



Effet de travail du sol sur les stocks et flux de C et N dans un sol limoneux de grandes cultures du bassin Parisien

Katrien Oorts

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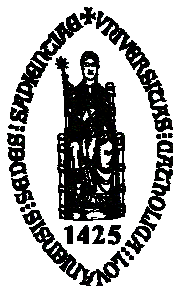
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door

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SUMMARY

For many centuries, the conventional mouldboard tillage system was used in agriculture to control the development of weeds, to incorporate crop residues into the soil, to recycle leached nutrients back to the surface and to create an adequate structure before planting. However, after the development of herbicides the need for ploughing was questioned and reduced tillage systems were introduced. These reduced tillage systems have two main characteristics: (i) the soil is not entirely turned over and (ii) the soil is always entirely or partially covered by residues. The shift from mouldboard ploughing to no-tillage so induces changes in the soil structure and in the location of soil organic matter and crop residues. This results in changes in soil climate (soil temperature and soil water content) and in several biological, chemical and physical soil properties. The combination of all these modifications has an important impact on C and N transformations in the soil.

The overall objectives of this work were twofold. First, we quantified the changes in C and N pools and in C and N fluxes between different long-term (32 years) tillage systems in cereal cropping systems in northern France, and second, we studied the effects of soil climatic conditions, soil structure and biological and physical properties of the soil on the differences in the C and N cycles between those tillage systems. This work focused mainly on those parameters with important agronomical or environmental impacts: soil organic C and N contents and distribution, soil mineral N dynamics and CO₂ and N₂O emissions. Two contrasting tillage systems were considered, i.e. conventional mouldboard ploughing to 20 cm depth (CT) and no-tillage (NT). These systems were studied on two different sets of plots with a maize-wheat rotation on the same experimental site at Boigneville in the Parisian Basin in Northern France.

After 32 years, NT presented 5-15% larger C stocks and 3-10% larger N stocks compared to CT, but these differences were not always statistically significant. Soil organic C and N concentrations decreased with increasing depth in NT, whereas they were relatively homogeneously distributed through the plough layer in CT. The small stock differences were further explored by examining the changes at different levels of structural complexity. Mineral-associated N and particulate organic matter each accounted for about 50% of the total difference in N stock. However, 66% of the total difference in C stock was due to differences in the particulate organic matter (58%) and free residues (8%) fractions. Additional C and N were detected in NT in the water stable macroaggregates. Our results suggest that the larger C and N stocks in NT are attributed to (i) enhanced macroaggregate formation in the 0-5 cm

layer due to higher microbial activity and SOM content and (ii) a better protection of soil organic matter in the 5-20 cm layer due to a larger proportion of small pores and lack of soil disruption by tillage or climate.

The tillage systems did not induce large differences in water and nitrate content in the 0-120 cm soil profile. When the LIXIM model was applied to these data the calculated 'in situ' N mineralisation rates, expressed both in calendar days and in normalised days (for soil temperature and moisture content), were comparable in both tillage systems and clearly demonstrated that the soil N supply in both systems was comparable.

NT always tended to emit more N_2O than CT. In addition, CT or NT emitted the larger amount of CO_2 in the absence of plants depending on the weather conditions (rainfall and temperature) and the amount and location of crop residues. The cumulated CO_2 emissions for the specific weather conditions of the measurement year were significantly larger for NT than for CT.

In the second part of our work we studied the effects of differences in soil climatic conditions, soil structure, organic matter location and soil biological and physical properties between the tillage systems on the observed differences in the C and N cycles. We first determined whether the differences in C and N stocks and fluxes in CT and NT were due to changes in the potential decomposition rate of the SOM. Our results clearly showed that after 32 years the potential C and N mineralisation of soil organic matter under controlled conditions (temperature and soil water pressure) was not smaller in NT compared to CT. The physical protection of the soil organic matter against mineralisation was evaluated by incubating soil samples after soil structures between 50 μm and 12.5 mm had been progressively destroyed. The samples were taken from four structural zones of the CT and NT plots: loose and dense structural zones in the plough layer of CT and the 0-5 and 5-20 cm soil layers in NT. Our results indicate that the structural zone with the largest C and N stocks and the largest amount of water stable aggregates (0-5 cm soil layer of NT) showed the smallest increase in N mineralisation and no increase in C mineralisation after soil structure disruption. Of the four structural zones, the 5-20 cm soil layer of NT showed the largest effect of physical protection of SOM.

