the same homogeneous state. The litterbags were then sealed by stapling the open side with metallic clips. The litterbags were placed in the central row space of each plot, with 1.5 m of spacing between each bag (Fig. 1). The litterbags were fixed at the soil surface using U-shaped metallic clamps. The experiment was launched at intervals of 7 days for each of the 3 types of N fertilizer to avoid potential cross-contamination of gas measurements by emitted NH3. The litterbags were sampled 6 times on days 14, 28, 49, 70, 91, and 120 after on-field deposition. For each fertilizer treatment and on each block, twelve litterbags corresponding to the 6 sampling dates and the two mulch quantities and two cylindrical gas chambers corresponding to the two mulch quantities were settled (Fig. 1). Only one mulch quantity was tested for the sewage sludge fertilizer (6 litterbags and 1 chamber per plot). For 5 treatments and 3 replicates, a total of 90 litterbags were settled.

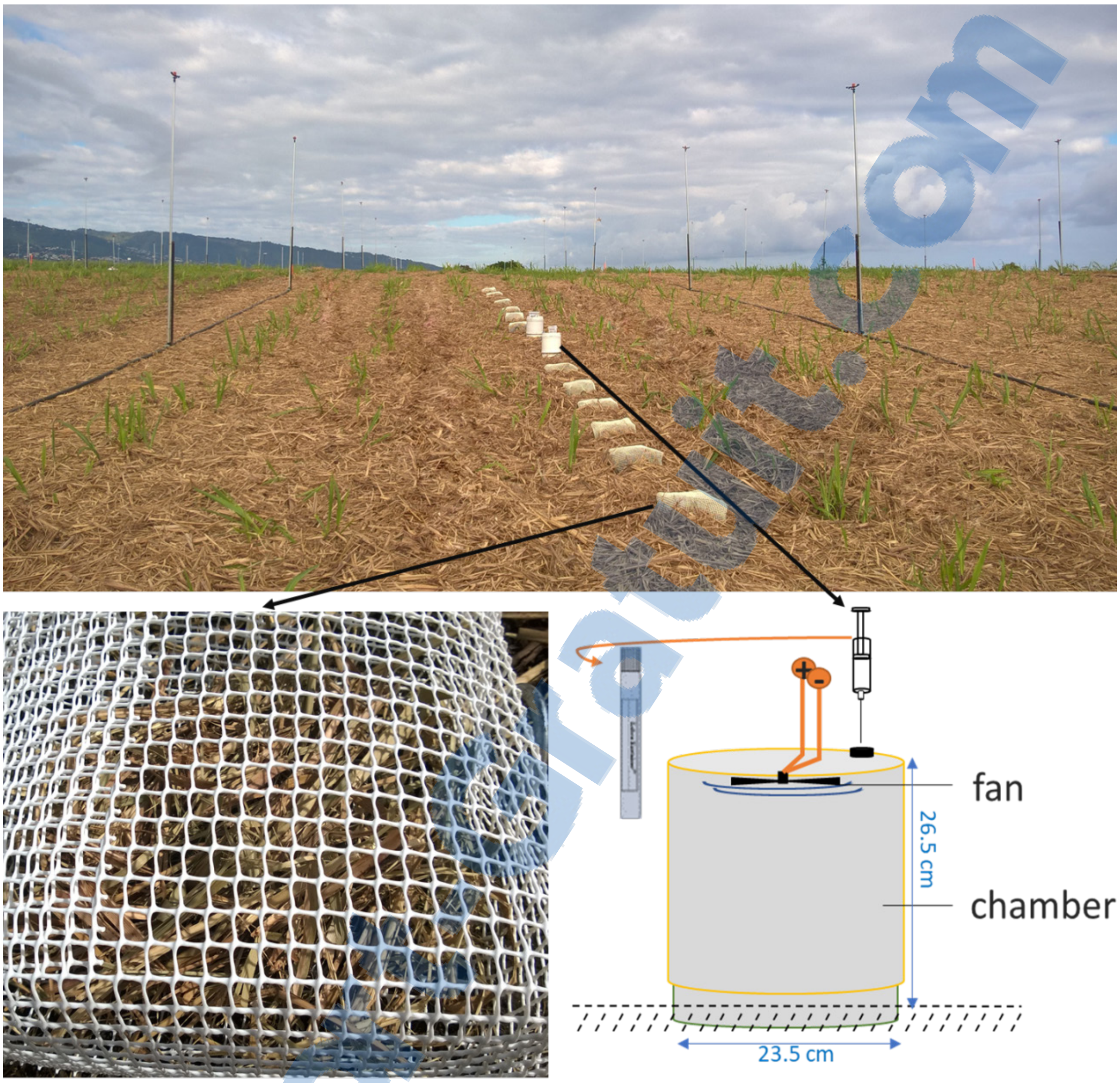

Figure 1: Design of litterbag and gas chamber experiments, measuring the decomposition and the emissions of CO_2 and N_2O of mulch-fertilizers mixtures on field. View of a plot with two quantities of sugarcane mulch.

2.4. N_2O and CO_2 Measurements Using Cylindrical Chambers

N_2O emissions were measured at 10 times shortly after mulch and N additions, from 5 days before the start of the experiment until 14 days after mulch and N application. CO_2 emissions were also measured with the same sampling procedure and schedule as these emissions represent a sensitive measure of decomposition activity, although this was not a main objective of the work. The gas measurements were collected by the static-opaque chamber

method approach (Hutchinson and Livingston, 1993). A cylindrical socket (d = 23.5 cm) was driven into the soil in the center of each plot (i.e., in the sugarcane row space) 2 weeks before the first sampling, with three replicates per treatment. At each sampling time, a cylindrical lid was fit onto the socket, forming a closed chamber of isolated atmosphere. The lids had a small fan that was activated at least 30 s before sampling. The homogenized atmosphere was sampled at chamber closure and then 30 and 60 minutes after. The sampling consisted of inserting a blood collection needle (TERUMO Neolus) through a septum on the lid and collecting the gas by filling a 20 ml syringe tank (BD Discardit II). The entire tank content was then introduced into a 12-ml pre-evacuated vacuum vial (Labco Exetainer ®). The vacuum was previously formed with a self-driven electric vacuum pump (SDEC, PAV2000). Vials were tested for vacuum when inserting the needle of the syringe tank, whose piston should naturally move down if a vacuum was present. In the laboratory, the vials containing gas samples were sealed with candle wax to prevent leaks and then transported for analysis within one week.

2.5. Analytical Methods

After sampling, the remaining mulch was gently washed with water to remove soil contamination and definitively determine the evolution of mulch DM and C, and oven-dried (60°C, 72 h) until a stable weight was reached. Sugarcane residues were ground to 1 mm, and total N and C were measured by dry combustion elemental analysis (Dumas method) as described in NF ISO 13878 (ISO 1998) and NF ISO 10694 (ISO 1995), respectively. N_2O concentrations were determined by gas chromatography with an electron capture detector (GC-ECD Varian 3800). CO_2 concentrations were determined by gas chromatography with a thermal conductivity detector (MTI M200 Micro Gas Chromatography). Four standard gases (Linde Gas) were used for calibration: $[CO_2]/[N_2O]$ = 0 ppm/0.5 ppm; 286 ppm/1 ppm; 1025 ppm/10 ppm; and 11,659 ppm/125 ppm.

2.6. Calculations and Statistical Analysis

The mulch decomposition rates were calculated on each date as the proportions of the initial dry matter, initial carbon, and initial nitrogen of the sugarcane residues remaining in the litterbags. The half-life time of the mulch, which corresponds to the time necessary to degrade half of the initial quantity of added mulch, was calculated as in Dietrich et al. (2017). Total water input was calculated (in mm) as the sum of rainfall and irrigation. The environmental variables used for gas concentration calculations were temporally matched with gas sampling periods.

The linear increase in gas partial volumes was verified between chamber closure and the last sampling (corresponding to 3 gas samplings at 0, 30, and 60 min after closure and deployment of each soil chamber) by calculating the coefficient of determination for each of the 165 samples. Samples with $R^2 < 0.7$ were considered outliers, and their values were not used to calculate the fluxes. The gas emission fluxes were calculated using the relation:

$$f_i = \frac{\Delta G}{\Delta t} \times \frac{V}{V_m} \times \frac{M}{A} \qquad \text{Eq. 1}$$

where f_i is the emission flow (µg m^{-2} h^{-1}), ΔG is the increase of the CO_2 or N_2O partial volumes (µL L^{-1}) within the chamber during the enclosure period Δt (h), V is the volume of the chamber (L), V_m is the molar volume of the gas at the sampling temperature (L mol^{-1}), M is the molar mass of C contained in CO_2 or the molar mass of N contained in N_2O, and A is the area covered by the chamber socket (m^2).

The mean fluxes per treatment were calculated from the ad hoc samplings between 0 and 14 days and were expressed as the absolute mean flux (g N_2O-N or kg CO_2-C ha^{-1} h^{-1}) using the relation:

$$F = \frac{1}{s \times n} \sum_{i=1}^{s\times n} f_i \qquad \text{Eq. 2}$$

where F is the mean N_2O-N or CO_2-C flux per treatment, f_i is the measured flux of sample i, s is the number of sampling occasions (from day 0 up to day 14, s = 8), and n is the number of replicates (n = 3). The normalized N_2O-N or CO_2-C fluxes (kg kg^{-1} ha^{-1}) were calculated by dividing the respective mean fluxes by the total N or C inputs from mulch and fertilizer for each treatment.

The normality of the distribution and homoscedasticity were verified using Shapiro-Wilk's normality and Bartlett's tests, respectively. When those conditions were met, a oneway ANOVA followed by a Tukey pairwise comparison test was applied. Otherwise, a non-parametric test (Kruskal-Wallis) was used to detect differences, followed by Wilcoxon's test adjusted by the Benjamini, Hochberg, and Yekutieli method. All statistical analyses were performed with R software (v3.2.2, R Core Team, 2015, R Foundation for Statistical Computing, Vienna, Austria), powered by RStudio (v0.99.484 – 2009-2015, RStudio Team, 2015, Integrated Development for R. RStudio Inc., Boston, MA).

3. Results

3.1. Climatic Conditions

The average, minimal, and maximal temperatures during the 120-day experiment were 26.6°C, 22.0°C, and 30.1°C, respectively. Over the same period, the plots received 1664 mm of water, including 83% from rain and 17% from irrigation. Soil conditions were not favorable for N_2O emissions by denitrification (Davidson et al., 2000) during the GHG measurement period (-5 days to +14 days after fertilization) as the water-filled pore space varied from 30 to 60%.

3.2. Mulch-C and N Dynamics

The mulch-C loss was found to be proportional to the mulch-DM loss during the experiment for all treatments ($R^2 > 0.99$, data not shown); consequently, only mulch-C data will be presented hereafter (Fig. 2). Overall, the mulch-C loss dynamics were not affected by the type of N fertilizer treatment, and on day 120, the mulch-C remaining was 40.2 ± 2.6%, 47.5 ± 1.6%, and 48.5 ± 2.2% of the initial mulch-C for the DS, PS, and U treatments, respectively, with no significant differences among the three treatments. Thus, the mean half-life time of the mulch was 100 ± 8 days. The mulch-N followed similar kinetics for the three N treatments apart from day 14 for the U treatment, which had a possible artifact due to the high variability of remaining mulch-N that was measured (Fig. 2d).

Mulch-C and mulch-N followed the same pattern for the U and PS treatments (Fig. 2a, b) during the first 49 days (62.8 ± 1.8% remaining C and 61.8 ± 3.2% remaining N). For the DS treatment, mulch-C disappeared faster than mulch-N during the same period. From day 49, while the mulch-C loss continued, the mulch-N stabilized at an average of 64.3 ± 4.8% of the initial mulch-N, with no significant differences among the treatments. The mulch C:N ratio remained relatively constant for all treatments until day 49 as a consequence of the synchronized mulch-C and mulch-N loss. Then, the ratio rapidly decreased from day 49 onwards, indicating relative enrichment of the N in the remaining mulch particles compared to C (Fig. 2g). The noticeable exception was the DS treatment that exhibited a regular and faster decrease of the C:N ratio along the decomposition period (Fig. 2g).

The mulch-C loss kinetics did not significantly differ for the two quantities of mulch application with urea (U and U0.5) and pig slurry (PS and PS0.5) treatments, although the remaining mulch-C tended to be slightly higher for the lower quantities (U0.5 and PS0.5). On day 120, the remaining mulch-C amounted to 48.5 ± 2.2% (U) and 53.9 ± 2.1% (U0.5) of the initial C (Fig. 2b), and 47.5 ± 1.6% (PS) and 52.2 ± 2.3% (PS0.5) of the initial C (Fig. 2c). These results indicate that the decomposition of mulch was proportional to the initial amount of mulch regardless of the different initial C:N ratios of the mixtures. The mulch quantity also did not affect the dynamics of mulch-N for either the U or PS treatments (Fig. 2e, f). Consequently, the evolution and the final C:N ratios of the mulch were not affected by the initial quantity of mulch (Fig. 2h, i). The fact that the decomposition was proportional to the initial quantity of mulch means that after 120 days of decomposition, twice as much C and N remains in the mulch when the initial amount of mulch is doubled.

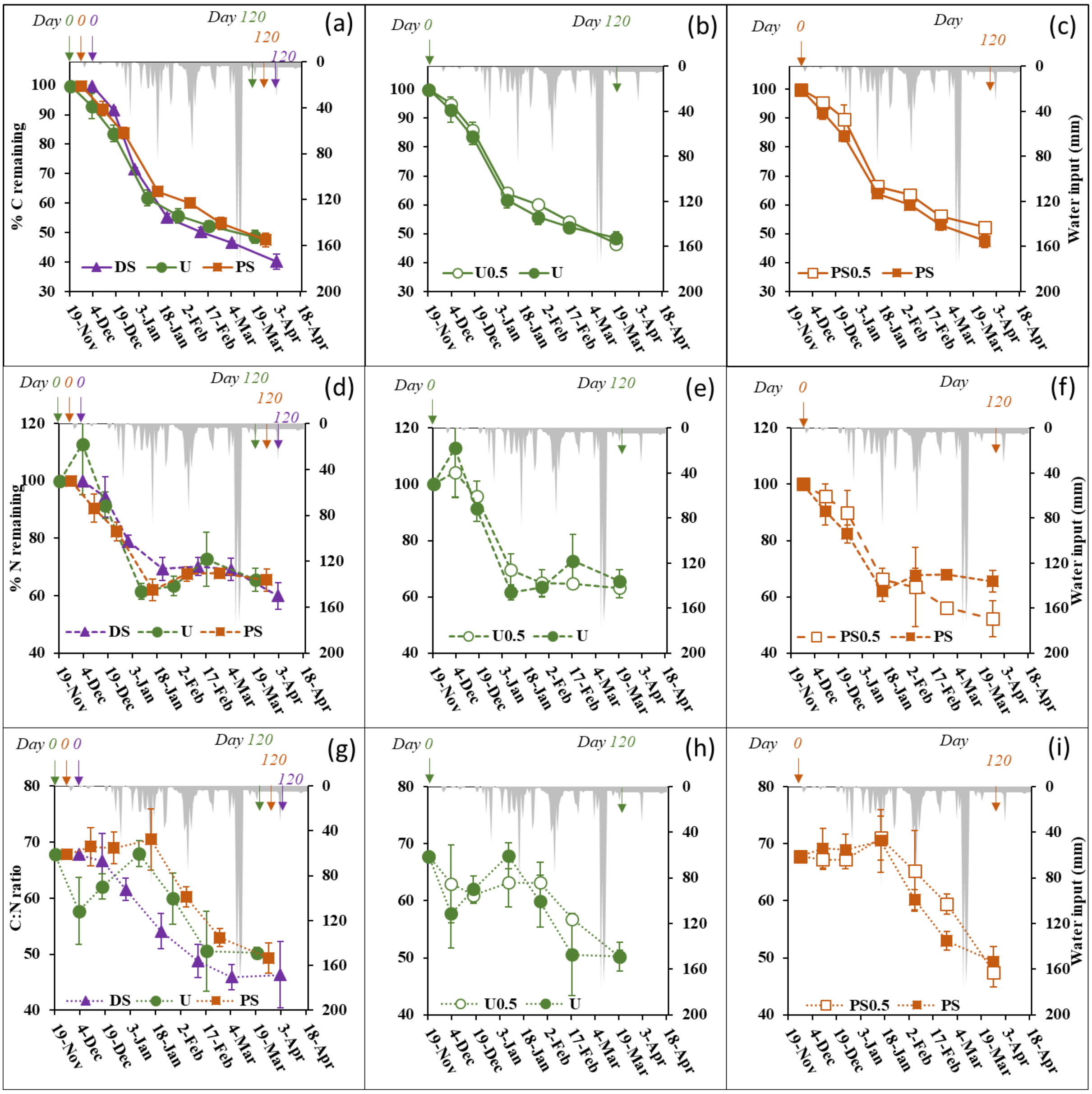

Figure 2 : Mulch-C remaining (a–c), mulch-N remaining (d–f), and C:N ratio of the sugarcane mulch (g–i) remaining the soil surface in a sugarcane stand measured on days 14, 28, 49, 70, 91, and 120 after on-field deposition for two rates of initial mulch and three N forms applied to mulch: 10 t DM ha^{-1} mulch + urea (U), 10 t DM ha^{-1} mulch + pig slurry (PS), 10 t DM ha^{-1} mulch + sewage sludge (DS), 5 t DM ha^{-1} of mulch + urea (U0.5), and 5 t DM ha^{-1} mulch + pig slurry (PS0.5). The bars on the graph indicate the standard errors of the mean (n = 3)

3.3. CO_2 and N_2O Emissions

3.4. Effect of N Fertilizer Treatments

The return to basal emission values measured at day -5 was verified for all treatments on day 49 (data not shown). Therefore, the kinetics observed mainly resulted from the fertilizer spreading events. The CO_2-C emitted ranged from 0.24 to 3.35 kg CO_2-C ha^{-1} h^{-1} for the three N treatments during the 0–14-day period. The PS treatment differed from the two other N treatments and exhibited a high peak of CO_2 (at 3.35 kg CO_2-C ha^{-1} h^{-1}), which occurred immediately after pig slurry application on day 0 (Fig. 3a). The PS treatment exhibited high N_2O-N fluxes during the first 9 days after pig slurry application (Fig. 3d). Conversely, the U treatment showed a gradual but steady increase in N_2O-N flux over the first 9 days followed by a decrease (Fig. 3e). The DS treatment showed the smallest N_2O-N flux, with only two peaks on days 1 and 2 following application and then on day 7 (Fig. 3d).