Secondly, our measurements indicated that differences in soil temperature and soil water content between CT and NT induced differences in 'in situ' soil organic matter decomposition. These differences were often small and not systematically more favorable to decomposition over time in a given tillage system. On the other hand, a large influence of the distribution and amount of rainfall and water evaporation on the dynamics of the CO_2 fluxes was observed. In NT, rainfall induced considerable residue decomposition and, consequently, bursts of CO_2 emissions due to a sudden increase in the water content of the surface residues.

However, after a rain event, the water content of the surface residues fell rapidly and, again, seriously limited their decomposition resulting in smaller CO₂ emissions compared to CT.

Finally, C and N fluxes were simulated using the PASTIS model. Modelling provides a better understanding of the individual effects and interactions of the determining factors on C and N dynamics. The simulations showed that the larger cumulative total CO₂ fluxes in NT resulted from a more extensive crop residue decomposition and not from an enhanced SOM decomposition because the large amount of accumulated residues of previous crops in NT more than compensated for the slower residue decomposition rate of surface residues in these long-term differentiated tillage systems. The water content of the surface crop residues was found to be key in determining the magnitude of the difference in decomposition rate between the incorporated residues in CT and the surface residues in NT.

RESUME

Pendant plusieurs siècles, le sol a été labouré pour contrôler le développement des mauvaises herbes, incorporer des résidus de culture et préparer le sol avant le semis. Après le développement des herbicides la nécessité de labourer a été posée et des systèmes de travail du sol réduit ont été introduits. Ces systèmes de travail du sol réduit ont deux caractéristiques : (i) le sol n'est plus labouré et, (ii) le sol est toujours complètement ou partiellement couvert avec des résidus de culture. Le passage du labour profond au semis-direct (un système de travail du sol réduit) induit des modifications dans la structure du sol et la localisation de la matière organique du sol (MOS) et des résidus de culture. Ceci entraîne des modifications dans le climat du sol (température et humidité) et certaines propriétés biologiques, chimiques et physiques du sol. La combinaison de toutes ces modifications a une influence importante sur les transformations de l'azote et du carbone dans le sol.

Les objectifs de notre étude ont été de (i) quantifier les différences des stocks et de flux de carbone et de l'azote entre différents systèmes de travail du sol différenciés depuis 32 années dans un sol limoneux de grande culture du bassin Parisien et, (ii) expliciter les effets du climat du sol, de la structure et des propriétés biologiques et physiques du sol sur les différences de fonctionnement des cycles du carbone et de l'azote du sol. Cette étude a été essentiellement focalisée sur les variables qui ont un impact agronomique ou environnemental : carbone et azote organique du sol, dynamique de l'azote minéral du sol et les émissions de CO₂ et N₂O. Deux systèmes de travail du sol ont été étudiés : le labour (CT) et le semis-direct (NT). Ces systèmes de travail du sol ont été suivis sur des parcelles en rotation maïs-blé du site expérimental de Boigneville (91) en France.

NT présente des stocks de carbone 5 à 15 % plus importants et des stocks d'azote 3 à 10% supérieurs à ceux mesurés pour CT, mais ces différences n'ont pas toujours été statistiquement significatives. Les concentrations de C et N diminuent avec la profondeur en NT alors qu'elles sont distribuées de façon homogène dans la couche labourée en CT. La différence de stock d'azote organique associé aux argiles et limons et la différence de stock d'azote associé à la matière organique particulaire (MOP) ont chacune expliqué 50 % de la différence de stock d'azote total entre les deux systèmes. 66 % de la différence du stock de carbone total du sol ont été explicités par la différence de stock de carbone présent dans la MOP (58 %) et les résidus de culture (8 %). Le carbone et l'azote additionnel dans NT se situe dans des agrégats. Nos résultats suggèrent que les stocks de C et N plus importants pour NT peuvent être attribués à (i) la formation de macroagrégats plus prononcée dans la couche

0-5 cm due à l'activité microbienne et aux stocks de MOS plus importants et, (ii) la meilleure protection de la MOS dans la couche 5-20 cm due à une porosité du sol plus faible et à l'absence de la destruction de la structure du sol par le travail du sol ou le climat.

Les modalités de travail du sol n'ont pratiquement pas eu d'influence sur les dynamiques de l'eau et de nitrates dans le profil (0-120 cm) du sol. L'interprétation des données avec le modèle LIXIM a permis de calculer des vitesses de minéralisation comparables pour les 2 systèmes que celles-ci soient calculées avec une échelle de temps exprimée en jours calendaires ou en jours normalisés (à une température et une humidité du sol de référence). Ces résultats montrent que la fourniture d'azote minéral par le sol est similaire dans les différents systèmes de travail du sol étudiés à Boigneville.

Par ailleurs, les émissions de N_2O ont eu tendance à être plus élevées pour NT que pour CT. Les émissions de CO_2 en absence de couvert végétal ont pu être plus importantes pour l'un ou l'autre des systèmes de travail du sol en fonction des conditions climatiques et de la localisation des résidus de culture. Le cumul des quantités de CO_2 émis par NT a été significativement plus important que pour CT.

Au cours d'une seconde partie du travail, nous avons cherché à montrer si les différences de stocks et de flux de C et N entre les différentes modalités de travail du sol étaient le résultat des modifications des conditions climatiques, de la localisation et des quantités de SOM et résidus de culture ou des propriétés biologiques ou physiques du sol. D'abord, nos résultats ont montré que la minéralisation potentielle du C et N en conditions contrôlées n'a pas été moins importante pour NT comparé à CT. Par ailleurs, la protection physique de la MOS contre la minéralisation du C et N a été évaluée par incubation d'échantillons de sol dont les structures entre 50 μm et 12.5 mm ont été progressivement détruites. Quatre zones structurales ont été considérées : zones avec une structure poreuse ou compacte pour CT et horizons 0-5 et 5-20 cm pour NT. Les résultats indiquent que la destruction de la structure de l'horizon 0-5 cm de NT induit une faible augmentation de la minéralisation de l'azote et pas d'augmentation pour la minéralisation du carbone. La protection de la MOS est en réalité la plus importante pour la couche 5-20 cm du NT.

Ensuite, les différences de décomposition de la MOS entre CT et NT au champ ont été influencées par des différences de la température et de l'humidité du sol. Toutefois ces différences ont été souvent faibles et les conditions n'ont pas été systématiquement plus favorables pour la décomposition dans l'un ou l'autre des systèmes de travail du sol. Néanmoins, la distribution et la quantité de pluie et l'évaporation d'eau ont eu une influence importante sur la dynamique des flux de CO_2 . Les pluies induisent une réhumectation rapide des résidus de surface qui entraîne une augmentation importante des flux de CO_2 pour NT par rapport à CT. Après les pluies, la teneur en eau des résidus de surface diminue rapidement ce

qui limite sérieusement leur décomposition entraînant des émissions plus faibles pour NT comparé à CT.

Finalement, les flux de C et N ont été simulés avec le modèle PASTIS. Les simulations ont montré que la quantité cumulée plus importante de CO₂ émise par NT a résulté d'une décomposition plus importante des résidus de culture et pas d'une différence de décomposition des MOS. En réalité, la plus grande quantité des résidus de culture accumulée à la surface du sol dans NT fait plus que compenser la plus faible vitesse de décomposition des résidus en surface pour NT comparé avec la situation de résidus enfouis pour CT. En définitive, c'est la teneur en eau du mulch de résidus qui contrôle le plus l'amplitude de la différence de vitesse de décomposition des résidus entre CT et NT.

SAMENVATTING

Ploegen was gedurende meerdere eeuwen een standaard praktijk in de landbouw om de groei van onkruid te controleren, om gewasresten in te bodem in te werken, om uitgespoelde nutriënten weer aan het oppervlak te brengen en de bodemstructuur losser te maken voor het zaaien. De ontwikkeling van herbiciden stelde echter de noodzaak van ploegen in vraag zodat systemen met minimale bodembewerking werden ingevoerd. Deze laatste systemen hebben twee kenmerken: (i) de bodem wordt niet meer gekeerd en (ii) het bodemoppervlak is steeds geheel of gedeeltelijk bedekt met gewasresten. Deze overgang van ploegen naar minimale bodembewerking veroorzaakt veranderingen in de bodemstructuur en in de locatie van bodemorganische stof (BOS) en gewasresten. Dit resulteert in een verandering in het bodemklimaat (bodemtemperatuur en bodemvochtgehalte) en in verscheidene biologische, chemische en fysische bodemeigenschappen. De combinatie van al deze veranderingen heeft een belangrijke impact op de transformaties van koolstof en stikstof in de bodem.