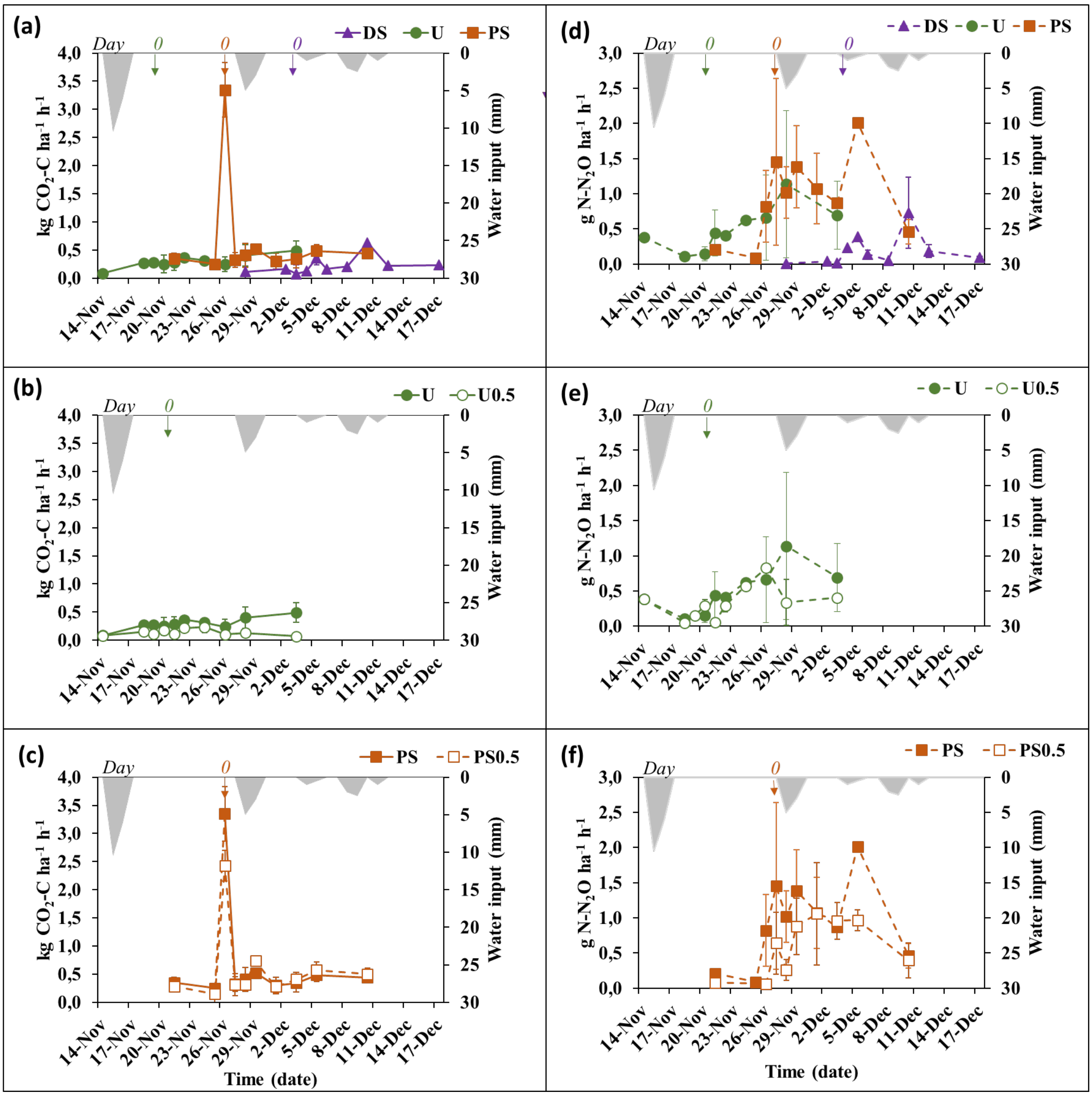

Figure 3 : Dynamics of CO_2-C fluxes (a–c) and N_2O-N fluxes (d–f) over the first 14 days following N application for 10 t DM ha^{-1} mulch + urea (U), 10 t DM ha^{-1} mulch + pig slurry (PS), 10 t DM ha^{-1} mulch + sewage sludge (DS), and 5 t DM ha^{-1} of mulch + urea (U0.5), 5 t DM ha^{-1} mulch + pig slurry (PS0.5). The bars on the graph indicate the standard errors of the mean (n = 3)

Considering that the quantities of C and N added in the form of mulch + fertilizer mixtures were different among the treatments (Table 2), these quantities should be expressed as per unit of N (or C) added to facilitate comparison and extrapolation to other agricultural situations. The CO_2-C emitted ranged from 0.02 to 1.01 g CO_2-C kg^{-1} added C h^{-1} for the three N treatments during the 0–14-day period. When the measured fluxes were averaged over the 0–14-day period (Fig. 4), the PS treatment exhibited the highest mean CO_2 emissions (at 2.3 ± 0.5 g CO_2-C kg^{-1} added C h^{-1}) compared to the DS (0.38 ± 0.1 kg CO_2-C kg^{-1} added C h^{-1}) and U (0.52 ± 0.17 g CO_2-C kg^{-1} added C h^{-1}) treatments; however, this difference was due to the CO_2 emitted during the first day after pig slurry addition.

The intensity and dynamics of the N_2O-N emissions varied among the N treatments and, when expressed relative to the amount of N fertilizer applied, were ranked as PS > U > DS. On average over the 0–14-day period, the PS treatment exhibited significantly higher mean N_2O-N emissions (at 0.0443 ± 0.0164 g N_2O-N kg^{-1} added N h^{-1}) compared to the DS and U treatments, which did not differ significantly (0.0098 ± 0.0036 g and 0.0145 ± 0.0083 g N_2O-N kg^{-1} added N h^{-1}, respectively).

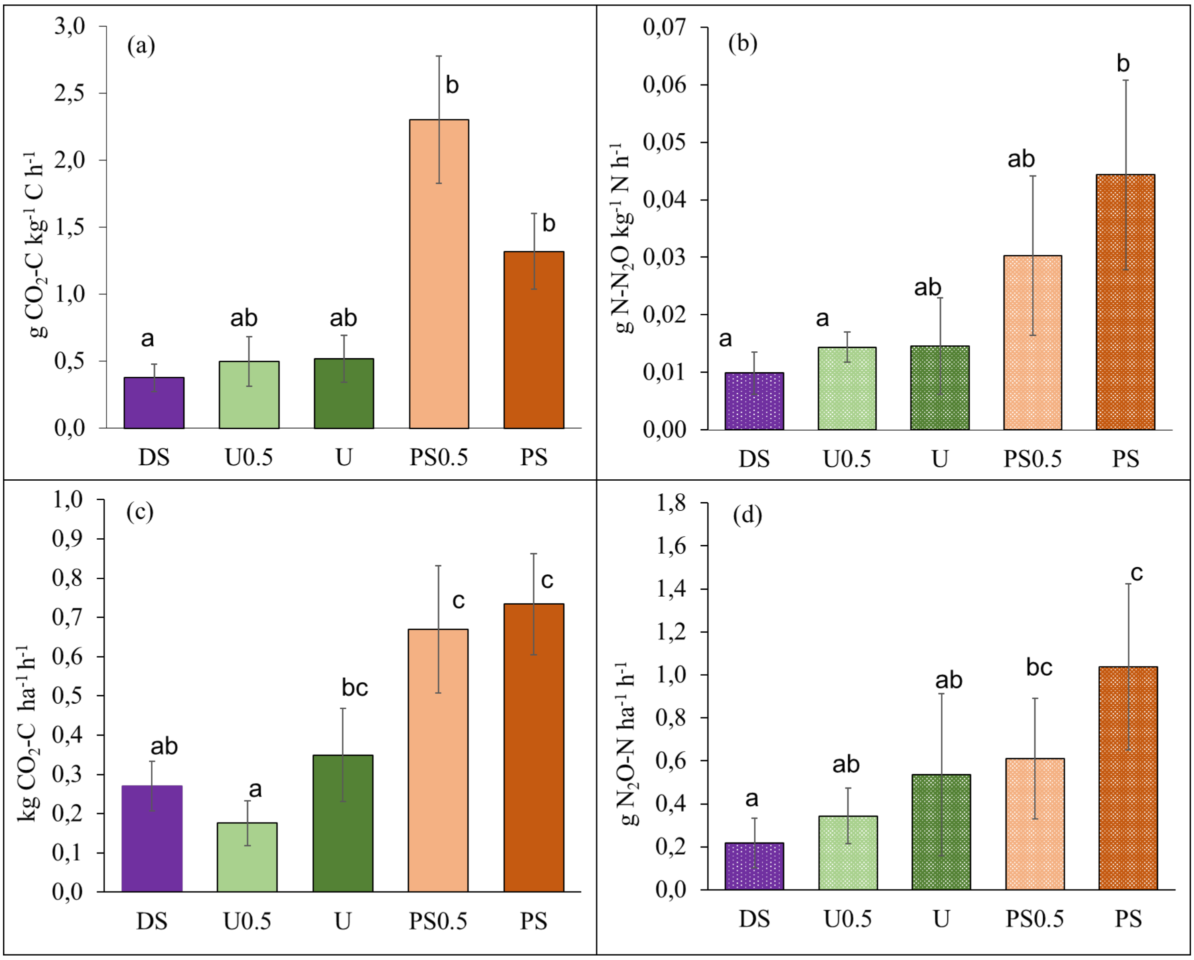

Figure 4 : Mean normalized CO_2-C (a) and N_2O-N (b) emissions and mean absolute CO_2-C (c) and N_2O-N (d) emissions over the first 14 days following N application for 10 t DM ha^{-1} mulch + urea (U), 10 t DM ha^{-1} mulch + pig slurry (PS), 10 t DM ha^{-1} mulch + sewage sludge (DS), and 5 t DM ha^{-1} of mulch + urea (U0.5), 5 t DM ha^{-1} mulch + pig slurry (PS0.5). Normalized emissions were calculated by unit of initially added C or N. The bars on the graph indicate the standard errors of the mean (n = 3). Same letters on top of the bars indicate no significant difference at $p = 0.05$

3.4. Effect of Mulch Quantity

Halving the quantity of mulch affected the CO_2-C fluxes (e.g., Fig. 3b). The peak CO_2 values measured in the PS (0.72 g CO_2-C kg^{-1} added C h^{-1}, data not shown) treatment immediately after pig slurry application were also observed for the PS0.5 (1.01 g CO_2-C kg^{-1} added C h^{-1}, data not shown) treatment. The N_2O-N emissions followed the same pattern regardless of the quantity of mulch applied (Fig. 3e, f), with a concomitant increase in N_2O-N emitted for the U0.5 and U treatments, although the difference between the two straw treatments was not significant due to the high variability among replicates.

When considering the mean absolute CO_2-C fluxes, the U treatment emitted 2-fold more (p value < 0.05) CO_2-C than the U0.5 treatment (Fig. 4a), while the PS and PS0.5 treatments emitted the same mean CO_2-C fluxes. For N_2O-N, the higher rate of mulch application tended to increase the absolute amount of N_2O-N emitted (Fig. 4b), although the differences were not significant due to the large variability in the fluxes. When expressed per unit of added C or unit of added N, the average measured emissions did not vary significantly with the initial quantity of mulch (Fig. 4c, d).

4. Discussion

4.1. Effects of the Type of Fertilizer N on Sugarcane Mulch Decomposition

Our results showed that the half-life time of mulch-C was 2-fold shorter than that in other studies (Cerri et al., 2004; Dietrich et al., 2017).We assume that this faster decomposition results from the more favorable conditions overall for decomposition in the present study, i.e., the relatively higher availability of N (low C:N ratio of the mulch and fertilizer mixture), the higher amount of water received by the plots during the four months of study (equivalent to 1 year of water input for those studies), and the higher mean temperature in our study area.

In our experimental conditions, the observed decrease in mulch-C and mulch-N after 120 days of decomposition did not differ between the three N fertilizer treatments, which indicates that the form and amount of N fertilizer added did not affect the mulch decomposition; in particular, no difference was found between the two organic fertilizers and urea. This result was not expected since the chemical and physical forms of N applied (and to a lesser extent the amount of N applied) varied greatly among the urea, solid sludge, and pig slurry treatments. This result suggests that N was not a limiting factor in the experimental conditions of this study, and other factors had greater impacts on the decomposition. Although DS exhibited a 1.7-fold higher humus potential than PS (see IROC, Table 1), the different C biodegradability between DS and PS did not affect mulch decomposition. In other surface decomposition conditions, the moisture conditions of the mulches were found to be the main controlling factor of decomposition (Coppens et al., 2006; Iqbal et al., 2015), and these conditions were related to the amount and frequency of rain (Vanlauwe et al., 2001).

However, on a shorter timescale (0–49 days), the different N treatments transiently affected the mulch decomposition kinetics, suggesting an interaction between the C dynamics and the added N. The initial C:N ratio of the mulch was 68, which is lower than the C:N ratios usually observed for sugarcane mulch (Fortes et al., 2012). This difference is the consequence of early sugarcane harvest 8 months after planting when the crop had barely reached the ripening stage, resulting in the relatively higher N content in the aboveground biomass compared to the first ratoon sugarcane that is currently harvested one year after planting (Leite et al., 2016). However, since the mulch had a C:N ratio greater than 20-25, mineral N was expected to be immobilized by decomposers to meet the microbial stoichiometric homeostasis required for efficient use of mulch-C (Tripolskaja et al., 2014). The C:N ratio stability observed for the PS and U mulches during the first 49 days indicates that the losses (either by mineralization or/and leaching) affected C and N in similar proportions. With PS and U having been applied to the mulch, substantial volatilization may have occurred (Freney et al., 1992; Pain et al., 1989; Sommer et al., 2003). For the DS treatment, the decrease in the C:N ratio observed from day 28 reflects relative enrichment of the N in the remaining mulch particles compared to C. Pig slurry is a liquid fertilizer (0.8% DM) with ammonia as the predominant form of N (77.8%, Table 1). Urea can be rapidly transformed in ammonia in the climatic and agricultural conditions of sugarcane crop (Pinheiro et al., 2019). Therefore, both PS and U may have contributed to decomposition of the mulch by providing both readily available N (ammonia) and easily assimilable C (low IROC for PS, Table 1); however, these elements are also highly susceptible to leaching by rains. Conversely, we hypothesize that the chemical form of N in DS (90% of N in organic form, Table 1) and the physical form of DS (solid pellets, 0.5 cm diameter and 1-2cm length cylinders) may have favored DS retention in the mulch layer and improved the accessibility to decomposers, leading to better recycling of the mulch-fertilizer mixture N by microorganisms. These hypotheses warrant further verification.

For all treatments, the kinetics of the mulch loss showed that the C:N ratio of the remaining particles dropped from day 49 to the end of the experiment, indicating that the mulch-C decomposition was sustained by efficient recycling of microbial N by the mulch decomposers, which suggests very efficient microbial turnover during decomposition, although the contribution of soil N to decomposers through fungi hyphae cannot be excluded (Frey et al., 2003).

These results stress the importance of considering the coupling of C and N inputs as an interactive system, which is particularly important from an agronomic point of view. As mulch decomposition depends on moisture conditions and N availability, considering not only the amount and chemical form of the applied N but also the physical form of the fertilizer and its interactions with the mulch properties (e.g., thickness and density) appears to be critical to better understanding the kinetics of C and N following the addition of mulch fertilizer mixtures, as recently underlined by Dietrich et al. (2017).

4.2. Short-term Effects of N fertilizer + Sugarcane Mulch Mixtures on CO_2 and N_2O Fluxes

In the short term, the CO_2 measurements provided a more sensitive picture than the mulch-C loss of the interactions between mulch-C and fertilizer-N. Indeed, we observed that the kinetics of the CO_2 emission fluxes with sewage sludge (DS) and urea (U and U0.5) treatments responded the most to the water inputs, which is consistent with the results from a previous study in tropical conditions (Wang et al., 2016). On average, the CO_2 fluxes were not affected by the form of N added, confirming the observations from the mulch-C decomposition measurements. We hypothesize that the low mulch water content resulted in moisture acting as the limiting factor of mulch-C mineralization. This result is in line with those obtained by Aguilera et al. (2013) and Charles et al. (2017) in their reviews for Mediterranean and global contexts, respectively. Conversely, with pig slurry (PS and PS0.5), the kinetics of the CO_2 emissions exhibited weak relationships with precipitation (Fig. 3c). We assume that the high peak of CO_2-C observed with the PS and PS0.5 treatments during the first 24 h following the addition of pig slurry resulted from a rapid release of CO_2 from the pig slurry itself, which is rich in carbonates (Chantigny et al., 2002) and contains a significant fraction of biodegradable C (see low IROC, Table 1). Carbon from PS may have been rapidly mineralized by microorganisms (Rochette et al., 2000), which is consistent with the PS humus potential at nearly 60% lower than that of DS (Table 1).

In the short term, the N_2O emissions responded substantially more to N fertilizer treatment than the CO_2-C emissions, which was expected. When expressed per hectare or per unit of added N, the application of pig slurry (PS and PS0.5) led to the highest N_2O emissions compared to the urea treatments (U and U0.5), while dried solid sludge (DS and DS0.5) had the lowest N_2O emissions. These results are consistent with the high emissions from liquid fertilizers and low emissions from synthetic or solid organic fertilizers observed by Gregorich et al. (2005). This finding is also consistent with the results from Meijide et al. (2005), who reported that the high concentrations of NH_4^+ in pig slurries induce relatively high N_2O-N emissions. Furthermore, the combined effect of N speciation, the liquid form of the pig slurry (de Oliveira et al. 2013) and the presence of available C from the mulch may lead to high microbial respiration and thus O_2 consumption. N-NH_4^+ from liquid pig slurry may have dropped from the mulch to the first centimeters in the soil. Therefore, not only nitrification of the NH_4^+ but also possible subsequent denitrification in hotspots at the interface between soil and mulch, as reported by Sommer and Møller (2000) and Paillat et al. (2005) for composting, may have been responsible for the higher N_2O production when pig slurry was added to the mulch compared to the other treatments. To compare total emissions from the different fertilizers, other emissions must be considered, such as those occurring during storage before their addition to mulch. Compared to pig slurry, the apparent higher stability of N from sewage sludge after its addition to mulch may also be due to the higher volatilization and denitrification during its previous storage stage (Rigby et al., 2016). Indeed, the sewage sludge was submitted to anaerobic digestion (for the production of methane), where a part of the organic N may have been transformed in mineral forms (mainly ammonium). During the first two post-digestion phases (centrifugation and drying), ammonium in the digestate may have been a source of volatilized ammonia and even N_2O emissions during nitrification. The digested sewage sludge was consequently limed during the last (third) post-digestion phase in order to sanitize and stabilize the product. It is well documented that the addition of lime favors the volatilization of ammonia from ammonium.

4.3. Effects of Sugarcane Mulch Quantity on Mulch Decomposition and CO_2 and N_2O Fluxes

We found that the rates of mulch decomposition and mulch-C loss were proportional to the initial mulch quantity for the two quantities tested here (5 and 10 Mg DM ha^{-1}), which also indicates that the amount of mulch lost was two times higher when the mulch quantity was doubled. The limiting factors of decomposition with a higher initial mulch mass may be i) the proportion of the mulch in contact with the soil (a smaller proportion with thicker mulch), which may induce lower or delayed microbial colonization and lower mulch moisture, and ii) the availability of nutrients to decomposers, particularly N, as the soil is supposed to “feed” the microbial N demands in the mulch, mostly by diffusion from the soil to the mulch layer (Findeling et al., 2007). Our results suggest that none of these processes were involved in our experimental conditions, and our results agree with the recent work of Dietrich et al. (2017), who also found that the rate of decomposition of sugarcane mulches in southern Brazil was similar regardless of the initial mulch quantities, which ranged from 4 to 12 Mg DM ha^{-1}.

The amount of crop residues remaining as mulch also influences the soil temperature, soil moisture and the rate of soil cover (Fracetto et al., 2017). Therefore, we expected that greater mulch cover would foster emissions by creating microclimate conditions that favor microbial activity. However, the mean CO_2-C flux was proportional to the initial mulch quantity in the U treatment since the flux observed in the U treatment was twice that observed in the U0.5 treatment. This relationship was not observed for the pig slurry treatments, but the main source of CO_2-C was due to the initial peak, which probably originated from the slurry itself.

For N_2O-N, there appeared to be a trend of higher N_2O fluxes when higher amounts of sugarcane mulch were added, although the change was not significant due to the high spatial and temporal variabilities of the emissions in the clayey soils (Grossel et al., 2014). Carmo et al. (2013) also found that higher emission factors for vinasse and filter cake applications were observed only when the mulch quantity exceeded 14 t DM ha^{-1}. In our study, the results suggest that despite the higher C:N ratios of the mixtures in the U and PS treatments compared with the U0.5 and PS0.5 treatments, the conditions for N_2O emissions were slightly more favorable when higher quantities of mulch were initially left at the surface. This result may be due to the enhanced soluble C and microbial activities in the mulch, as observed by Iqbal et al. (2015) in controlled conditions with mixed-species mulches, and higher moisture in the soil under thicker mulch as observed by Pinheiro et al. (2019).

5. Conclusion

The joint recycling of crop residues and organic fertilizers, such as sugarcane trash and pig slurry or sewage sludge in the context of this study, are likely options for sustainable management of sugarcane growing systems since substitution of urea with pig manure or solid sludge did not alter the overall C and N fluxes. However, while the form of N that was applied on the mulches did not influence their rate of decomposition, the N form influenced N_2O emissions in the short term, with more N_2O emitted from the liquid form (pig slurry). Doubling the mass of sugarcane residues did not significantly influence the fluxes, indicating that our main hypothesis regarding the effects of the C:N ratios of mulch-fertilizer mixtures on C and N transformation was invalid. However, leaving more mulch on the soil increased the amount of mulch-C and mulch-N remaining in the soil over the short term, and the amount of C and N inputs into the soil over the long term. Saving some straw to maintain or increase the amount of carbon sequestered in the soil and combining it with organic fertilization in place of mineral fertilization may therefore not only favor cost savings regarding N fertilizer purchase as well as improving soil characteristics linked to soil organic matter content but may also improve the overall GHG balance of the sugarcane crop, which must be considered when determining strategies for exporting crop straw for cattle or energy purposes. In our conditions, the minimum rate of sugarcane trash (5 t DM ha^{-1}) that cane growers want to keep on the field in order to control weeds seems to be an appropriate amount to maintain C and N inputs to soil while allowing the possibility to export substantial amounts of sugarcane trash (7 t DM ha^{-1}) for cattle or energy uses. Locally, this last option represents an interesting bioenergy alternative to imported coal, a nonrenewable resource that emits GHG during extraction, sea transport over 2800 km, and combustion in a coal-fired power station. Lastly, and more generally, this work underscores the importance of jointly studying C and N dynamics and considering both soil carbon and gas emissions as well as soil physical conditions to assess the greenhouse gas balance of farming practices and fossil-C substitutions.

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CHAPITRE 3

IV. Fertilisation de la canne à sucre avec des déchets organiques en présence de paillis : Comment la composition des fertilisants organiques et la quantité de paille influencent la dynamique du C et le N_2O émis ?

Résumé

La paille de canne à sucre est de plus en plus prisée pour son potentiel de production d'agrocarburants de deuxième génération. L'importante surface cultivée en canne à sucre présente un potentiel intéressant de recyclage des produits résiduaires organiques, en tant que fertilisants organiques. Par conséquent, l'exportation de la paille pour une utilisation par ailleurs, comme la production d'énergie ou de matériaux, et le recyclage des déchets organiques dans les systèmes de canne à sucre restent un défi en raison du manque de connaissances sur l'impact environnemental sous-jacent. Les objectifs de ce travail étaient de comprendre l'effet des applications combinées des fertilisants organiques et du paillis de canne à sucre, sur la dynamique de la décomposition du paillis et sur les émissions de gaz à effet de serre (GES) (N_2O et CO_2). L'effet de la quantité différente du paillis a été mesuré en comparant des mélanges avec des rapports C:N similaires, en appliquant de l'urée avec 10 ou 20 t MS ha^{-1} de paillis (U10L vs. U20L, respectivement), et les traitements témoins sans fertilisation avec 10 ou 20 t MS. ha^{-1} de paillis (C10 vs. C20, respectivement). L'effet du rapport C:N a été mesuré en comparant des mélanges avec une quantité identique de paillis, mais une quantité d'azote différente sous forme de lisier de porc à 138 ou à 1700 kg N ha^{-1} (respectivement PS20H vs. PS20L) et de boues de station d'épuration à 121 ou à 1211 kg N ha-1 (DS20H vs. DS20L, respectivement). L'effet du type d'engrais-N a été mesuré en comparant les traitements avec des rapports C:N similaires et une quantité identique de paillis, mais avec des fertilisants-N différent, comme le lisier de porc, les boues de station d'épuration digérées et séchées et l'urée (PS20L vs. DS20L vs. U20L, respectivement). Nos résultats ont montré que la quantité initiale de paillis n'influençait pas le taux de décomposition relatif et les émissions de GES. Ainsi, la quantité de C retournée au le sol peut être prédite avec précision à partir de la quantité initiale de paillis et de sa constante de dégradation. L'ajout d'environ 10 fois plus de N dans les PS20L et DS20L par rapport aux PS20H et DS20H a eu pour effet l'augmentation des taux de décomposition du paillis et des émissions de N_2O. Le taux de décomposition du paillis-C était plus élevé avec les fertilisants organiques par rapport à l'urée, ce qui a été attribué aux différences de qualité physique et chimique (N) entre le lisier de porc, les boues de station d'épuration et l'urée. La dynamique des émissions de GES a été corrélée principalement à la teneur en N minéral du sol et à l'humidité du sol. Ainsi, les flux de GES les plus élevés ont été mesurés avec PS20L immédiatement après l'application du lisier de porc, riche en N minéral et en eau. Nous concluons que le seul rapport C:N des mélanges de paille/fertilisants n'est pas un indicateur suffisant pour les émissions de N_2O. Les caractéristiques chimiques et physiques des fertilisants influent sur d'autres facteurs importants, tels que la concentration en O_2, la

localisation de la biomasse microbienne et des éléments nutritifs, et par répercussion sur les dynamiques et la magnitude d'émissions de GES.

Sugarcane fertilization with organic wastes in the presence of straw mulch: How the composition of the organic fertilizer and the amount of straw influence the mulch C dynamics and the N_2O emitted?

Abstract

The sugarcane straw is growingly valued biomass for its potential for second generation biofuel production. The large cultivation area for sugarcane manifests a great potential for organic wastes recycling by organic fertilization. Hence, mulch removal for further energy or row material uses and organic waste recycling in sugarcane systems remain challenging, because of the lack of knowledge about the impact on the environment. The objectives of this study were to understand the effect of combined applications of organic fertilizers and sugarcane residues as mulch, on the dynamics of mulch decomposition and on the greenhouse gas emissions (GHG) (N_2O and CO_2). The effect of different mulch quantity was estimated comparing mixtures with similar C:N ratios, using urea with 12 or 22 t DM ha^{-1} of mulch (U10L vs. U20L, respectively), and control treatment without fertilization with 11 or 22 t DM ha^{-1} of mulch (C10 vs. C20, respectively). The effect of the C:N ratio was assessed by comparing mixtures with identical mulch quantity added, but different N application rate as pig slurry at 138 or 1700 kg N ha^{-1} (PS20H vs. PS20L, respectively), and sewage sludge at 121 or 1211 kg N ha^{-1} (DS20H vs. DS20L, respectively). The effect of the type of fertilizer-N was assessed by comparing treatments with similar C:N ratios and identical mulch quantity, but different N-fertilizer type, as liquid pig slurry, digested and dried sewage sludge and urea (PS20L vs. DS20L vs. U20L, respectively). Our results showed that the initial mulch quantity did not influence the relative decomposition rate and GHG emissions. Thus, the amount of C return to the soil can be fairly predicted from the initial mulch quantity and its degradation constant. Adding ~10 times more N in PS20L and DS20L compared to PS20H and DS20H, led to higher mulch decomposition rates and N_2O emissions. Mulch-C decomposition rate was higher with organic fertilizers compared to urea which was attributed to the different physical and chemical (N) forms between pig slurry, sewage sludge and urea. The GHG emission dynamics responded mainly to mineral N content of the soil and to soil moisture, thus the highest GHG fluxes were measured with PS20L immediately after application of pig slurry, rich in mineral N content and

moisture. We concluded that the C to N ratio of the straw-fertilizer mixtures is not sufficient to determine N_2O emissions, and the chemical and physical characteristics of fertilizers, influencing other important factors, such as O_2 concentration or location of microbial growth and nutrients can influence the emissions.

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Supplementary material

1. Introduction

Sugarcane straw is growingly valued biomass for its potential for second generation biofuel production (Cantarella et al. 2013, Leal et al. 2013a, Sordi and Manechini 2013, Aditiya et al. 2016). This is because the transformation of the straw is considered as a sustainable solution to avoid conflicts between food, feed and energy land uses in agriculture (Ladanai and Vinterbäcck 2010). Sugarcane represents the largest biomass production worldwide with 1.9×10^9 tones produced in 2016. Therefore, it has a high potential of straw production (between 10 to 30 t DM ha^{-1}), which may vary between sugarcane varieties, pedoclimatic conditions, ratoon number and harvesting technique (Hassuani et al. 2005, Franco et al. 2013, Leal et al. 2013a). (WRB-FAO 2014). Furthermore, the large cultivation area for sugarcane cropping systems (2.7×10^7 ha^{-1} in 2016) manifests a great potential for organic wastes recycling by organic fertilization. The association of both mulch and organic fertilizers is seen as a lever to tackle soil quality and fertility issues, and to increase C storage, that is targeted by the international research program ‘4 per mille’ (Minasny et al. 2017).

Hence, mulch removal for further energy or row material uses and organic waste recycling in sugarcane systems remain challenging, because of the lack of knowledge about the impact those practices may induce on the environment. It is well known that total removal of sugarcane residues from the soil surface can compromise critical soil functions (Lisboa et al. 2019), such as the buffer effect on soil structure (Johnson et al. 2016), on soil temperature and moisture (Corrêa et al. 2017), but also the weed and the erosion controls (Lal 1974, Cerri et al. 2011, Comte et al. 2012, Azevedo et al. 2014, Mthimkhulu et al. 2016). There is no agreement about the impact of mulching on greenhouse gases (GHG) emissions, some studies reported the enhancement of GHG emissions with mulching, with higher N_2O emissions (Cerri et al. 2009, Carmo et al. 2013, Fracetto et al. 2017, Pinheiro et al. 2019). On the other hand Chen et al. (2017) have not found an effect of mulch compared to control (no-mulch). And recently some authors have even measured a decrease of N_2O emissions, when increasing mulch biomass (Congreves et al. 2017, Pitombo et al. 2017).

The fertilizers used for sustaining crop growth are synthetic or organic, and their physical and biochemical quality can differ greatly. It remains unclear whether organic or synthetic fertilizers are higher emitters of GHG, as studies have been led in various environmental conditions with a multitude of fertilizer qualities (Snyder et al. 2009, Abalos et al. 2016). On the contrary, it is well known that the potential of C and N mineralization of organic inputs is tightly related to their physicochemical characteristics (Lashermes et al. 2010). Therefore, in a recent meta-analysis, Charles *et al.* (2017) pointed out the need to classify N fertilizers more precisely, according to their physical and chemical properties, which are driving their GHG potential. Combining straw and organic fertilizers, as promoted for sustainable agricultural management, has been poorly studied in sugarcane field conditions. In a maize-wheat cropping system, N_2O emissions were found to increase with N and straw quantity increase (Huang et al. 2017). Hence, Chantigny et al. (2001) found synergetic relations for C decomposition and antagonistic relation for N_2O emissions, probably due to higher N immobilization, after fertilization with pig slurry in a barley-straw managed system. In another field study, Carmo et al. (2013) have found that combining sugarcane straw with organic fertilizers in liquid form, rich in dissolved organic carbon (vinasse), fueled nitrification, denitrification and microbial respiration, and lead to higher N_2O and CO_2 emissions, compared to mixture with urea or a solid amendment (filter cake). Increasing sugarcane straw left on the surface have led to increased GHG emissions too (Carmo et al. 2013). In a laboratory experiment Aita et al. (2012) showed that the mineralization of straw-C, applied on soil surface, was increased when pig slurry was added compared to a situation where straw was applied alone. Another study showed that straw-C mineralization was not increased when oat straw and sewage sludge were incorporated together, although N from sewage sludge was immobilized in a straw + sewage sludge mixture compared to sewage sludge applied alone (Giacomini et al. 2015). Finally, interactions have been largely discussed for litter mixtures (Makkonen et al. 2012), and have been reported to be the result of N concentration differences between the two components of the mixtures (Thomas 1968, Hoorens et al. 2003), or due to a secondary compounds, such as lignin-like or tannins (Schimel et al. 1998). Wardle et al. (2003) pointed out the physical quality as a driver for synergetic C and N transformation dynamics, when mixing feather mosses with high water holding capacity and a N-rich litter. Although the mechanisms at stake that lead to C and N transformation interactions were discussed for litter mixtures, interaction mechanisms between straw and fertilizers in mixtures are rare or have been discussed only in controlled conditions (Kyulavski et al. 2019b).

However, the contrast between straw and organic fertilizer's characteristics is more pronounced than between the litters of mixtures tested in upper cited studies.

Therefore, the objectives of this study were to understand the effect of combined applications of organic fertilizers and sugarcane residues as mulch on the soil surface on a sugarcane crop in field, on the dynamics of mulch decomposition and on the Greenhouse gas emissions (N_2O and CO_2). To better understand the interactions between the mainly carbonaceous substrate (straw) and the mainly nitrogenous substrate (fertilizer), we have chosen to vary either the form of N applied, or the amount of N supplied or the initial mass of sugarcane mulch. In this way, we have explored mixtures of various composition with low and high C:N ratios, even if this means moving away from the usual agricultural conditions of farming practices in the La Réunion area.

2. Materials and Methods

2.1. Site

The experiment was conducted on SOERE PRO (*i.e.* International network for long-term observation and experimentation systems for environmental research on organic waste products), on the experimental station at La Mare (Sainte-Marie), Réunion island (20°54'12.2"S 55°31'46.6"E). This zone is characterized by a tropical climate with an average annual temperature of 24.6°C. The lowest mean temperature is 21.9°C in August and the highest temperature 27.2 °C in February. The annual average rainfall is 1600 mm, and 80% of which falls between December and June. The soil is a Nitisol (WRB-FAO 2014) and the top 0-20 cm layer sampled contains high clay and fine silt content (> 70%), with Ca 140, Mg 37.5, K 24.8, P 100 mg kg^{-1}, CEC 109 mmol kg^{-1}, N 1.82 g kg^{-1} and soil organic carbon (SOC) 20.6 g kg^{-1}. The sugarcane trial was planted on March 2014 from viable buds of R579 sugarcane variety and placed in 1.5 m row spacing in 6 rows per plot. The experiment was conducted during the second ratoon of sugarcane (from November 2015 to October 2016) on 10 plots (10.5 m × 28 m) over 3 separated blocks (replicates). Each block that was cultivated with sugarcane included 3 plots that received the 3 fertilizer types: pig slurry, digested solid sewage sludge and urea. As sugarcane growers always leave sugarcane mulch at the soil surface for containing weed growth, a control plot received only sugarcane mulch, with no fertilizer addition.

2.2. Experimental Design

The treatments aimed to study the effect of mulch quantity, the type of fertilizer-N and the C to N ratio of the mixtures, on mulch decomposition and N_2O emissions. To do this, two initial quantities of sugarcane trash: 20 t DM ha^{-1} (corresponding to the amount of sugarcane trash measured at harvest) and half this amount, i.e. 10 t DM ha^{-1}, and three forms of fertilizer-N were applied on the sugarcane trash: pig slurry, solid sewage sludge and urea. N fertilizers were applied over the mulch under their specific physical form: liquid for pig slurry, solid dried and limed pellets for digested sewage sludge, and solid granules for urea. The main characteristics of organic materials are shown in Table 1.

Table 1: *Physical and chemical characteristics of the organic materials*

Organic material	DM	TOM	TOC	Total N	$N\text{-}NH_4^+$	C:N	pH_{H2O}
	(g kg^{-1})					-	-
Pig slurry	65	46	25	4	2	6	8
Digested sewage sludge	913	544	302	44	4	7	12
Sugarcane mulch	891	921	414	4	*n.d.*	95	*n.d.*

DM = Dry Matter is expressed in g kg^{-1} fresh weight, TOM = Total Organic Matter, TOC = Total Organic Carbon, total N, $N\text{-}NH_4^+$ are expressed in g kg^{-1} dry weight, *n.d.* = not determined

The sugarcane residues, consisting of leaves (blades and sheaths) and tops, were collected immediately after sugarcane harvesting, by randomized spatial sampling (n=113). The sugarcane residue samples were oven dried at 60°C until reaching constant weight (72 hours). Eight treatments were studied, including six mixtures and two controls without fertilizer application. The C and N amounts (kg ha^{-1}) and their C:N ratios for the different treatments are reported in Table 2.

Table 2: *Initial total C and N inputs and C:N of the mixtures (sugarcane mulch + N fertilizer) and control treatments*

Treatment name	**Organic input**	**C total**	**N total**	**C:N**
		(kg ha^{-1})	**(kg ha^{-1})**	**-**
PS20H	pig slurry	825	138	43
	sugarcane mulch	9208	97	
PS20L	pig slurry	10173	1700	11
	sugarcane mulch	9208	97	
DS20H	sewage sludge	821	121	46
	sugarcane mulch	9208	97	
DS20L	sewage sludge	8210	1211	13
	sugarcane mulch	9208	97	
U10L	urea	133	311	13
	sugarcane mulch	4604	49	
U20L	urea	289	674	12
	sugarcane mulch	9208	97	
C10	-			95
	sugarcane mulch	4604	49	
C20	-			95
	sugarcane mulch	9208	97	

H = high C:N, low N input
L = low C:N, high N input

The treatments aimed to detect the effects of mulch quantity, C:N ratio of the mixture and the type of fertilizer on mixture decomposition and GHG emissions. The effect of different mulch quantity of sugarcane was estimated comparing mixtures with similar C:N ratio, (comparison U10L vs. U20L, C10 vs. C20). The effect of the C:N ratio was assessed by comparing mixtures with identical mulch quantity added, but different N application rate (comparison PS20H vs. PS20L, DS20H vs. DS20L). The effect of the type of fertilizer-N was assessed by comparing treatments with similar C:N ratio and identical mulch quantity, but different N-fertilizer type (comparison PS20L vs. DS20L vs. U20L).