De algemene doelstellingen van dit werk zijn tweevoudig. Eerst werden de verschillen in zowel bodemstocks als bodemfluxen van koolstof en stikstof bepaald tussen verscheidene lange termijn (32 jaar) bodembewerkingssystemen in Noord-Frankrijk. Ten tweede werden de effecten bestudeerd van bodemklimaat, bodemstructuur en biologische en fysische bodemeigenschappen op de verschillen in deze bodemstocks en bodemfluxen. Dit werk legt voornamelijk de nadruk op die stocks en fluxen van koolstof en stikstof die een grote impact hebben op landbouw of milieu: de hoeveelheden van bodemorganische koolstof en stikstof, de dynamiek van minerale stikstof in de bodem en de emissies van CO₂ en N₂O. Twee contrasterende bodembewerkingssystemen werden bestudeerd, namelijk conventioneel ploegen tot 20 cm diepte (CT) en minimale bodemwerking (NT). Deze twee systemen werden gevolgd op twee verschillende sets van percelen met een maïs-tarwe rotatie op dezelfde experimentele site in Boigneville in Noord-Frankrijk.

Na 32 jaar onderwerping aan verschillende bodembewerkingssystemen had NT 5-15% grotere koolstofstocks en 3-10% grotere stikstofstocks in vergelijking met CT. Deze verschillen waren echter niet altijd statistisch significant. In NT daalde de concentratie van zowel organische koolstof als stikstof met toenemende diepte onder de bodemoppervlakte, terwijl deze concentraties in CT eerder homogeen verdeeld waren over de ploeglaag. De verschillen in stocks werden verder onderzocht door ze op verschillende niveaus van de complexe bodemstructuur te bestuderen. In NT maakten stikstof geassocieerd met kleimineralen en plantendeeltjes (< 2 mm) elk ongeveer 50% uit van het totale overschot aan

organische stikstof in de bodem. Daarentegen werd 66% van het totale verschil in de koolstofhoeveelheid tussen CT en NT verklaard door verschillen in koolstof geassocieerd met plantendeeltjes kleiner dan 2 mm (58%) en gewasresten groter dan 2 mm (8%) en dus maar 34% door verschillen in koolstof geassocieerd met kleimineralen. De additionele koolstof en stikstof in NT situeerde zich in water stabiele macroaggregaten. Onze resultaten suggereren dat de grotere hoeveelheden organische koolstof en stikstof in NT konden toegeschreven worden aan (i) verhoogde vorming van macroaggregaten in de 0-5 cm bodemlaag door hogere microbiële activiteit en door een grotere hoeveelheid BOS en (ii) een grotere fysische protectie van BOS in de 5-20 cm bodemlaag ten gevolge van een groter percentage kleine poriën en van de afwezigheid van bodemverstoring door ploegen of door het klimaat.

De verschillende bodembewerkingssystemen hadden geen grote impact op de water- en nitraathoeveelheden in het 0-120 cm bodemprofiel. Met behulp van deze data, berekende het LIXIM model vergelijkbare snelheden van ‘in situ’ stikstofmineralisatie zowel uitgedrukt in kalenderdagen als in genormaliseerde dagen (voor bodemtemperatuur en -vocht) tussen de verschillende bodembewerkingssystemen. Deze resultaten tonen duidelijk aan dat het stikstofleverende vermogen van de bodem in de bestudeerde bodembewerkingssystemen niet verschillend is.

Wat betreft de emissies van de broeikasgassen CO₂ and N₂O onder veldomstandigheden is NT zeker niet verkiesbaar boven CT onder de specifieke weersomstandigheden tijdens het meetjaar in Boigneville. Het NT systeem had altijd de neiging om meer N₂O uit te stoten dan CT. Afhankelijk van de weerscondities (regenval en temperatuur) en de hoeveelheid en locatie van gewasresten waren de CO₂ emissies groter in CT of NT. In tegenstelling tot onze initiële hypothese bleek de cumulatieve CO₂ emissie over de hele meetperiode significant groter in NT dan in CT.