The decomposition of mulch-fertilizers mixture was measured using squared litterbags, while the N_2O (and CO_2) fluxes from mulch-fertilizers mixture were measured using cylindrical chambers (Kyulavski et al. 2019a). The 3 replicates of each treatment were located on 3 separated blocks. The replicates for control treatments were located on the same plot, which was the only one that did not receive N fertilizer during the experiment (Fig. 1). Rainfall, temperature and atmospheric pressure were measured continuously at the nearest meteorological station (20°88’S 55°52’E).

Figure 1: Design of litterbag and cylindrical chamber on-field experiments. Litterbags were used for measuring the decomposition of mulch or mulch-fertilizers mixtures, while cylindrical chambers were used for measuring CO_2 and N_2O emissions from mulch or mulch-fertilizers mixtures. View of the plot with control treatments, with two quantities of sugarcane mulch, and 3 replicates.

2.3. Measurement of Mulch Decomposition with a Litterbag Approach

The dynamics of mulch-C during decomposition was assessed using a litterbag approach. To do so, litterbags (30 cm × 30 cm × 15 cm) of white polyvinylchloride garden grid, with a mesh of 1 cm (assuring access for most of the macrofauna decomposers), were prepared. Litterbags' soil areas were bereft of any visible particulate organic matter before litterbags deposition. The litterbags were placed in the central row space of each plot, with 1.5 m of spacing between each other. For the control the litterbags were placed in the 3 inner row spaces, on a unique plot (1.5 m of spacing). The litterbags were fixed at the surface of the soil using U-shaped metallic clamps. The mulch was filled into opened litterbags and the N-fertilizers were added within the litterbags over the mulch. Litterbags were then sealed by stapling the open side. The experiment was launched at intervals of 7 days, for each of the 3 types of fertilizers, to avoid potential cross-contamination of gas measurements. The litterbags were sampled 6 times up to 344 days after on-field deposition. For the 8 treatments (with 3 replicates and 6 times of analysis) a total of 144 litterbags were settled.

2.3.1. Soil Sampling Under Litterbags

The soil located under the litterbags was sampled (0-10 cm layer) at each litterbag sampling occasion. Soil samples were promptly transported in cooled conditions to the laboratory where they were stored at 4°C before analysis.

2.3.2. Soil Temperature Measurement

The soil temperature was recorded every 15 min (CR1000 datalogger, Campbell Scientific®, USA) using the time domain reflectometers (CS655, Campbell Scientific, USA) probes buried at 7.5 cm depth under the C10, C20, DS20L, PS20L and U20L treatments in the first block.

2.4. N_2O and CO_2 Emissions Using Cylindrical Chambers

N_2O and CO_2 emissions were measured 13 times for all treatments, from day - 6 before fertilizer application until day 192 after application. The gas samples were collected by the static-opaque chamber method (Kyulavski et al. 2019a). A cylindrical socket (d = 23.5 cm) was driven into the soil in the center of each plot (*i.e.* in the sugarcane row space) 2 weeks before the first sampling, with 3 replicates per treatment. At each sampling time, a cylindrical lid was fit onto the socket, forming a closed chamber of isolated atmosphere. The lids had a small fan that was activated at least 30 seconds before sampling. Homogenized atmosphere was sampled at chamber closure and then 30 and 60 minutes after. The sampling consisted of inserting a blood collection needle (TERUMO Neolus) through a septum on the lid and collecting the gas by filling a 20 mL syringe tank (BD Discardit II). The entire tank content was then introduced into a previously evacuated vial of 12 mL (Labco Exetainer ®). The vials containing gas samples were then sealed with candle wax to prevent leakage during transportation for analysis.

2.5. *Analytical Methods*

After sampling, the remaining mulch was gently washed with water to remove soil contamination, and oven dried (60 °C, 72 h) until reaching constant weight. Sugarcane residues were ground to 1 mm, and total N and C were measured by dry combustion elemental analysis (Dumas method). Biochemical fractioning was performed on the mulch samples for measuring the soluble, neutral detergent fibre (NDF), acid detergent fibre (ADF), and acid detergent lignin (ADL) contents. The Van Soest soluble fraction (SOL), the hemicellulose-like (HCE), cellulose-like (CEL) and lignin-like (LIG) fractions were determined using the Van Soest method (AFNOR 2016b). Ash content was determined by loss on ignition (550 °C).

CO_2 concentrations were determined by gas chromatography with a thermal conductivity detector (MTI M200 Micro Gas Chromatography, USA) and N_2O concentrations were determined by gas chromatography with an electron capture detector (GC-ECD Varian 3800, the Netherlands). Ammonium and nitrates contained in the soils sampled under the litterbags were extracted by 1 M KCl solution and then measured by the spectrometric method NF-ISO 14256 (2000).

2.6. *Calculations and Statistical Analysis*

The mulch-C decomposition rate was calculated at each destructive sampling of litterbags as the proportion of the initial sugarcane mulch-C remaining in the litterbags. Total water input was calculated (in mm) as the sum of rainfall and irrigation. The environmental variables used for gas concentration calculations were temporally matched with gas sampling periods.

The linear increase of gas partial volumes was verified between chamber closure and the last sampling (60 min after closure) by independently calculating the coefficient of determination for each treatment (3 samples from 0, 30 and 60 seconds after chamber closure). Samples with $R^2 < 0.7$ were considered outliers, and their values were not used to calculate the fluxes. The gas emission fluxes were calculated using the relation:

$$f_i = \frac{\Delta G}{\Delta t} \times \frac{V}{V_m} \times \frac{M}{A} \qquad \text{Eq. 1}$$

where f_i is the emission flux (µg m^{-2} h^{-1}), ΔG is the increase of CO_2 or N_2O partial volumes (µL L^{-1}) within the chamber during the enclosure period Δt (h), V is the volume of the chamber (L), V_m is the molar volume of gas at the sampling temperature (L mol^{-1}), M is the molar mass of C contained in CO_2 or the molar mass of N contained in N_2O, respectively, A is the area covered by the chamber socket (m^2).

The mean fluxes per treatment were calculated from the *ad hoc* samplings between 0 and 192 days and were expressed as the absolute mean flux (g N_2O-N) using the relation:

$$F_i = \frac{1}{s \times n} \sum_{i=1}^{s \times n} f_i \qquad \text{Eq. 2}$$

where F_i is the mean N_2O-N flux per treatment or control, f_i is the measured flux of sample i, s is the number of sampling occasions (from day 1 up to day 192, $s = 12$), n is the number of replicates ($n = 3$).

The magnitude of the CO_2-C mean fluxes towards control CO_2-C summed fluxes were calculated using the relation:

$$M_{ip} = \mathrm{Log}_{10}\left(\frac{F_{ip}}{F_{Cp}}\right) \qquad \text{Eq. 3}$$

where Mi_p is the magnitude of the CO_2-C summed fluxes per treatment i, F_{ip} is the calculated (eq. 2) summed flux for the treatment i over the period p (0-49 or 49-192 days following N applications), F_{Cp} is the calculated (eq. 2) mean flux for the control C over the period p (0-49 or 49-192 days following N applications).

The CO_2-C or N_2O-N dynamic fluxes (mg or g kg^{-1} soil) were calculated by dividing the respective mean fluxes by the bulk density multiplied by the surface of measurement of the 0-10 cm soil layer.

The normality of the distribution and homoscedasticity were verified using Shapiro & Wilk normality and Bartlett's tests, respectively. When those conditions were met, a one-way ANOVA followed by Tukey pairwise comparison test was applied. Otherwise, a non-parametric test (Kruskal-Wallis) was used to detect differences, followed by Wilcoxon’s test adjusted by Benjamini, Hochberg, and Yekutieli method. All statistical analyses were conducted with R software (v3.2.2, R Core Team, 2015, R Foundation for Statistical Computing, Vienna, Austria), powered by RStudio (v0.99.484 – 2009-2015, RStudio Team, 2015, Integrated Development for R. RStudio Inc., Boston, MA).

3. Results

3.1. Mulch Decomposition

The C loss and C:N ratio dynamics showed in this section were measured on a water-washed sugarcane straw, after harvesting for analyses, and not on the mixtures as settled on the field.

The decomposition dynamics of the mulch was characterized by a high rate of C loss for all treatments during the first 5 months after litterbag deposition (Fig. 2). After this initial period and until the end of the measurements, the decomposition rate decreased. At the end of the experiment (day 344) the remaining mulch-C ranged from 4 ± 1 % added mulch-C for PS20L to 12 ± 4 % added mulch-C for PS20H.

The mulch quantity did not influence the relative C loss rate, neither for the controls (C10 vs. C20) nor for urea-N treatments (U10L vs. U20L), (Fig. 2a). After 344 days, the remaining mulch-C ranged from 9 (± 2 %) to 11 % (± 3 %) for urea treatments and controls. For a given N treatment, the mulch quantity had no significant effect on the dynamics of the mulch C to N ratio (Fig. 2b), but this ratio evolved differently depending on whether urea was added or not. For C10 and C20 the mulch C to N raised, indicating a higher N loss than C from mulch within the first 49 days of the measurements. The C to N ratio decreased after and the C10 and C20 curves reached those of the U10L and U20L after 120 days. For U10L and U20L the C to N ratio decreased all along until 120 days and then stabilized to 40 (± 5). The final C to N ratios were 42 (±1) and 49 (±6) for U20L and U10L, and 44 (±3) and 44 (±2) for C10 and C20, respectively.

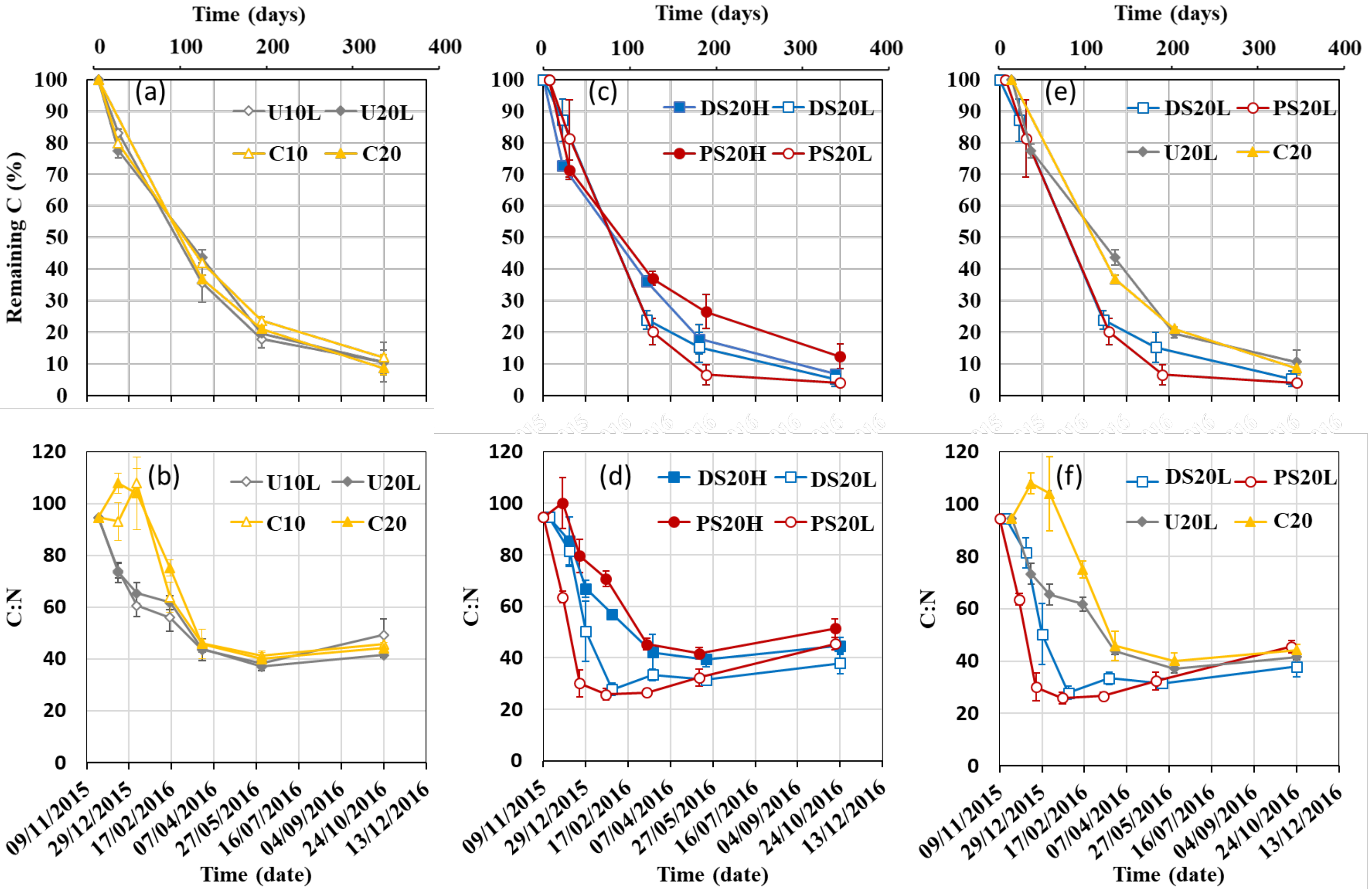

Figure 2 : Comparison of the dynamics of Mulch-C remaining and C: N ratio, according to : mulch quantity (a, b), C:N ratio of the mixtures (b, c), and type of N-fertilizer (e, f) for the treatments: 11 t DM ha^{-1} sugarcane mulch (C10), 22 t DM ha^{-1} sugarcane mulch (C20), *high C:N mixtures* with 22 t DM ha^{-1} mulch + sewage sludge (DS20H) and 22 t DM ha^{-1} mulch + pig slurry (PS20H); *low C:N mixtures* with 22 t DM ha^{-1} mulch + sewage sludge (DS20L), 22 t DM ha^{-1} mulch + pig slurry (PS20L); 22 t DM ha^{-1} mulch + urea (U20L), 11 t DM ha^{-1} mulch + urea (U10L). Bars in graph indicate standard error of the mean (n=3).

The C to N ratios of a given straw + organic fertilizer mixture influenced the straw C-loss rate and dynamics (Fig. 2c). The decomposition rate in the first 24 days tended to be lower for PS20L and DS20L (low C:N), compared to PS20H and DS20H (high C:N), respectively. This trend was then reversed between 24 and 120 days, with remaining-C of PS20L < PS20H on days 120 and 190 (p-value < 0.05). At day 344 remaining C was 12 ± 4 % (PS20H), 7 ± 1% (DS20H), 5 ± 2 % (DS20L) and 4 ± 1 % (PS20L) with no significant difference according to the initial C:N ratio values. The C:N ratio for all the mixtures with organic fertilizers decreased during the first 3 months then from day 185 to day 344 the C to N ratio slightly increased, which was the consequence of a higher rate of N loss from mulch (Fig. 2d). The dynamics of change in the C to N ratio of the mixtures sampled over time was markedly influences by the initial C to N ratio of the mixtures, with low C:N ratio mixtures (U10L; U20L, DS20L, PS20L) having a faster decrease of the mulch C to N ratio than the high C:N ratio mixtures and controls (C10, C20, PS20H, DS20H).

The type of fertilizer-N added affected the decomposition dynamics with higher C-loss observed with organic fertilizer during the first 5 months, compared to urea fertilizer and control (Fig. 2e). This led to a significantly lower (p-value < 0.05) remaining mulch-C for DS20L and PS20L treatments compared to U20L and the control (C20), between day 120 and day 190. Adding urea-N proportionally to mulch quantity, did not influence the decomposition rate and dynamics compared to controls (U10L vs. C10, and U20L vs. C20), despite the lower C to N ratio of the mulch-urea mixtures. At day 344, remaining C was 5 ± 2 % (DS20L), 4 ± 1 % (PS20L), 11± 4 % (U20L) and 9± 2 % initial mulch-C (C20), with no significant difference between treatments. Mulch C:N decreased within the first 120 days for all N-fertilized treatments (Fig. 2f), with faster decrease for the low C:N mixtures (DS20L and PS20L) than for the urea-N fertilizer and control treatments (U20L and C20).

3.2. Evolution of the Biochemical Composition of the Mulch

For all treatments, the relative enrichment in the lignin-like fraction (LIG) of the remaining mulch particles significantly increased as decomposition proceeded, as expected (Fig 3a). The fastest and highest increase in LIG was observed for PS20L, which showed also the fastest mulch- C decrease. On the contrary, the change in LIG was significantly smaller for C10 and C20 (Fig.3a). For a given mixture type, there was a tendency that the final LIG content was higher for the low compared to the high C:N ratio, particularly with pig slurry (PS20L vs. PS20H) although the differences were not significantly different. The relation between remaining C (%) and the % LIG changed depending on whether N-fertilizer was added or not, with a lower relative increase in LIG content for C10 and C20 at similar rate of mulch-C remaining (Fig. 3b).

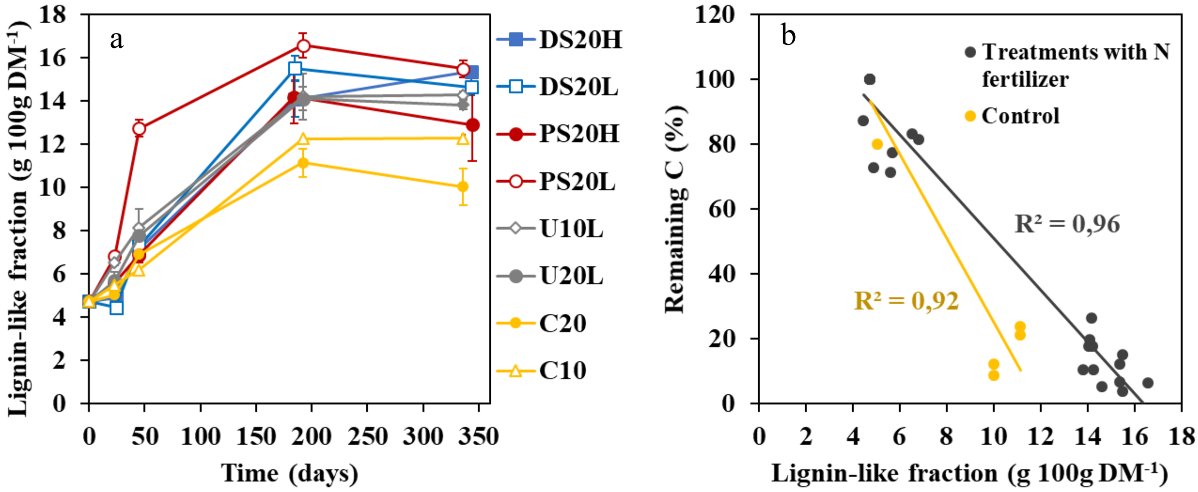

Figure 3 : (a) Evolution of the lignin-like fraction content (g 100g^{-1} dry matter) of the remaining sugarcane mulch during 344 days of decomposition, and (b) relationship between remaining C (% of initial) and lignin-like fraction (g 100g^{-1} dry matter) for N-mulch mixtures and for the controls. Bars in graph indicate standard error of the mean (n=3).

Controls: 11 t DM ha-1 (C10) and 22 t DM ha^{-1} of sugarcane mulch (C20),

Mixtures high C:N ratio: 22 t DM ha^{-1} mulch +sewage sludge (DS20H), 22 t DM ha^{-1} mulch + pig slurry (PS20H),

Mixtures low C:N ratio: 22 t DM ha^{-1} mulch + sewage sludge (DS20L), + pig slurry (PS20L), + urea (U20L); 11 t DM ha^{-1} mulch + urea (U10L).

See Fig. 1S (supplementary material) for evolution of the overall biochemical quality of the mulch over the measuring period.

3.3. N_2O Emissions

The mean N_2O emissions (day 0 - day 192) tended to be higher for all N-treatments compared to controls, except for PS20H. The highest mean emissions were measured with the mixture of pig slurry and straw at low C:N (PS20L) and were significantly different (p-value < 0.05) from all other treatments (Fig. 4). Remarkably, for this treatment the mean N_2O emission at 21 ±1 g N_2O-N ha^{-1} h^{-1}, was about 30 times higher than for the two other N treatment with sewage sludge (DS20L) and urea (U20L). For a given mixture, a lower C:N ratio (i.e. higher N content) tended to enhance emissions, with DS20L > DS20H and PS20L >> PS20H (p-value < 0.05), as expected. The mulch quantity did not affect mean N_2O emissions neither for control (C10 vs. C20) nor urea-N treatments (U10L vs U20L).

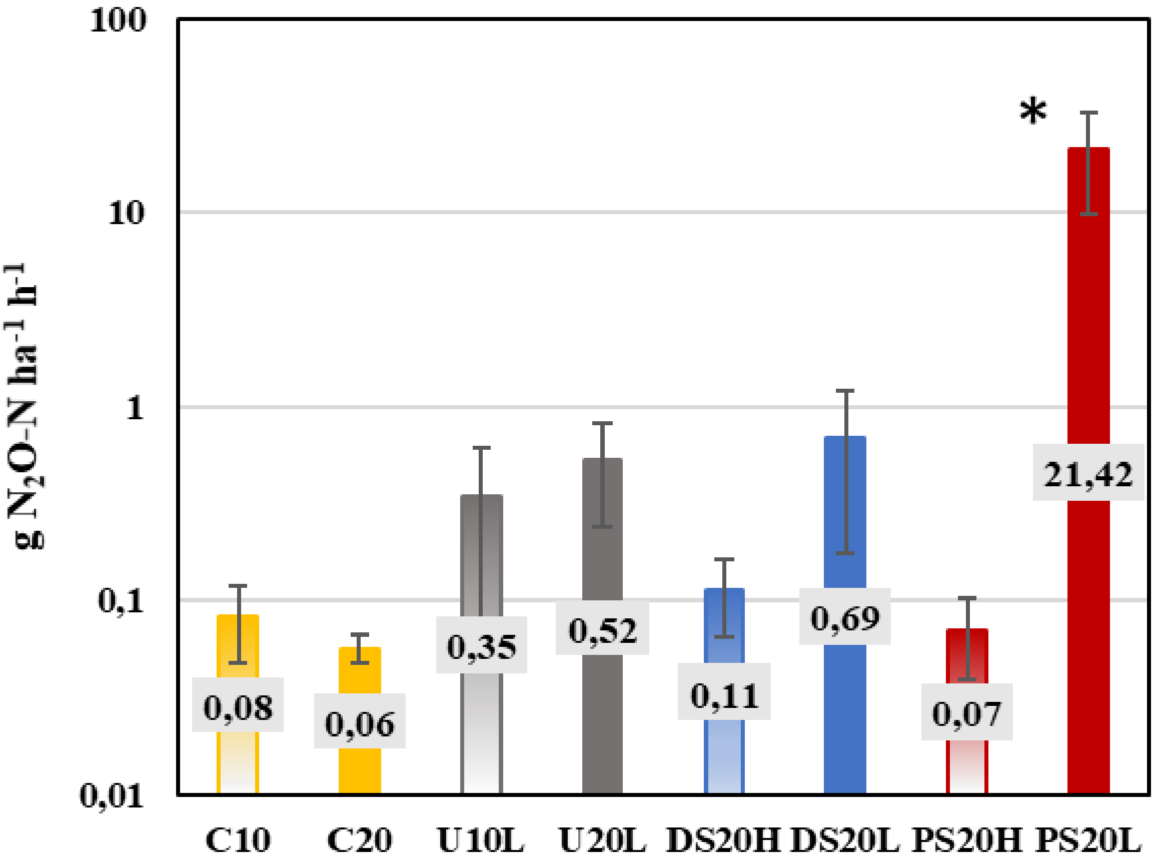

Figure 4 : Mean N_2O-N fluxes over the period day 0 -day 192 following N applications. Bars in graph indicate standard error of the mean (n=3). Controls: 11 t DM ha-1 (C10) and 22 t DM ha-1 of sugarcane mulch (C20). Mixtures high C:N ratio: 22 t DM ha-1 mulch +sewage sludge (DS20H), 22 t DM ha-1 mulch + pig slurry (PS20H). Mixtures low C:N ratio: 22 t DM ha-1 mulch + sewage sludge (DS20L), + pig slurry (PS20L), + urea (U20L); 11 t DM ha-1 mulch + urea (U10L)

3.3.1. *Effect of Mulch Quantity*

The quantity of mulch had no significant effect on the dynamics of N_2O emissions, neither for urea-N treatments nor for controls. A peak emission was measured on day 23 for urea-N treatments, which coincided with a higher mineral N present in soil (Fig. 5a). At that time, despite a fivefold higher soil mineral N measured for U10L (517 ± 4.5 mg N_{min} kg^{-1} soil), compared to U20L (97 ± 5.3 mg N_{min} kg^{-1} soil), the N_2O emissions were not significantly different ($1.7\ 10^{-3}$ to $2.1\ 10^{-3}$ mg N-N_2O kg^{-1} soil). The soil moisture was stable for the urea treatments (Fig. 5b). On the other hand, a dry episode was observed twice for controls, especially on day 122 for the lower quantity of mulch (C10), compared to the higher quantity of mulch C20.

3.3.2. *Effect of the C:N Ratio of the Mixtures*

The C:N of the mixtures did not influence the dynamics of N_2O emissions for DS20L and DS20H treatments (Fig. 5a). For pig slurry mixtures, the highest N_2O emissions were observed for PS20L on days 7 and 9, and for PS20H on day 43. The dynamics of soil mineral N matched the dynamics of N_2O emissions, and this relation was measured for all organic-N treatments, except for the PS20L, which responded more to soil moisture dynamics. Yet, the peak N_2O emission for PS20H treatment and for DS20H treatment, were measured during soil rewetting episodes, on day 43 (Fig. 5b).

3.3.3. *Effect of the Type of Fertilizer-N*

The dynamics of the N_2O emission differed between DS20L and PS20L, while U20L was similar to DS20L. For PS20L the dynamics was characterized by peak emissions within the first 9 days after input. As for DS20L and U20L the highest emissions were measured after 23 days following the input. For those treatments, the soil mineral N content dynamics (Fig. 5a) and the soil water content dynamics (Fig. 5b) matched the N_2O emission dynamics and were also different compared to PS20L. Adding pig slurry (PS20L), has led to high soil moisture just after the beginning of the experiment, which was not observed for the mixtures with solid N fertilizer treatments (DS20L and U20L).

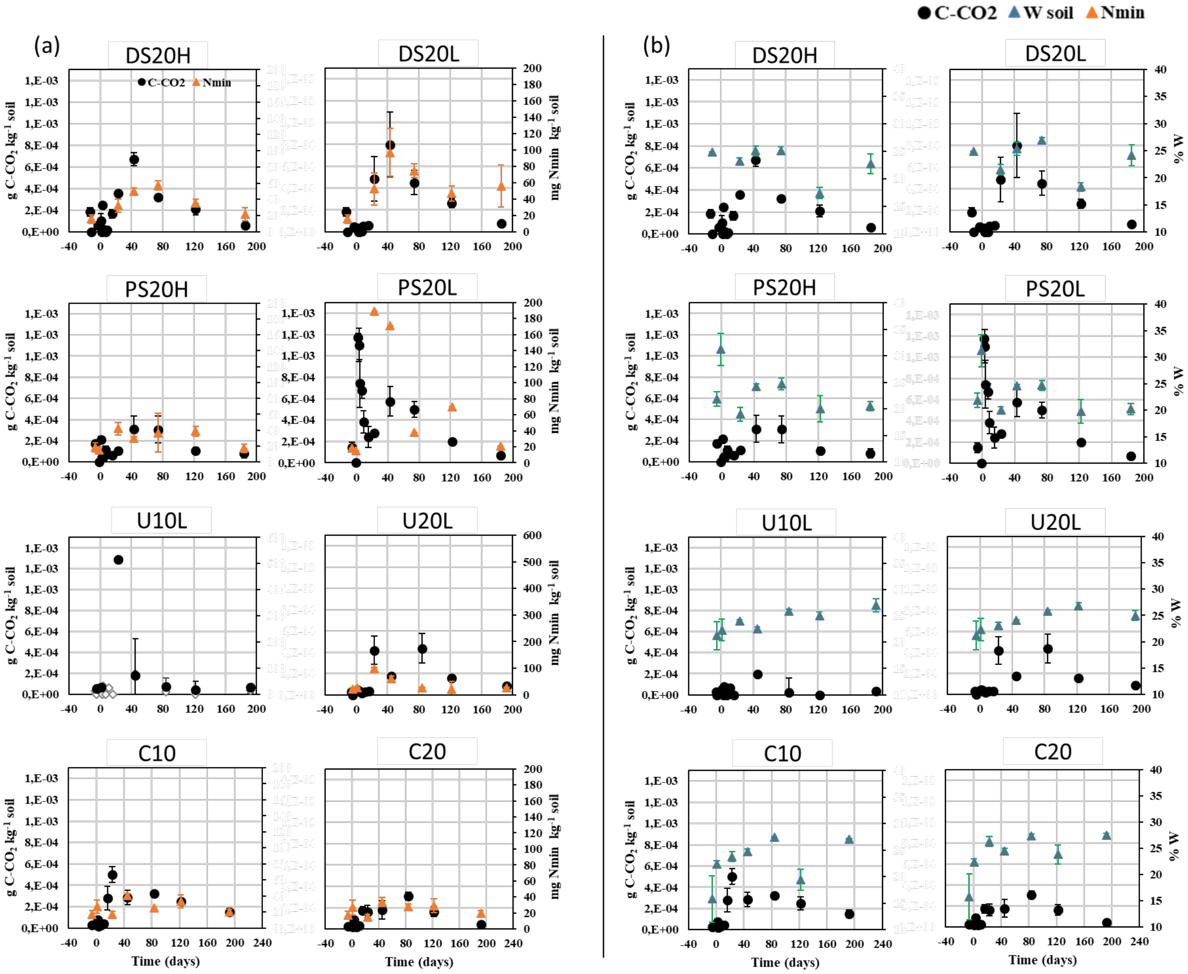

Figure 5 : Dynamics of N$_2$O-N fluxes with mineral N content dynamics of the 0-10 cm top layer soil (a) or with soil water content dynamics (b) over the measuring period between day 0 and day 192 following N applications, for the control with 11 t DM ha^{-1} of sugarcane mulch (C10) or with 22 t DM ha^{-1} of sugarcane mulch (C20), the mixtures with high C to N ratio and 22 t DM ha^{-1} of sugarcane mulch with sewage sludge (DS20H), pig slurry (PS20H), and for the mixtures with low C to N ratio and 22 t DM ha^{-1} of sugarcane mulch with sewage sludge (DS20L), pig slurry (PS20L), urea (U20L), or 11 t DM ha^{-1} of sugarcane mulch (U10L) Bars in graph indicate standard error of the mean (n=3)

3.4. CO_2 Emissions

The cumulative C-CO_2 emissions for all the treatment with N-fertilizer were higher than for control except for U10L from day 45 to 192 (Fig. 6). The C-CO_2 emissions decreased over time (0 - 45 day period > 45 - 192 day period) except for U20L. PS20L emitted the highest CO_2 quantities compared to all other treatments, especially in the first two weeks after addition. The CO_2 emissions were sensitive to both soil water content and soil mineral N content for all treatments including the controls (cf. supplementary material, Fig.2S).

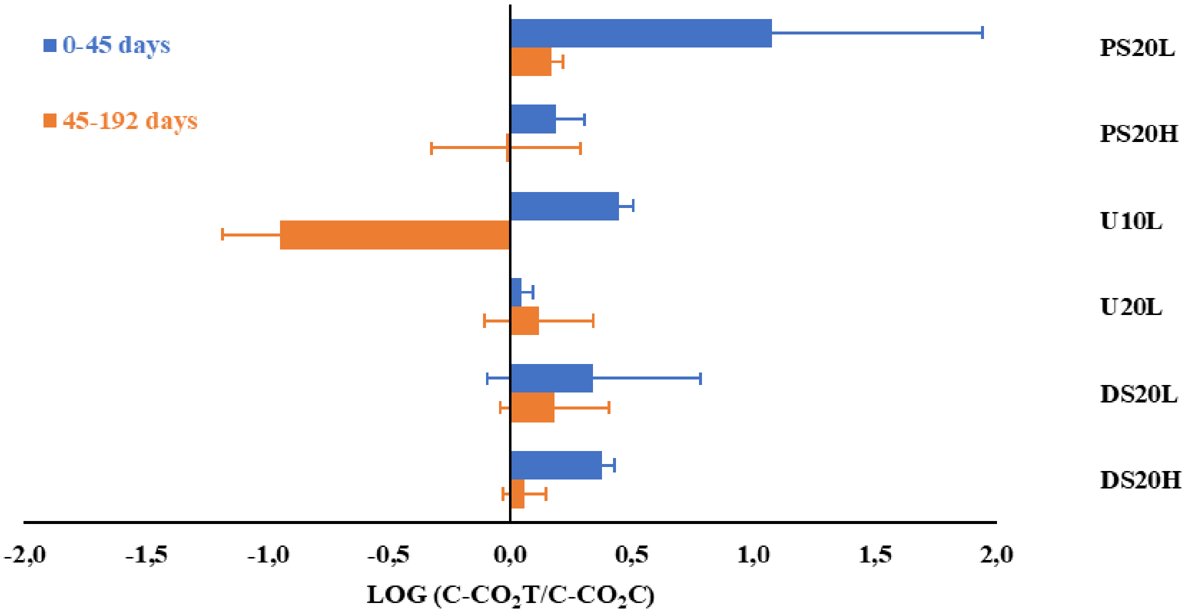

Figure 6 : Magnitude (M) of the CO_2-C summed fluxes towards control CO_2-C summed fluxes for the control with 11 t DM ha^{-1} of sugarcane mulch (C10) or with 22 t DM ha^{-1} of sugarcane mulch (C20), the mixtures with high C to N ratio and 22 t DM ha^{-1} of sugarcane mulch with sewage sludge (DS20H), pig slurry (PS20H), and for the mixtures with low C to N ratio and 22 t DM ha^{-1} of sugarcane mulch with sewage sludge (DS20L), pig slurry (PS20L), urea (U20L), or 11 t DM ha^{-1} of sugarcane mulch (U10L) Bars in graph indicate standard error of the mean (n=3)

4. Discussion

4.1. Effect of the Type of Fertilizer-N

At the end of the experiment (day 344), the remaining mulch-C was low (about 10% added straw-C) and similar for all treatments, which indicates that overall there was no water nor N limitations considering mulch decomposition. On the contrary, the kinetics of mulch-C decomposition and its C:N, were different according to treatments, particularly between organic fertilizers and urea, or control treatments. This difference between DS20L and U20L means that N accessibility for mulch decomposers was higher for organic-N fertilizers than for urea. This is not surprising since there were major differences in the physical and chemical (N) forms between pig slurry, sewage sludge and urea. Pig slurry had only 6.5 % DM and thus was pasty. This facilitated both the diffusion to the soil and the adherence on the mulch particles compared to the sewage sludge which was solid (91.3 % DM). This created probably a proximity between straw C and pig slurry N, optimizing decomposition rates. Thus, the higher decomposition rate for PS20L treatment, which resulted in the lowest level of remaining mulch-C until day 180 compared to all treatments, has led to a higher soil temperature measured for this treatment afterwards (cf. supplementary material Fig. 3S). This result matches the assumption that organic mulches are useful in reducing the maximal diurnal temperature in tropical agroecosystems (Awan 1964). Furthermore, pig slurry was rich in ammonia (57.1% of total N) compared to sewage sludge (8.9 % of total N), which might be more available for microbial assimilation during decomposition.

On the contrary, adding urea did not differ from not adding N at all such as in control straw treatments, which was a surprising result, especially because we observed an evolution of the C:N of remaining mulch, different between U20L and C20. Conversely, for urea-N +mulch (U20L) the mulch C-decomposition rate was lower, and similar to no N addition (C20). This result could be explained by the fact that urea is rapidly dissolved and infiltrated within the soil after irrigation input, and thus distanced from mulch (Pinheiro et al. 2018). Still, mulch N concentration rose during the first 4 to 6 months for all N-fertilized treatments, i.e., mulch C:N dropped. This indicates that while mulch-C and –N were jointly released by mineralization during decomposition, N mineralized was recycled (assimilated) by the decomposing microorganisms adhering to the mulch, to meet their nutrient needs. This decrease in C:N during decomposition (relative increase in N content) is generally observed during mulch decomposition (e.g. Dietrich et al. 2017, 2019). This trend was not confirmed for the C to N kinetics for C20 which was different and first rose, before dropping. As a result, in the control treatments one could hypothesize that there was higher mulch-N consumption during the first 24 days, which is possible in case of microbial population shift. In fact, many studies have shown correlation between the structure of the microbial community and the substrate quality and C:N ratio (Lozupone and Knight 2005, Fierer and Jackson 2006, Nemergut et al. 2010, Liu et al. 2016). Overall, N was not a limiting factor for the mulch-C decomposition for all the treatments as ultimately the remaining mulch-C was very low with no differences one year after deposition. The biochemical quality of the remaining mulch particles was very different on day 344 compared to the initial mulch quality. The lignin-like fraction was larger for all the N-treatments, compared to controls at similar % remaining C. This suggests a more effective use of lignin-like compounds as a resource for soil microorganisms and supports the hypothesis for microbial shift, when N is added, although additional data should be necessary to confirm these results.

The GHG emission fluxes responded mainly to mineral N content of the soil and to soil moisture. N_2O and CO_2 emissions tended to be higher with mixtures of straw and N-fertilizers compared to controls, but a significant difference was observed only with PS20L, which exhibited the highest emission rates. This was not surprising since soil mineral N content for PS20L was 4 and 2-fold higher on day 23, compared to DS20L and U20L, respectively. In addition, the soil moisture for PS20L rose significantly during the first week after application, when the measured N-N_2O fluxes were also the highest for this treatment. The soil moisture is a major driver for N_2O emissions, as high soil water content and the high content of dissolved organic C in pig slurry leads to an overconsumption of O_2 (Snyder et al. 2009). This creates temporary anoxic conditions, that does not lead only to an end product of the denitrification, N_2, but also to N_2O fluxes from microsites lacking oxygen supply (Parkin 1987, Charles et al. 2017). The soil moisture drives also the infiltration for surface applied fertilizers and the diffusion of N within the soil (Garnier et al. 2008, Babey et al. 2017), which is particularly important for dry fertilizers such as urea and solid sewage sludge. Thus, the kinetics of N-N_2O fluxes responded to the physical characteristics of the fertilizers-N, as pig slurry highest denitrification rates were measured just after input, compared to urea and sewage sludge that depended on irrigation and rainfall for N infiltration and diffusion within the soil. Therefore, the form of N applied as fertilizer, and particularly the physical characteristics influenced the rate of decomposition of the mulch and the N_2O emitted on the short-term, with faster decomposition and higher N_2O emitted with pig slurry compared to solid sewage sludge and urea.