Het tweede deel van dit onderzoek bestudeerde de effecten van de verschillen in bodemklimaat, bodemstructuur en de biologische en fysische bodemeigenschappen op de waargenomen verschillen in de koolstof- en stikstofcycli tussen de verschillende bodembewerkingssystemen. Eerst werd bepaald of de verschillen in de koolstof- en stikstofcycli het resultaat waren van de potentiële afbraaksnelheid van BOS. Onze resultaten tonen aan dat na 32 jaar de potentiële koolstof- en stikstofmineralisatie (gemeten onder constante temperatuur en waterpotentiaal) zeker niet kleiner zijn onder NT dan onder CT. De fysische protectie van BOS tegen koolstof- en stikstofmineralisatie werd bestudeerd door de bodemstructuren tussen 50 µm and 12.5 mm in toenemende mate te vernietigen. Dit alles werd gedaan voor vier verschillende structurele bodemzones in NT en CT: ‘losse’ en ‘compacte’ structurele bodemzones in de ploeglaag van CT en de 0-5 en 5-20 cm bodemlagen in NT. Onze resultaten toonden aan dat na de vernietiging van de fysische protectie van de

BOS de bodemzone met de grootste C en N hoeveelheden en de grootste hoeveelheid waterstabile aggregaten (0-5 cm bodemlaag in NT) de kleinste stijging vertoonde in N mineralisatie en geen stijging vertoonde in C mineralisatie. Het grootste effect van fysieke protectie van BOS werd waargenomen in de 5-20 cm bodemlaag van NT.

Vervolgens toonden ‘in situ’ metingen aan dat verschillen in bodemtemperatuur en vochtgehalte tussen CT en NT een impact kunnen hebben op de afbraaksnelheid van BOS. Maar deze verschillen waren vaak klein en niet systematisch gunstiger voor afbraak in één van de twee bodembewerkingssystemen. De verdeling en de hoeveelheid regenval en waterevaporatie daarentegen had een grote invloed op de waargenomen CO₂ fluxen. In NT liggen de gewasresten steeds aan het bodemoppervlak. Daardoor induceerde regenval in NT een plotse stijging van het vochtgehalte van de gewasresten met grote pieken in CO₂ emissies tot gevolg. Na elke regenval daalde het vochtgehalte van de gewasresten aan het bodemoppervlak echter snel, wat de afbraak opnieuw sterk afremde met grotere CO₂ emissies in CT dan NT tot gevolg.

Tot slot werden de koolstof- en stikstofluxen gesimuleerd met het PASTIS model. Modelleren geeft meer inzicht in zowel het geïsoleerde effect als de interacties tussen de verschillende determinerende factoren van de C en N cycli. De simulatieresultaten toonden aan dat de grotere cumulatieve CO₂ emissies in NT het resultaat waren van een grotere afbraak van gewasresten en niet van een grotere afbraak van BOS. De grotere hoeveelheid gewasresten in NT (door de aanwezigheid van gewasresten van vorige jaren) overcompenseerde immers de tragere afbraak van gewasresten. Het verschil in afbraaksnelheid tussen de gewasresten in CT (in de bodem) en NT (op het bodemoppervlak) werd voornamelijk bepaald door het vochtgehalte van de gewasresten.

LIST OF ABBREVIATIONS

The abbreviations concerning the modeling with PASTIS (Chapter 6 and Appendix C) are not included in this list.

BOS	bodemorganische stof
C	carbon
CEC	cation exchange capacity
C _{sandfree}	C concentration expressed on a sandfree basis
CT	conventional mouldboard tillage system
CT _{Dense}	soil zone with dense structure in the 0-20 cm layer in CT
CT _{Loose}	soil zone with loose structure in the 0-20 cm layer in CT
CT _{rep}	semi-repetition of CT (crop residues were exported from 1983 to 1984 in CT _{rep} but never exported in CT)
EF	modeling efficiency
Free mineral-associated-C	C associated with silt and clay not occluded in aggregates
Free-POM	particulate organic matter not occluded in aggregates
Inter-mM-OM	OM inside macroaggregates but outside microaggregates
Intra-mineral-associated-C	C associated with silt and clay occluded in aggregates
Intra-mM-mineral-associated-OM	Mineral-associated OM in microaggregates within macroaggregates
Intra-mM-POM	POM in microaggregates within macroaggregates
Intra-POM	particulate organic matter occluded in aggregates
MD	mean difference
MOP	matière organique particulaire
MOS	matière organique du sol
MWD	mean weight diameter
N	nitrogen
nd	normalised days for temperature and water pressure

$N_{\min}$	mineral nitrogen
NT	no-tillage system
NT_{0-5}	0-5 cm soil layer of the no-tillage system
NT_{5-20}	5-20 cm soil layer of the no-tillage system
NT_{rep}	semi-repetition of NT (crop residues were exported from 1983 to 1984 in NT_{rep} but never exported in NT)
OM	organic matter
PET	Potential soil evapotranspiration
POM	particulate organic matter
RMSE	root mean square error
SOM	soil organic matter
ST	superficial tillage system
TDR	time domain reflectory
V_p	'in situ' potential N mineralisation rate
WFPS	water filled pore space

CHAPTER 1: INTRODUCTION AND OBJECTIVES

1.1 The use of different tillage systems

Today, there is a multitude of different tillage systems. In this work, only mouldboard ploughing and no-tillage systems are considered since they represent two extreme tillage systems that are used in France. The Wikipedia encyclopaedia gives the following two definitions for these tillage systems (www.wikipedia.org). The mouldboard plough is defined as a form of plough consisting of a share (blade) and hitch attached to either a tractor or livestock. The share digs into the earth and overturns the soil, burying organic matter from the previous harvest. The no-tillage system, also known as conservation tillage, is a way of growing crops from year to year without disturbing the soil through tillage. In no-till farming crop residues are left on the soil surface.

In order to better understand the use of the different tillage systems, it is interesting to remind some historical data (Robert *et al.*, 2004; Sebillotte and Meynard, 2004). The mouldboard plough was introduced in Europe by the Romans to control the development of weeds, to incorporate undecomposed residues into the soil, to recycle leached nutrients back to the surface and to break up the soil before planting. It was only after the development of herbicides that the improvement of soil structure became the main purpose of tillage and, hence, the necessity of ploughing was questioned. The first agronomical systems without soil inversion were introduced in the United States of America, Australia and Brazil. By covering the soil surface, the farmers in these countries tried to reduce the catastrophic effects of water and wind erosion that ravaged the cultivated land. In addition, reduced tillage systems were implemented in dry regions because the presence of a mulch layer on the soil surface lowered the evaporation of water and avoided desertification. In regions where nutrient resources are scarce, farmers try to improve the efficiency of nutrient recycling by using reduced tillage systems. In those regions of Western Europe where erosion is not a major problem, the main incentives to adopt reduced tillage systems are the reduction of labour time and energy per hectare — ploughing is often the most time- and energy-consuming intervention in large-scale cropping systems. Other main incentives of the farmers and especially of the governments in these regions concern the protection of the environment (quality of water, air and soil, biodiversity, etc.) in spite of the lack of consensus regarding the effectiveness of reduced

tillage in decreasing the environmental impact. Nowadays, reduced or no-till systems receive attention because of their potential to stock C into the soil.

The various reasons for the introduction of reduced tillage systems mentioned above, as well as the diversity in climate and soil conditions, have resulted in a wide variety of reduced tillage systems with two main common characteristics: (i) the soil is always entirely or partially covered by residues and (ii) the soil is not entirely turned over. Worldwide, approximately 70 million hectare cultivated land was managed with reduced tillage or no-tillage systems in 2003 (congress Iguacu 2003). Large areas with no-till systems are present in America (32% of the cultivated surface under no-tillage in the world), Brazil (25%), Argentina (19%), Australia (13%), Canada (6%) and Paraguay (2%).

Apart from the benefits, reduced tillage systems may also entail several disadvantages compared to mouldboard ploughing systems. First of all, indications exist that N₂O emissions are more elevated in no-till soils. Secondly, due to the need for weed control, more herbicides are used in reduced tillage systems. Thirdly, the surface mulch of residues may promote the presence of parasites and damaging slugs, rats and snakes. Finally, structure may cause problems in no-till systems for the establishment or emergence of some crops.