4.2. Effect of the C to N Ratio of the Mixtures

The C to N ratio of the mixtures have been modified by changing the ratio straw:fertilizer, to uncorrelate the effect of the availability of N from the form of added N. Pig slurry and sewage sludge are organic fertilizers, and thus also contain C, that might be used for microbial growth. This could explain the lower decomposition rates during the first 25 days for PS20L and DS20L, which contain more C added with pig slurry compared to PS20H and DS20H, respectively. However, adding 10 times more N in PS20L and DS20L compared to PS20H and DS20H, led to higher decomposition rates thereafter, until day 190 for PS20L and day 120 for DS20L. Adding more N to the mixtures enhanced mulch-C microbial uptake and impacted also the relative mulch-N concentration, which rose for PS20L and DS20L (C to N ratio drop), compared to PS20H and DS20H. However, at the end of the experiment, the remaining mulch-C was not different between DS20H and DS20L while it tended to be higher for PS20H compared to PS20L. Therefore, the mulch-C decomposition kinetics depends not only on N availability, but also on other factors such as the physical and the biochemical characteristics of fertilizer (Chen et al. 2014b) and could be only partially explained by the initial C to N ratio of the mixture (total N input), as it was previously shown by other studies (Fang et al. 2007, Caihong et al. 2013).

The highest CO_2 and N_2O emission rates were observed for mixtures with lower initial C to N ratio which was an expected result (Aguilera et al. 2013, Charles et al. 2017), as both microbial degradation and N_2O emission can be enhanced by the enhanced availability of mineral N to soil microorganisms. The highest N-N_2O emission was measured with PS20L, which was significantly higher than all other treatments having similar initial mixture C to N ratios. The reason for this might be the high carbonate content in pig slurry, which as a labile C, is highly available for microorganisms that consume O_2 (Velthof et al. 2003, Oenema et al. 2008). The liquid form of pig slurry induces higher anoxic conditions within soil microsites, that favor denitrification, which was observed with vinasse (liquid fertilizer) in a sugarcane cropping systems in Brazil (Carmo et al. 2013). Therefore, C to N ratio of the straw-fertilizer mixtures is not sufficient in itself to determine N_2O emissions, and the chemical and physical characteristics of fertilizers, influencing other important factors, such as O_2 concentration or location of microbial growth and nutrients can influence the emissions.

4.3. Effect of Initial Mulch Quantity

The remaining mulch C after 344 days represented 4 to 12 % added mulch C, which was a range reported for other sugarcane systems (Fortes et al. 2012, Carvalho et al. 2017). However, the mulch initial quantity (22 and 11 t DM ha^{-1}) did not influence the relative decomposition rate, which is in line with Dietrich et al. (2017) and Dietrich et al. (2019). This means that for twice more sugarcane mulch left on the soil, the remaining amount of mulch C was twice at the end. The initial mulch quantity influenced the temperature of the top soil layer (0 - 7.5 cm) as shown previously (Olasantan 1999, Jun et al. 2014, Wang et al. 2019), with higher temperature measured with lower mulch quantity (see. supplementary material Fig 3S). For all treatments the mean soil moisture was comprised between 20% and 30% WHC, with a remarkable dry episode around day 120. The mulch quantity influenced the drying and rewetting dynamics, with faster soil drying in C10, compared to C20, which is in line with other studies (Ball-Coelho et al. 1993, Cerri et al. 2011, Azevedo et al. 2014). This trend was also noticed for U10L and U20L, but to a lesser extent. To understand this result, one should add that, sugarcane growth between day 24 and day 180 was significantly higher in urea fertilized plots than in control plots (personal observation), modifying the micro-climate at the soil-atmosphere interface. Fertilized plots such as U10L and U20L had denser canopy and thus less solar radiation reaches soil surface.

Despite these differences, the mulch quantity (11 and 22 t DM ha^{-1}) had no significant effect on GHG emission, as the dynamics and mean fluxes of N_2O and CO_2 emissions were similar for U10L and U20L, and for C10 and C20. Mulching was reported to enhance N_2O emissions especially compared to no mulching (Cerri et al. 2009, Fracetto et al. 2017), but considering mulch quantity, there are few studies that reported an effect (Pinheiro et al. 2019). Thus, Carmo et al. (2013) found that leaving more mulch promote more N_2O emission, when a liquid fertilizer was added (sugarcane vinasse), and underlines the importance of the physical state of fertilizer. Therefore, in our experimental conditions, the higher C input to soil with higher initial mass of sugarcane mulch left on the soil, was not counterbalanced by an increase emission of N_2O with higher mulch mass, at least with no fertilizer (control) or urea applied.

5. Conclusion

Promoting the use of organic fertilizers such as pig slurry and sewage sludge for substitute to mineral fertilizers in sugarcane production are reliable options in circular economy perspective.

Nevertheless, the undergoing environmental impact of adding organic fertilizers in mulch cropping systems is an important question to be addressed. Our results showed that in our pedoclimatic conditions, the mulch decomposition rate was proportional to the initial mulch quantity, which means that leaving more straw on the soil surface allows more carbon to be recycled to the soil. This amount of C can be fairly predicted from the initial amount left and the degradation constant, and this is important to consider for trash management. The combined recycling of organic fertilizer and sugarcane straw affected the dynamics of decomposition, with faster decomposition observed when pig slurry was applied and when high rate of organic fertilizer was used, highlighting the importance of physical form of organic waste and the accelerating role of nitrogen on residue decomposition. While it no longer had any effect on the residual mulch after one year, this combined effect of straw-C and fertilizer-N application had transient consequences on soil and microclimatic properties, and therefore on N losses and crop nutrition. Particularly the higher N_2O emission observed on the short-term with the liquid-N fertilizer (pig slurry) at high N-rate application might question the global greenhouse gas balance, as N_2O emissions can reduce or even negate the positive effect of additional carbon storage in soil. This work underscores the need of jointly studying carbon and nitrogen dynamics and considering both soil carbon and gas emissions to assess the greenhouse gas balance of sugarcane farming practices and fossil-C substitutions.

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Supplementary material

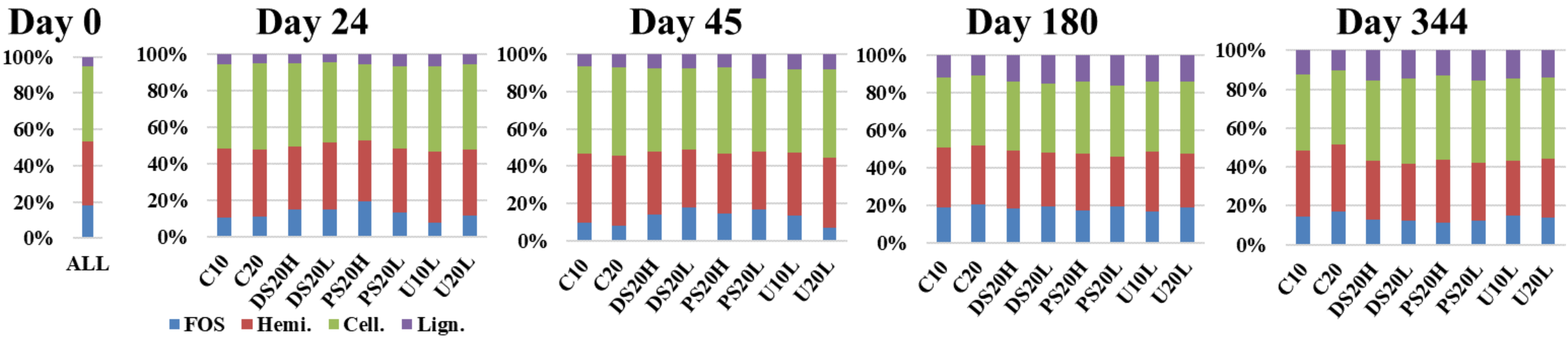

Figure 1S : Evolution of the biochemical quality of the mulch over the measuring period between day 0 and day 192 following N applications, for the control with 11 t DM ha^{-1} of sugarcane mulch (C10) or with 22 t DM ha^{-1} of sugarcane mulch (C20), the mixtures with high C to N ratio and 22 t DM ha^{-1} of sugarcane mulch with sewage sludge (DS20H), pig slurry (PS20H), and for the mixtures with low C to N ratio and 22 t DM ha^{-1} of sugarcane mulch with sewage sludge (DS20L), pig slurry (PS20L), urea (U20L), or 11 t DM ha^{-1} of sugarcane mulch (U10L) Bars in graph indicate standard error of the mean (n=3)

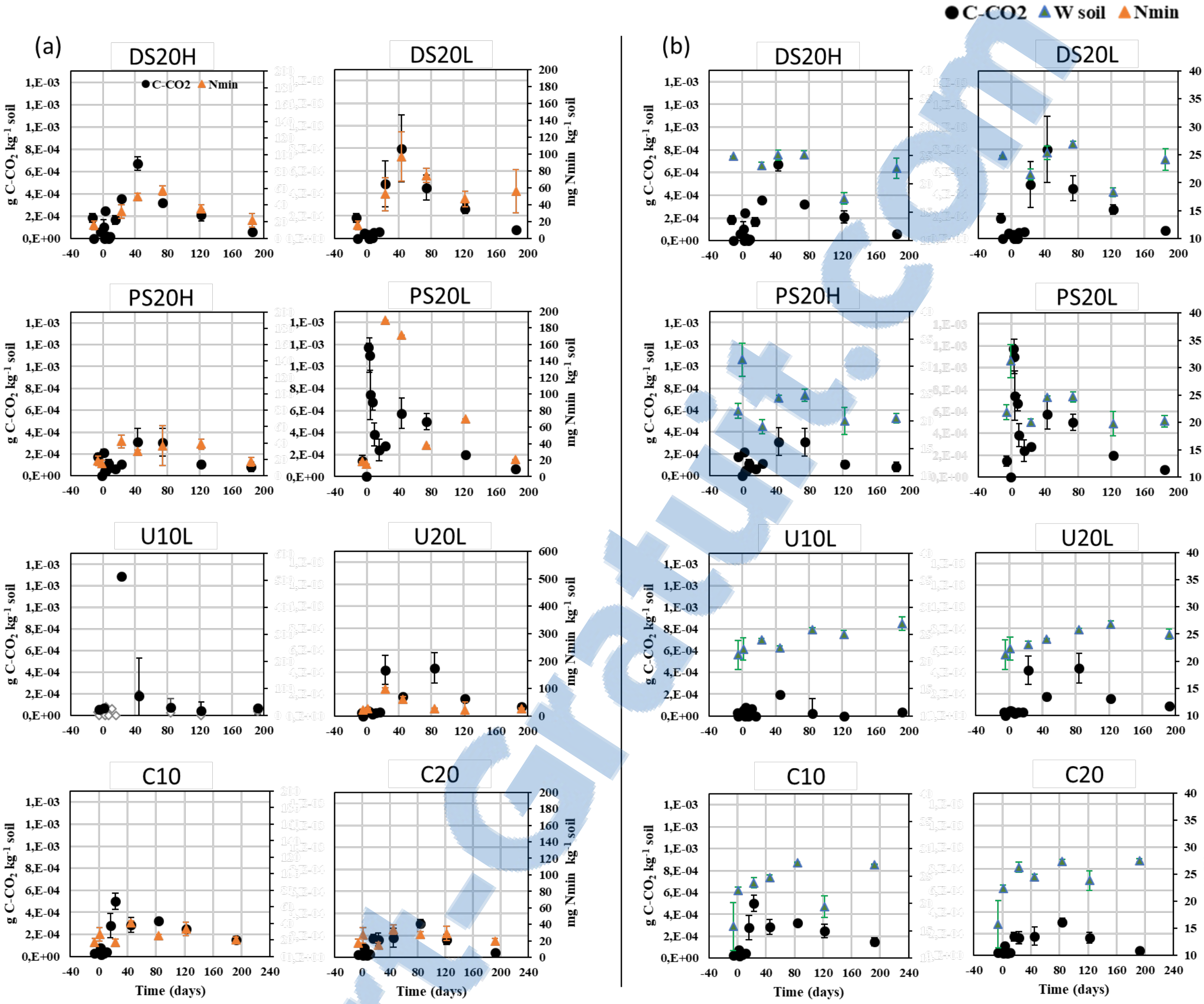

Figure 2S : Dynamics of CO_2-C fluxes with mineral N content dynamics of the 0-10 top layer soil (a) or with soil water content dynamics (b) over the measuring period between day 0 and day 192 following N applications, for the control with 11 t DM ha^{-1} of sugarcane mulch (C10) or with 22 t DM ha^{-1} of sugarcane mulch (C20), the mixtures with high C to N ratio and 22 t DM ha^{-1} of sugarcane mulch with sewage sludge (DS20H), pig slurry (PS20H), and for the mixtures with low C to N ratio and 22 t DM ha^{-1} of sugarcane mulch with sewage sludge (DS20L), pig slurry (PS20L), urea (U20L), or 11 t DM ha^{-1} of sugarcane mulch (U10L) Bars in graph indicate standard error of the mean (n=3)

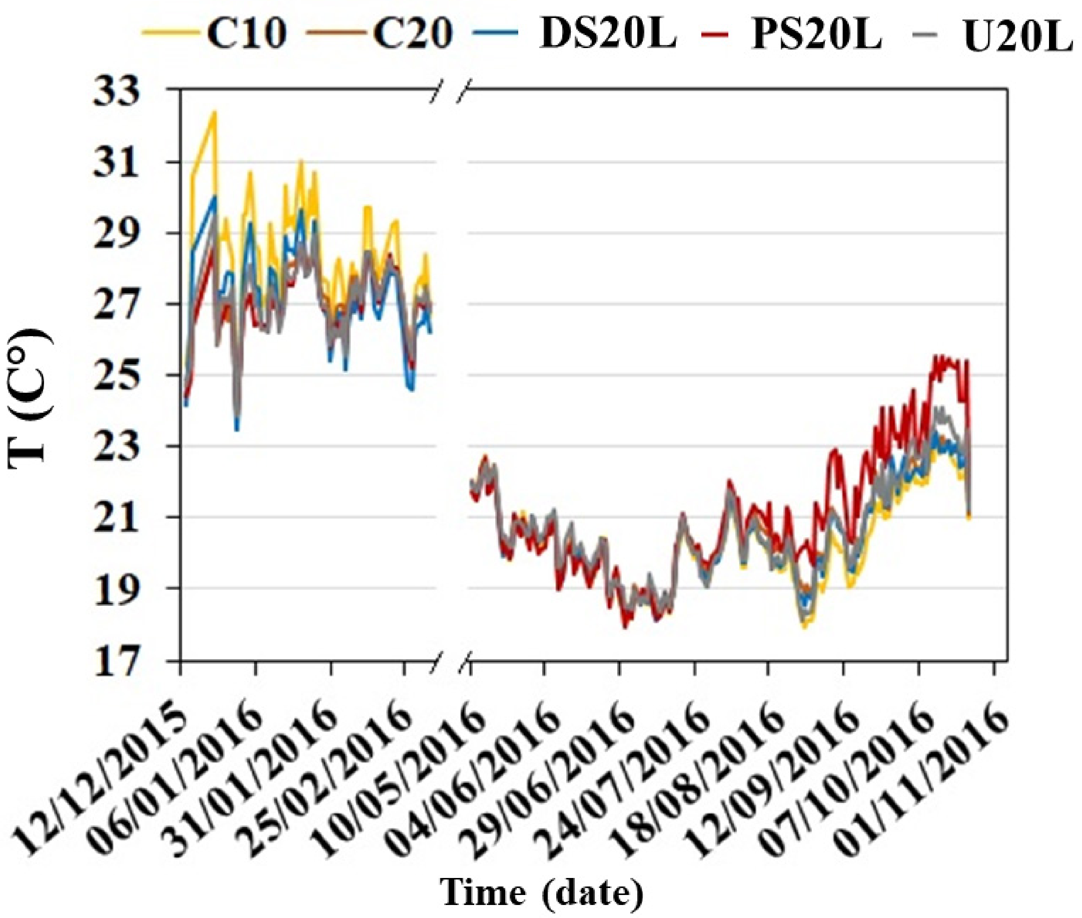

Figure 3S : Temperature of the soil at 7.5 cm under the control treatments with 11 t DM ha^{-1} of sugarcane mulch (C10) and 22 t DM ha^{-1} of sugarcane mulch (C20), and the mixtures with low C to N ratio and 22 t DM ha^{-1} of sugarcane mulch with sewage sludge (DS20L), pig slurry (PS20L) and urea (U20L).

CHAPITRE 4

Conclusion générale

Liste des figures

V. Conclusion générale

L'objectif général de la thèse est l'évaluation des intérêts agronomique et environnemental du recyclage conjoint de paillis de résidus de canne à sucre et de produits résiduaires organiques (PRO) de nature physicochimique différentes, sur les sols agricoles en culture à la Réunion. On s'est proposé de quantifier et de modéliser l'impact du système paillis-PRO sur les dynamiques du C et du N, et sur les émissions de gaz à effet de serre d'un sol tropical cultivé en canne à sucre. Concrètement la question générale que l'on s'est posée peut être formulée de la manière simple suivante :

Est-ce que la substitution de l'engrais azoté de synthèse par des fertilisants organiques recyclés a un impact sur la disponibilité de l'azote, sur le stockage du carbone et sur les émissions de GES ?

Plus spécifiquement elle s'articule par la suite en quelques sous-questions auxquelles nous avons apporté des réponses par les expérimentations réalisées :

Quelle sont les différences entre :

les additions d'N minéral ou organique ?

les différents types de fertilisants organiques recyclés ?

les différentes quantités de résidus de culture?

les différents C:N des mélanges paille/fertilisants ?

1. Approche méthodologique

La littérature spécifique souligne l'importance des propriétés physiques et biochimiques des matières organiques en mélange, comme facteurs majeurs d'interactions pour les dynamiques de transformation (Aerts 1997, Pérez-Harguindeguy et al. 2008, Meier and Bowman 2010, Berglund and Ågren 2012, de Urzedo et al. 2013). Pour mieux comprendre ces dynamiques, nous nous sommes d'abord intéressés aux cinétiques de minéralisation du carbone et de l'azote des apports en tenant compte de la diversité de leurs propriétés physicochimiques. L'utilisation d'un modèle additif, simple (Kyulavski et al. 2019b) et un modèle mécaniste de transformation – CANTIS (Garnier et al. 2003, Chalhoub et al. 2013) nous a permis d'évaluer la capacité prédictive de ces modèles pour les dynamiques de transformation de la MO et de cerner les interactions qui influencent ces dynamiques lorsque l'on apporte un mélange de paille de canne à sucre et de fertilisants azotés. Nous avons réalisé des incubations en conditions contrôlées en incorporant de la paille de canne à sucre et des fertilisants azotés, seuls ou en mélange, à un sol agricole classiquement cultivé en canne à sucre à la Réunion. Cette méthode permet de standardiser/maîtriser les facteurs externes comme le climat, les pratiques agricoles, le type de sol ou encore le type de couvert végétal, qui, en conditions « de terrain », influencent les dynamiques de transformation (Adu and Oades 1978, Leirós et al. 1999, Whalen and Sampedro 2010, Dequiedt et al. 2011). Elle devait permettre de s'affranchir des variations de ces facteurs non pris en compte par les modèles choisis. Un des défis majeurs était de prédire correctement les dynamiques de minéralisation de divers mélanges, à partir des données de minéralisation de chaque matière organique incubée individuellement.