1.2 No-tillage in France

The only data on the use of reduced tillage systems in France are provided by the survey 'Pratiques culturales' conducted in 1994 and 2001 by SCEES (Service Central des Etudes et Enquêtes Statistiques du Ministère en charge de l'Agriculture) and reported by Trocherie and Rabaud (2004). This study indicates that the soil surface with reduced tillage increased between 1994 and 2001. The dominant incentives to adopt reduced tillage systems are economic in nature. The reduced tillage systems consist both of superficial tillage (< 15 cm) and no-tillage systems. For the most important crops in France, reduced tillage systems account for 6-44% of the cultivated area (in 2001), the specific percentage depending on the crop type. Although wind erosion is rare in France, water erosion is becoming an important problem in some parts of the country. It is in these regions — vulnerable to water erosion — that reduced tillage practices are the most widely adopted. Regarding soil erosion, it should be noted that American publications mainly concentrate on rill-interrill erosion, whereas erosion in France is usually caused by large and deep runoff in specific areas; little data is available on the effect of no-tillage on this type of erosion (Boiffin and Monnier, 1991).

1.3 Effects of conventional tillage and no-tillage systems on the soil structure and the movement of water through the soil profile

1.3.1 Soil structure

Soil tillage induces changes in the soil structure at different scales ranging from the dimension of the soil profile to a few micrometers (Balesdent *et al.*, 2000). The soil structure is the result of the balance between compaction (by machine traffic and soil weight), agglomeration of aggregates (by moderate compaction, climate and/or fauna), fragmentation (by climate, fauna and/or tillage) and displacement (by tillage) of soil (Roger-Estrade *et al.*, 2000). In regions with a temperate climate, frost only affects soil structure near the soil surface. In conventional mouldboard tillage systems (CT), the soil structure is mainly created by tillage operations, whereas in no-tillage systems (NT), biological processes and climate account for most structural development (Foy, 2003). Mouldboard ploughing systems result in a plough layer with an extremely heterogeneous soil structure that consists of fine soil, large soil clods with dense or loose structure, clusters of crop residues situated between the furrows and large voids (Staricka *et al.*, 1991; Roger-Estrade *et al.*, 2004). The degree of fragmentation and compaction of the ploughing layer depends largely on the soil moisture conditions during harvest and tillage operations (Roger-Estrade *et al.*, 2004). In no-tillage systems, the soil structure is less heterogeneous due to the absence of both incorporated residues and soil displacement and fragmentation during tillage (Foy, 2003). With respect to the soil aggregates, no-tillage systems increase the amount of water stable aggregates larger than 250 μm (Balesdent *et al.*, 2000; Six *et al.*, 2004) and aggregate stability is lower in CT than NT (Tebrügge and Düring, 1999).

1.3.2 Bulk density

Many authors (Roth *et al.*, 1988; Wu *et al.*, 1992; Hubbard *et al.*, 1994; Tebrügge and Düring, 1999) report a significant increase in bulk density in NT compared to CT for the soil layer corresponding to the depth of the plough layer in CT (i.e. the equivalent plough layer). However, some authors observed no significant difference (Dao, 1993; Lal, 1999; Anken *et al.*, 2004; Blanco-Canqui *et al.*, 2004) or no systematic difference (Kitur *et al.*, 1993) in bulk density between CT and NT. In fact, the upper centimetres of the NT field often present a significantly lower bulk density than CT due to a larger OM content and the continuous loosening of soil by the soil fauna (Tebrügge and Düring, 1999; Shukla, 2003). The plough pan in CT generally has a larger bulk density than the equivalent soil depth in NT (Roth *et al.*,

1988; Tebrügge and Düring, 1999). The bulk density in NT remains rather constant throughout the year. In CT, after the soil has been loosened by tillage, soil bulk density will increase again by reconsolidation under the weight of the soil mass and machinery and due to the impact of raindrops and drying/rewetting cycles (Kitur *et al.*, 1993; Hubbard *et al.*, 1994; Tebrügge and Düring, 1999).