Pour autant, les conditions de laboratoire ne traduisent qu'un potentiel en termes de dynamique de transformation et les résultats obtenus ne sont pas directement transposables au champ. Les limites de cette méthode sont bien connues (Linlin et al. 2015). Nous avons mené également des expérimentations au champ pour être en conditions réelles de culture de canne à sucre, et pour répondre aux questions pratiques/opérationnelles suivantes : *quel est l'intérêt agronomique de ces pratiques de fertilisation et quel impact environnemental ont-elles en termes d'émissions de GES ?* Deux campagnes expérimentales indépendantes sur deux années de culture de canne à sucre, dans des conditions pédoclimatiques similaires ont été réalisées. Les designs expérimentaux ont été pensés avec la volonté d'assurer d'une part la possibilité de comparer les résultats obtenus les deux années et d'autre part d'élargir le gradient et le nombre de variables mesurées pour affiner l'évaluation de l'impact du système paillis-PRO sur les dynamiques du C et du N, et sur les émissions de gaz à effet de serre. La première année correspondait à l'installation de la culture de la canne à sucre récoltée après un cycle court de 9 mois laissant des résidus au sol en moindre quantité et plus riches en N. La seconde année, correspondant à la 1ère repousse de canne à sucre, était donc différente en termes de quantité (davantage de paille) et qualité (teneur en N plus faible) du mulch. De plus, les limites expérimentales relevées lors de la première campagne nous ont permis de faire évoluer l'expérimentation menée la deuxième année, où nous avons réalisé des mesures de paramètres physiques et chimiques supplémentaires en couvrant un cycle cultural entier.

2. Synthèse des résultats

Pour exprimer de façon synthétique les résultats, nous avons comparé les traitements aux champs dans les quatre tableaux suivants.

Pour cela, les sommes des données brutes mesurées en kg ha^{-1} ont été utilisées. Pour comparer les effets des fertilisants et de la paille sur les stocks de C et sur l'azote disponible (N minéral du sol), des indices, qui tiennent compte des quantités de C et N totaux apportés, ont été utilisés.

Notamment, le potentiel de stockage de carbone potentiel (à court terme) a été représenté par la différence de la quantité apportée et la somme des données brutes d'émission sur les 49 jours après épandage, rapportée à la somme des données brutes d'émission sur les 49 jours après épandage :

$$Potentiel\ de\ stockage\ de\ C = \frac{C_{app} - \sum_{j=0}^{49} CO_2 - C}{\sum_{j=0}^{49} CO_2 - C}$$

Où, C_{app} est la somme du C apporté par la paille et le fertilisant en kg ha^{-1} et les émissions de CO_2-C en kg ha^{-1} ont été cumulées sur la période de jour 0 à jour 49 après le début de l'expérimentation. Plus cet indice est élevé, plus la quantité de carbone potentiellement stockée dans le sol est grande.

Potentiel d'émission de N_2O a été calculé comme la réciproque du potentiel d'immobilisation de l'azote :

$$Potentiel\ d'émission\ N_2O = 1 \Big/ \frac{N_{app} - \sum_{j=0}^{49} N_2O - N}{\sum_{j=0}^{49} N_2O - N}$$

Où N_{app} est la somme du N apporté par la paille et le fertilisant en kg ha^{-1} et les émissions de N_2O-N en kg ha^{-1} ont été cumulées sur la période de jour 0 à jour 49 après le début de l'expérimentation. Plus cet indice est élevé, plus le potentiel d'émission de N_2O est grand.

Le N disponible a été représenté par la somme du N minéral du sol :

$$N\ disponible = \sum_{j=0}^{49} N_{min}$$

Où, N_{min} est la somme de N minéral en mg kg^{-1} sol (NO_3^- et NH_4^+), pendant la période de jour 0 à jour 49 après le début de l'expérimentation.

<table>
<tr><td rowspan="4">Hypothèse</td><td colspan="4"><i>N fertilisant organique vs N fertilisant de synthèse</i></td></tr>
<tr><td>CO_2 ↗</td><td>Stockage C ↗</td><td>N_2O ↘</td><td>N disponible ↘</td></tr>
<tr><td colspan="4">Selon la littérature
Les émissions de CO_2 seraient plus importantes pour les sols recevant des fertilisants organiques, dû à une quantité plus grande de C apporté par ces derniers (Carmo et al. 2013).
Les potentiels de stockage de carbone dans le sol seraient plus importants avec les fertilisants organiques, dû à la quantité plus grande de C apporté par ces derniers et à la disponibilité de l’azote plus faible des fertilisants organiques (Wardle 1992, Diacono and Montemurro 2010).
Les potentiels d’émissions de N_2O seraient plus importants avec le fertilisant de synthèse, car le design expérimental adopté implique un apport d’eau supplémentaire qui favorise la dissolution rapide et une disponibilité de l’azote plus grande, plus rapidement avec l’urée (Ball et al. 2004, Meijide et al. 2007)
Le N disponible serait moins grand dans le sol avec les fertilisants organiques dû à la stabilité chimique et la protection physique de l’azote présent sous forme organique (Clark et al. 1998, Poudel et al. 2002, Herencia et al. 2007)</td></tr>
<tr></tr>
<tr><td rowspan="4">Résultats</td><td colspan="4"><i>N fertilisant organique vs N fertilisant de synthèse</i></td></tr>
<tr><td>CO_2 ↗</td><td>Stockage C</td><td>N_2O</td><td>N disponible ↘</td></tr>
<tr><td>6,3 vs 2,5
kg ha^{-1} h^{-1}</td><td colspan="3">Dépend de la nature des fertilisants organiques</td></tr>
<tr><td colspan="4">Les émissions de CO_2 sont plus importantes avec les fertilisants organiques car ils apportent davantage de C dont une partie est facilement biodégradable.
Les potentiels de stockage de carbone dans le sol sont plus importants avec les PRO secs, tels que la boue de STEP sèche, dû à la quantité plus grande de C organique et d’azote organique apportés, comparé au fertilisant de</td></tr>
</table>

	synthèse (urée). A contrario, l'ajout de fertilisant synthétique (urée) avec de la paille permet de stocker plus de C dans le sol que l'ajout de fertilisant organique liquide, tel que le lisier de porc avec de la paille. Les potentiels d'émissions de N_2O sont plus importants avec les fertilisants organiques liquides, tel que le lisier de porc, avec une intensité d'émission importante immédiatement après apport. Les potentiels d'émissions de N_2O plus importants avec les fertilisant de synthèse (urée) qu'avec les fertilisants organiques secs (boue de STEP) probablement dû à l'homogénéité d'apport d'eau (système irrigué et pluviométrie importante), la dissolution plus rapide et la disponibilité de l'azote plus grande avec l'urée. Le N disponible du sol est moins grand avec le fertilisant de synthèse, urée qu'avec la moyenne des deux fertilisants organiques. Ceci est uniquement le cas avec la grande quantité de paille (20 t ha-1). La modalité à petite quantité de paille + urée représente la plus grande quantité de N minéral du sol (688 mgN kg^{-1} sol). Le N minéralisable de l'urée est supérieur aux deux fertilisants organiques (données disponibles).

<table>
<tr><td rowspan="3">Hypothèses</td><td colspan="4"><i>Boue de STEP vs Lisier de porc</i></td></tr>
<tr><td>CO_2 ↘</td><td>Stockage C ↗</td><td>N_2O ↘</td><td>N disponible ↘</td></tr>
<tr><td colspan="4">Selon la littérature
Les émissions de CO_2 seraient plus importantes avec le lisier de porc, car c'est un fertilisant liquide, contenant une concentration de carbone dissous plus importante (Tejada and Gonzalez 2005, Carmo et al. 2013)
Par conséquent, le potentiel de stockage de carbone serait plus important avec la boue de STEP, car les pertes en CO_2 pour ce fertilisant seraient moins importantes que pour le lisier de porc
Le potentiel d'émissions de N_2O serait plus importantes avec le lisier de porc, car c'est un fertilisant organique riche en NH_4^+, précurseur de la nitrification dénitrifiante, qui est rapidement transformable en NO_3^- dans le sol et représente le précurseur de la dénitrification (Allen et al. 2010, Nguyen et al. 2014)
Le N disponible serait plus grand avec l'application du lisier de porc, riche en N minéral à la base (Aguilera et al. 2013)</td></tr>
<tr><td rowspan="4">Résultats</td><td colspan="4"><i>Boue de STEP vs Lisier de porc</i></td></tr>
<tr><td>CO_2 ↘</td><td>Stockage C ↗</td><td>N_2O ↘</td><td>N disponible ↘</td></tr>
<tr><td>2,5 vs 4,5
kg ha^{-1} h^{-1}</td><td>90 vs 35</td><td>2,51 10^{-4}
vs 1,17 10^{-3}</td><td>326 vs 488
mg N kg^{-1} sol</td></tr>
<tr><td colspan="4">Toutes les hypothèses ont été validées. Notre étude montre l'importance des émissions de CO_2 avec le lisier de porc dont les carbonates sont rapidement décarboxylés par voie chimique, contrairement aux émissions de N_2O, dont la cinétique suggère un mécanisme biologique contrôlé par la qualité physicochimique du lisier de porc (liquide et riche en N minéral).</td></tr>
</table>

<table>
<tr><td rowspan="3">Hypothèses</td><td colspan="4">Grande quantité de mulch vs Petite quantité de mulch</td></tr>
<tr><td>CO_2 ↗</td><td>Stockage C ↗</td><td>N_2O ↗</td><td>N disponible↘</td></tr>
<tr><td colspan="4">Selon la littérature
Les émissions de CO_2 seraient plus importantes avec une quantité de paille plus grande (Carmo et al. 2013)
Le potentiel de stockage de carbone serait plus grand lorsque la quantité de paille est plus grande, car la proportion de C stabilisé serait proportionnelle à la quantité de C récalcitrant apportée (Lou et al. 2011)
Le potentiel d'émissions de N_2O serait plus important avec une quantité de paille plus grande. Ceci serait dû à la préservation de l'humidité du sol par la paille et à la quantité de C apportée d'autant plus grande (de Oliveira et al. 2013, Wang et al. 2016b)
Le N disponible serait plus grand lorsque la quantité de paille est plus petite, l'immobilisation de N est plus importante lorsque la quantité de C est plus importante</td></tr>
<tr><td rowspan="4">Résultats</td><td colspan="4">Grande quantité de mulch vs Petite quantité de mulch</td></tr>
<tr><td>CO_2 ↗</td><td>Stockage C ↗</td><td>N_2O ↗</td><td>N disponible↘</td></tr>
<tr><td>3,3 vs 2,8
kg ha^{-1} h^{-1}</td><td>91 vs 52</td><td>$6\ 10^{-4}$ vs $3\ 10^{-4}$</td><td>205 vs 426
mg N kg^{-1} sol</td></tr>
<tr><td colspan="4">Toutes les hypothèses ont été validées. Nous avons constaté un potentiel de stockage de C proportionnel à la quantité de carbone apporté (par le mulch). Cependant, les émissions de CO_2 n'ont pas été proportionnelles à la quantité de C apporté. Cela est dû au fait que la qualité des fertilisants apportés joue un rôle les émissions de CO_2 en mélange (décarboxylation chimique pour le lisier de porc, respiration microbienne pour la boue de STEP).</td></tr>
</table>

<table>
<tr><td rowspan="3">Hypothèses</td><td colspan="4">C:N bas vs C:N élevé du mélange</td></tr>
<tr><td>CO_2 ↗</td><td>Stockage C ↘</td><td>N_2O ↗</td><td>N disponible↗</td></tr>
<tr><td colspan="4">Selon la littérature
Les émissions de CO_2 seraient plus importantes lorsque le C:N des mélanges est plus bas. Ceci a été observé pour des résidus de coupe de C:N plus bas (Gentile et al. 2011)
Le potentiel de stockage de carbone serait plus grand lorsque le C:N des mélanges est plus grand, car la cinétique de décomposition est plus lente (Aerts 1997, Chen et al. 2014a)
Le potentiel d’émission de N_2O serait plus important lorsque le C:N du mélange est plus petit, car la quantité de N apportée est plus grande dans ce cas. Les émissions sont corrélées positivement avec la quantité de N (Snyder et al. 2009)
Le N disponible serait plus grand lorsque le C:N des mélanges est plus bas. Ceci a été démontré pour des résidus de culture de C:N plus bas (Trinsoutrot et al. 2000)</td></tr>
<tr><td rowspan="4">Résultats</td><td colspan="4">C:N bas vs C:N élevé du mélange</td></tr>
<tr><td>CO_2 ↗</td><td>Stockage C ↘</td><td>N_2O ↗</td><td>N disponible↗</td></tr>
<tr><td>5,9 vs 3,7
kg C ha^{-1} h^{-1}</td><td>53 vs 95</td><td>5,4 10^{-4} vs 3,9 10^{-4}</td><td>407 vs 180
mg N kg^{-1} sol</td></tr>
<tr><td colspan="4">Toutes les hypothèses ont été validées. L’effet de l’azote en grande quantité n’a été validé ici qu’à court terme. Les stockages potentiels de carbone sont très proches et peuvent être affectés différemment à long terme avec la quantité d’azote plus grande (Berg and Matzner 2011).
Le N disponible est plus élevé lorsque le C:N du mélange est bas. Toutefois, la proportion de N minéralisable est plus faible pour les mélanges à C:N bas qu’avec le mélange à C:N élevé (données disponibles).</td></tr>
</table>

3. Discussion générale

3.1. Décomposition de la paille

Les résultats des incubations en conditions contrôlées des mélanges paille/fertilisant ont permis d’identifier une décomposition plus lente de la paille, lorsque la source d’azote était apportée par des fertilisants organiques, comparé au cas où l’azote est apporté sous forme de nitrates en solution aqueuse (K-NO_3). Ainsi, pour modéliser correctement les mélanges paille/fertilisant organique en utilisant le même jeu de paramètres que celui utilisé pour les matières organiques seules, nous avons été obligés de modifier un paramètre dans CANTIS, celui qui traduit l’accessibilité du C et du N pour la biomasse microbienne du sol – le facteur de contact (K_{MZ}), dont la valeur est inversement proportionnelle à l’accessibilité de l’azote. Pour obtenir une prédiction correcte, nous avons attribué un K_{MZ} plus fort au mélange paille/boue de STEP, qu’au mélange paille/lisier de porc, alors que pour la paille/KNO_3 le facteur de contact a été fixé à 0. Ce résultat indique que la forme physique et la qualité chimique des apports azotés sont probablement des facteurs majeurs d’interaction de décomposition en mélange dans des conditions de température et humidité constantes. Nous avons déduit que l’apport d’engrais organique solide, comme la boue de STEP, engendre une difficulté de diffusion de l’azote dans le mulch et dans la matrice du sol, ce qui mène à une localisation hétérogène des deux ressources pour les microorganismes, le C et le N. Au champ, nous avons évalué l’influence du type de mélange sur la cinétique de décomposition de la paille. À partir de ces résultats nous avons déduit le potentiel de stockage de carbone dans le sol, à court terme. Pendant la première campagne d’expérimentation au champ, qui a duré 4 mois (mesures du taux de décomposition de paille sur 120 jours et des émissions de GES sur 14 jours), la proportion de paille décomposée n’a été affectée ni par la quantité de paille laissée au sol, ni par le type de fertilisant apporté. Les résultats obtenus lors de la deuxième campagne, qui a duré 11 mois (mesures du taux de décomposition de paille sur 340 jours et des émissions de GES sur 49 jours) ont confirmé ceux obtenus lors de la première, c.à.d. que la quantité de paille décomposée est proportionnelle à la quantité initiale laissée, ce qui est en adéquation avec Dietrich et al. (2017). D’autre part, si on considère l’efficience de stockage de carbone indépendante de la quantité de carbone apporté (Liang et al. 2016), cela implique également qu’une quantité de paillis plus importante, augmente proportionnellement le stock de carbone dans le sol, ce qui a été observé dans des méta-analyses sur les résidus de culture (Liu et al. 2014) et sur les fertilisants organiques (Maillard and Angers 2014).

Les types de fertilisants des deux campagnes d'expérimentation ont été identiques (urée, lisier de porc, boue de STEP). Une différence notable entre les deux campagnes concerne la qualité du lisier de porc qui, lors de la deuxième campagne, contenait 6% de masse sèche, contrairement à la première campagne, où il était très liquide (0.8% de masse sèche). La cinétique de décomposition a été influencée par le type de fertilisant et particulièrement par la qualité des fertilisants. Lors de la première campagne la cinétique de décomposition de la paille est caractérisée par une perte de C plus rapide sur le premier mois, suivant l'application de boue de STEP. Lors de la deuxième campagne au champ la perte de C de la paille est plus rapide après ajout de lisier de porc en grande quantité (PS20L). Ce résultat montre, comme lors des incubations, que les qualités chimiques et physiques des fertilisants interagissent avec la paille et sont des déterminants des dynamiques de décomposition. Même s'il parait difficile d'identifier les mécanismes impliqués, nous pouvons proposer deux hypothèses. Par exemple :

les *pellets* (en français « granulés bouchons ») de boue séchée tombent à la surface du sol et lors des épisodes d'apport d'eau, s'imbibent et grossissent avant de se désagréger. Or, la capacité de rétention d'eau de la boue de STEP est importante, ce qui est dû à sa siccité importante (91 % de MS). Ceci pourrait créer des conditions d'humidité qui accélèrent la décomposition de la paille à court terme ; nous avons constaté une adhérence du lisier de porc riche en MS sur le paillis. On peut imaginer une proximité spatiale des deux ressources de C et de N accrue, ce qui favorise la décomposition par accessibilité de l'azote accrue (cf. paragraphe précédent).

Pendant la deuxième campagne, nous avons testé l'effet de l'apport de N 10 fois supérieures aux quantités « conventionnelles », sur certaines caractéristiques du sol (0-10cm) et du paillis après 350 jours de décomposition. L'agrégation de l'ensemble des données dans une analyse en composantes principales (Fig.1) n'a cependant pas permis d'opposer clairement les traitements avec un apport fort en N (traitements contenant la lettre « L », avec rapport C:N du mélange « low »). De même, aucune distinction n'a pu être faite par rapport à la quantité de paille apportée. En revanche, une opposition marquée a été observée pour la qualité biochimique du paillis en mélange avec des fertilisants organiques et le paillis en mélange avec l'urée, ou sans apport en azote. Dans la figure 1, l'axe 1, qui explique près de 60 % de la variabilité (l'inertie), oppose clairement la proportion forte en cendres, et en moindre mesure la matière organique soluble à la proportion forte en lignine, en cellulose et en C total. D'autre part à gauche, la projection des traitements (objets), indique clairement que la paille des mélanges avec lisier de porc et boue de STEP contiennent les proportions de lignine et de cellulose les plus grandes. L’application de PRO a aussi un effet sur la chimie de la surface du sol, car sous les mélanges paille/PRO le sol contient plus de NO_3^- et NH_4^+, comme l’indique la figure 1.

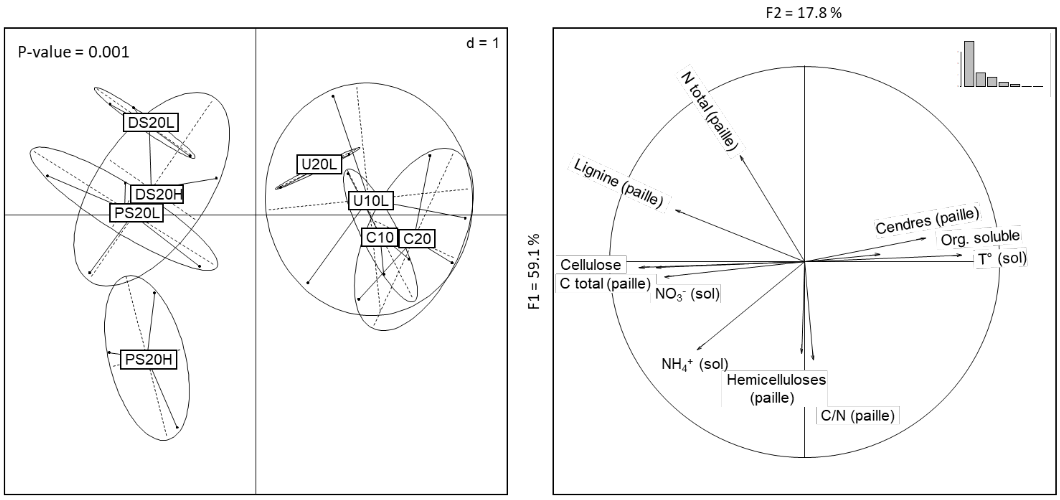

Figure 1 : Analyse en composantes principales sur les paramètres physicochimiques du sol (T°, NO_3^-, NH_4^+) et de la paille (Teneur en composés organiques solubles, hémicelluloses, cellulose, lignine, cendres, N total, C total et C:N) en fin d’expérimentation (~jour 350), en fonction des mélanges de paille à 20 t ha^{-1} (20) ou 10 t ha^{-1} (10) et fertilisants organiques, lisier de porc (PS) et boue de STEP (DS) ou urée (U), avec une quantité d’azote en apport conventionnel (C:N élevé, H) ou 10 fois l’apport conventionnel (C:N faible, L) ; ou en fonction des témoins avec l’apport en paille à 20 t ha^{-1} (C20) ou 10 t ha^{-1} (C10)

3.2. Émissions de GES

Nos résultats soulignent le rôle clé du type de fertilisant sur les émissions de GES à court terme après la fertilisation. Nous avons observé que les flux d’émissions moyens de CO_2 et de N_2O les plus élevés étaient obtenus en appliquant le lisier de porc qui a une forte teneur en eau et est riche en N minéral. L’application du lisier de porc s’est traduite dans les deux campagnes expérimentales par des pics d’émissions de CO_2 et de N_2O forts dans les premiers jours après l’application. Nous avons attribué ce résultat à, d’une part la spéciation du C et du N, et d’autre part à la forme physique sous forme liquide ou pâteuse du lisier. Les pics de CO_2 ont été attribués à une décarboxylation (Chantigny et al. 2002, Aita et al. 2012) et les pics de N_2O à une nitrification dénitrifiante ou une dénitrification provoquée par l’oxydation de l’ammonium, suivie de la réduction incomplète des nitrates en nitrites, et consécutivement en protoxyde d’azote dans des conditions de sol appauvrit en oxygène localement, suite à l’infiltration du lisier liquide. A l’inverse, les cinétiques des émissions de GES provenant des engrais solides ont été régis par des facteurs environnementaux, dont certains pourraient être contrôlés, tels que l'apport en eau ou la quantité de fertilisant (Charles et al. 2017). La première campagne a montré que les pics d'émissions de N_2O survenaient lors des apports d'eau par précipitation ou irrigation. Le même cas de figure a été observé pendant la deuxième campagne, avec aussi la possibilité d'identifier la teneur en N minéral du sol comme facteur important. Si on s'intéresse aux moyennes d'émissions de GES, calculées sur les périodes de j0 à j49, sur les deux années d'expérimentations, on constate que malgré les quantités différentes de paillis, les émissions de GES des mélanges paille/boue de STEP sont pratiquement identiques (Fig. 2). Or ce n'est pas le cas pour le lisier de porc. En effet, les occurrences de mesures différentes entre les deux campagnes expérimentales peuvent partiellement expliquer ces résultats. Lors de la première année, les mesures ont commencé quelques heures après épandage, alors que lors de la deuxième année ces mesures ont commencé le lendemain de l'épandage. La différence de qualité physicochimique, préalablement discutée, mais aussi la quantité de paille plus importante lors de la deuxième année d'expérimentations peuvent aussi expliquer ces résultats. Le lisier plus riche en matière sèche et la quantité de paille plus importante lors de cette deuxième année aurait pu retarder/limiter l'infiltration de l'azote du lisier dans le sol, ce qui a pour conséquences d'augmenter les pertes de N par volatilisation, en absence d'irrigation immédiate après épandage, ce qui était le cas dans notre étude. En effet, Genermont et al. (2003) avaient constaté une perte totale de l'azote ammoniacal du lisier de porc épandu au bout de trois jours avec un apport d'eau limité (quelques mm), alors que la volatilisation était réduite à partir de 10 mm d'eau apportés. En conséquence, les émissions de GES réduites mesurées pour le lisier de porc lors de la deuxième année, reflètent une moindre accessibilité de l'azote pour la biomasse microbienne, due à un retard d'infiltration dans le sol et une volatilisation plus importante.

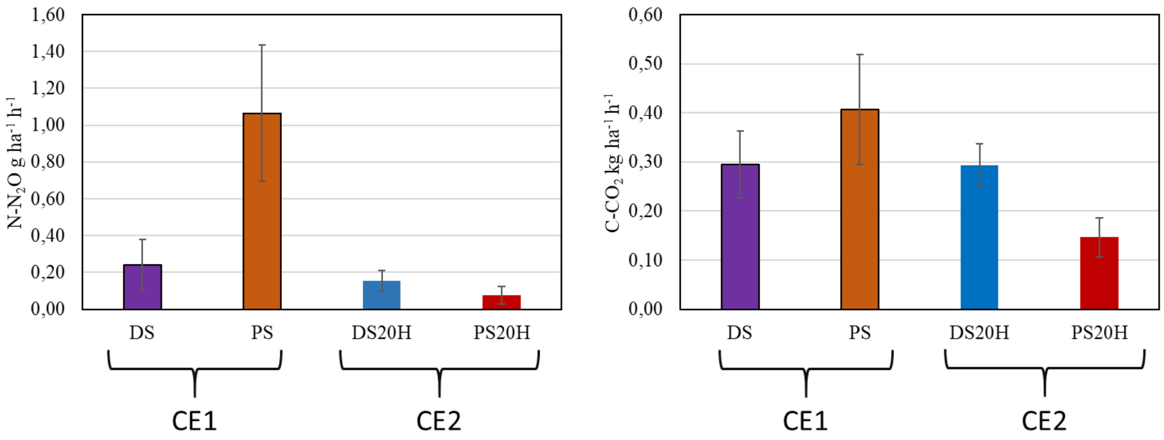

Figure 2 : Comparaison des moyennes d'émissions de GES entre la première campagne

expérimentale (CE 1) et la deuxième campagne expérimentale (CE 2), pour les traitements avec apport en N en quantités conventionnelles en tant que boue de STEP (DS) et lisier de porc (PS) sur 11 t ha^{-1} de paille, respectivement DS et PS, et sur 22 t ha^{-1} de paille, respectivement DS20H et PS20H.

3.3. Quelques préconisations

Notre étude suggère qu'une quantité de paille plus importante laissée au sol est plus propice au stockage du carbone dans le sol. Ce résultat devrait être considéré lors de la mise en place de programmes visant l'augmentation de la séquestration du carbone dans le sol. A l'inverse les acteurs intéressés par l'exportation de la paille pour une valorisation hors champ devraient prendre en compte la possibilité de compromettre les avantages environnementaux du paillage.

L'utilisation de fertilisants organiques doit être considérée bénéfique dans le cas de l'application de fertilisants riches en N organique et pauvre en teneur en eau comme la boue de STEP séchée. En termes de disponibilité de l'azote minéral dans le sol, celui-ci est minéralisé de manière progressive et disponible avec un effet « retard », expliqué par le besoin d'être prélevé et minéralisé par les microorganismes du sol. Ces informations devraient être considérées lors de l'élaboration des plans d'épandage pour répondre aux besoins spécifiques de la biologie de la plante (recherche d'une synchronisation entre l'offre en formes minérales de N et les besoins des cultures).

Enfin, nous suggérons que les efforts soient concentrés sur une meilleure intégration dans les modèles des mécanismes d'interactions entre les intrants organiques et les résidus de culture dans les agroécosystèmes où le paillage est pratiqué. Une telle connaissance permettrait des scénarios de modélisation fournissant des informations plus précises, en réponse à des défis tels que l'utilisation concurrentielle de la biomasse, la transition vers une économie circulaire et la séquestration du carbone.

4. Perspectives

4.1. Utiliser les données expérimentales dans une approche intégrative

Ce travail a permis de mettre en évidence la nécessité d'adopter une approche systémique pour répondre à la problématique d'émissions de GES et de fertilisation de la canne à sucre. Il peut aussi servir pour l'intégration des mécanismes identifiés dans des modèles de transformation de la biomasse à court terme lors d'apport de mélanges de paillis et de PRO. Dans ce sens, nos résultats peuvent servir comme des entrées potentielles pour des modèles intégratifs à l'échelle de la parcelle ou pour des modèles spatialisés (ensemble de parcelles sur un territoire) et la gestion territorialisée de la biomasse. Une telle approche permettrait de réaliser des bilans du système étudié et servir de base à l'élaboration de scénarios pour d'autres conditions. À partir de ces simulations, une approche analytique par bouquets de services écosystémiques (Gos 2013), pourrait fournir des pistes pour des stratégies de gestion dans des agroécosystèmes canniers variés.

4.2. Affiner les mécanismes

Même si nos résultats peuvent défendre une certaine généricité à court terme, car traitant de la qualité physicochimique des fertilisants et des quantités de paille de canne à sucre proches des pratiques agricoles, nous ne pouvons pas prétendre que dans des conditions pédoclimatiques différentes on aboutirait aux mêmes résultats. On pourrait donc envisager d'étendre la démarche expérimentale à des types de sol ou de climats contrastés, avec des dynamiques d'apport en eau ou de fertilisants différents, comme par exemple un système non-irrigué ou une fertilisation séquentielle. Ainsi, une des perspectives potentielles serait d'adopter une approche de quantification expérimentale des entrées et des sorties (bilan), pour laquelle il serait nécessaire d'effectuer des mesures d'émissions de GES de manière continue (automatisée) et d'estimer les autres sources/puits des deux éléments C et N, parmi lesquelles on peut citer les plantes (et les racines) et la biomasse du sol au sens large. La volatilisation de l'azote et les pertes verticales (lixiviation) ou horizontales (ruissellement), sont aussi à prendre en compte, ainsi que la diffusion de ces éléments dans un espace complexe à 3 dimensions tel que le sol, devraient y être intégrés. Toutes ces dynamiques peuvent être affinées en utilisant des techniques de marquage isotopique. Le site expérimental utilisé comme support des essais de terrain de nos travaux de thèse s'y prête.

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Titre : Émissions de gaz à effet de serre et stocks de carbone du sol associés aux mélanges de résidus de récolte et de produits résiduaires organiques dans la culture de la canne à sucre

Mots clefs : Paillis de canne à sucre, Fertilisants organiques, Émissions de gaz à effet de serre, Carbone, Azote, Modélisation

Résumé

Dans le contexte environnemental actuel il est nécessaire d'optimiser l'utilisation des ressources pour réduire les déchets et les émissions de gaz à effet de serre (GES). L'utilisation de ressources renouvelables et le recyclage sont au cœur du modèle d'économie circulaire, selon lequel le déchet est une ressource mobilisable, transformable, réutilisable et par conséquent non limitée. En agriculture, le cercle vertueux de l'économie circulaire peut être mis en place à travers le recyclage des matières organiques en tant que fertilisants et l'adoption d'une approche agroécologique qui privilégie les services écosystémiques pour lutter contre les nuisibles, protéger le sol et augmenter ses stocks de carbone, dont un des exemples est le paillage. Dans la culture de canne à sucre, la paille laissée à la surface du sol, est aussi de plus en plus convoitée par les industriels pour la production d'agrocarburant de seconde génération. L'émergence d'une telle concurrence d'usage de la biomasse et les enjeux environnementaux sous-jacents imposent d'évaluer à la fois les avantages et les inconvénients agronomiques et environnementaux du recyclage conjoint du paillis de canne à sucre et des fertilisants organiques. Les objectifs de ce travail sont donc i) de comparer l'effet des fertilisants de qualité physicochimique contrastée, sur la décomposition d'un paillis de canne à sucre, et ii) d'évaluer les potentialités d'émission de GES des mélanges paillis-fertilisant en fonction de la quantité de paille laissée et de la qualité des fertilisants apportés. La stratégie de recherche adoptée visait dans un premier temps à détecter des interactions carbone/azote lors du recyclage conjoint de la paille et des fertilisants organiques. Nous avons ainsi testé la prédictibilité des dynamiques de minéralisation du C et du N des matières organiques seules, ou en mélange (paille/fertilisant) en laboratoire, par un modèle additif simple et un modèle mécaniste de transformation du carbone et de l'azote dans le sol – CANTIS. Nos résultats ont montré que les deux modèles ont surestimé la minéralisation du C et n'ont pas prédit correctement la minéralisation du N des deux mélanges. Cette interaction antagoniste pour les mélanges a pu être corrigée par l'application d'un facteur de contact dans CANTIS, qui reflète la diminution de la biodisponibilité du C et du N, due à des hétérogénéités de distribution à une échelle fine du sol. Dans un deuxième temps, nous avons effectué des essais au champ (en condition réelles) pour suivre à la fois la décomposition de la paille de canne à sucre en mélange avec des fertilisants organiques et les émissions de GES. La quantité de paille décomposée a été proportionnelle à la quantité initiale laissée et n'a été affectée ni par la quantité de paille initiale, ni par le type de fertilisant apporté. Cette proportionnalité est transposable aux potentialités de stockage de carbone dans le sol et devrait être considérée lors de la mise en place de programmes de séquestration de carbone dans le sol ou lors de l'exportation de la paille pour une utilisation par ailleurs. Le type de fertilisant a un rôle clef sur les émissions de GES à court terme après la fertilisation. Les flux d'émissions moyens de CO_2 et de N_2O les plus élevés ont été obtenus en appliquant du lisier de porc, qui a une forte teneur en eau et est riche en N minéral. A l'inverse, les cinétiques des émissions de GES des fertilisants solides ont été régis par des facteurs environnementaux, dont certains pourraient être contrôlés, tels que l'apport en eau ou la quantité de fertilisant. L'utilisation de fertilisants organiques est bénéfique lorsqu'ils sont riches en N organique et pauvres en teneur en eau, comme la boue de station d'épuration sèche, mais la minéralisation de l'azote dans ce cas est progressive et nécessite l'élaboration de plans d'épandage spécifiques pour répondre aux besoins des cultures. Une meilleure intégration des interactions entre les différentes sources de N et de C devrait être envisagée, afin de développer la modélisation en tant qu'outil précis pour la gestion d'un agro-écosystème.

Title : Greenhouse gas emissions and soil carbon stocks associated with crop residues and organic fertilizers mixtures in sugar cane cropping systems

Keywords : Sugarcane Mulch, Organic Fertilizers, Greenhouse Gas Emissions, Carbon, Nitrogen, Modeling

Abstract

In the current environmental context, it is crucial to optimize the use of resources to reduce waste and greenhouse gas (GHG) emissions. Renewable resources use and recycling lies at the heart of the circular economy model, in which a waste is a mobilizable, transformable, reusable and therefore renewable resource. In agriculture, the "virtuous circle" of circular economy can be achieved through the recycling of organic wastes as fertilizer and the adoption of an agroecological approach that favors ecosystem services for pest control, soil protection and carbon stocks increase, one of the examples of which is mulching. In sugarcane cropping systems straw left on the soil surface is also increasingly coveted by industrial sector to produce second-generation agrofuels. The emerging competition for biomass use and the underlying environmental issues make it necessary to assess both, the agronomic and the environmental advantages and disadvantages of joint recycling of sugarcane mulch and organic fertilizers. Therefore, the objectives of this work are i) to compare the effect of fertilizers of contrasting physicochemical quality, on the decomposition of a sugarcane mulch, and ii) to evaluate the potential of GHG emissions from mulch-fertilizer mixtures, according to the quantity of straw left and the quality of the fertilizers applied. The research strategy adopted in a first step aimed to detect carbon/nitrogen interactions during the combined recycling of straw and organic fertilizers. We have thus tested the predictability of the mineralization dynamics of C and N of the organic materials alone, or in mixture (straw/fertilizer) in the laboratory, by a simple additive model, and a mechanistic model of carbon and nitrogen transformation in the soil - CANTIS. Our results showed that both models overestimated the C mineralization and did not correctly predict the N mineralization of the mixtures. This antagonistic interaction for the mixtures was corrected by the application of a contact factor in CANTIS, which reflects the bioavailability decrease of C and N, due to distribution heterogeneities at a fine scale within the soil. In a second step, we conducted field trials (under real conditions) to measure both the decomposition of sugarcane straw mixed with organic fertilizers and GHG emissions. The amount of straw decomposed was proportional to the initial amount left and was affected neither by the amount of straw nor the type of fertilizer provided. This proportionality is transposable to the potential for carbon sequestration in the soil and should be considered when setting up carbon sequestration or when exporting the straw for alternative use. The type of fertilizer plays a key role in GHG emissions in the short term after fertilization. The highest average CO_2 and N_2O emission fluxes were obtained by applying pig slurry, which has a high water content and is rich in mineral N. Conversely, the kinetics of GHG emissions from solid fertilizers have been governed by environmental factors, some of which could be controlled, such as water intake or fertilizer quantity application. The use of organic fertilizers is beneficial when they are rich in organic N and poor in water content, such as dry sewage sludge, but the mineralization of nitrogen in this case is gradual and requires elaboration of a specific application strategies to meet crop needs. A better integration of the interactions between the different N and C sources should be considered, in order to develop modeling as a precise tool for the management of an agroecosystem.

POLE RECHERCHE
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