1.3.3 Porosity and preferential flow

Soil tillage affects structural but not textural porosity (Guérif *et al.*, 2001). Structural porosity is the result of the arrangement of clods and aggregates whereas textural porosity is mainly determined by soil texture. Total porosity in the equivalent plough layer is generally smaller in NT due to a generally higher bulk density (Roth *et al.*, 1988; Shipitalo *et al.*, 2000). The total area of macropores can be larger (Wu *et al.*, 1992), smaller (Roth *et al.*, 1988) or similar (Anken *et al.*, 2004) in NT compared to CT. However, it is generally accepted that the NT system presents a larger number of continuous vertical macropores (Wu *et al.*, 1992; Shipitalo *et al.*, 2000; Anken *et al.*, 2004). The periodical tillage operations destroy the continuity of macropores (root channels, biopores and cracks) and the rate of formation of biopores is larger in NT (see below). This larger number of continuous vertical macropores induces more pronounced preferential flow in NT than in CT (Elliott and Coleman, 1988; Shipitalo *et al.*, 2000; Anken *et al.*, 2004).

1.3.4 Soil-water retention characteristics

Most authors (Wu *et al.*, 1992; Benjamin, 1993; Hubbard *et al.*, 1994; Lal, 1999; Blanco-Canqui *et al.*, 2004; Fuentes *et al.*, 2004) report the absence of consistent or major differences in soil-water retention characteristics in the equivalent plough layer between NT and CT. However, Shukla (2003) measured larger volumetric water contents at a given pressure head for the 0-10 cm layer of NT than CT. Differences in soil-water retention characteristics between the two tillage systems are mainly the result of modifications in both organic matter concentrations and bulk densities since soil texture is not affected by the tillage differentiation. Larger amounts of surface mulch result for the 0-5 cm soil layer both in increased saturated water contents and in larger water retention for the low pressure heads. For high pressure heads the opposite occurs: more surface mulch results in less water retention (De Vleeschauwer *et al.*, 1980). In addition, higher bulk densities result in a smaller water retention for the low pressure heads and a larger water retention for the high pressure heads (Reicosky *et al.*, 1981).

1.3.5 Hydraulic conductivity

There is no consensus in the literature regarding the effect of tillage systems on the saturated hydraulic conductivity (K_{sat}). Datiri and Lowery (1991), Wu *et al.* (1992), Lal (1999) and Blanco-Canqui *et al.* (2004) generally did not observe any significant differences in K_{sat} between the equivalent plough layer of CT and NT. Some sites present larger K_{sat} values in the equivalent plough layer of NT (Wu *et al.*, 1992; Fuentes *et al.*, 2004) whereas Hubbard *et al.* (1994) reported smaller K_{sat} values for NT. For the surface soil layer of NT, Benjamin (1993) and Shukla (2003) observed larger K_{sat} values while the latter measured similar values for the 10-20 cm soil layer of both NT and CT. The unsaturated hydraulic conductivity as a function of soil water pressure is similar in both tillage systems (Datiri and Lowery, 1991; Wu *et al.*, 1992; Benjamin, 1993; Fuentes *et al.*, 2004). These contradictory results found in literature are probably due to the fact that the effect of tillage on the hydraulic conductivity is mainly the result of two opposing factors: a generally smaller total porosity in NT decreases the hydraulic conductivity whereas a continuity and/or a larger number of macropores in NT increases the hydraulic conductivity compared to CT.

1.3.6 Surface sealing, water infiltration, water percolation and run-off

It is generally accepted that no-tillage reduces soil losses by water run-off (Kitur *et al.*, 1993; Tebrügge and Düring, 1999). These reduced soil losses are mainly the result of the better protection against raindrop impact in NT and, to a lesser extent, to a lower amount of water run-off in NT (Boiffin and Monnier, 1991; Tebrügge and Düring, 1999). A larger amount of surface mulch (De Vleeschauwer *et al.*, 1980) or an increased mulch cover (Roth *et al.*, 1988) reduces both the impact of raindrops on the soil surface and sealing of the soil surface. As a result, surface sealing is reduced in NT (Tebrügge and Düring, 1999; Shipitalo *et al.*, 2000).

Water infiltration is generally greater and water losses by run-off are usually smaller in NT systems (Kitur *et al.*, 1993; Hubbard *et al.*, 1994; Tebrügge and Düring, 1999; Shipitalo *et al.*, 2000; Shukla, 2003) due to the reduced surface sealing, the larger number of continuous vertical macropores and/or the absence of a plough pan. However, the surface roughness is smaller in NT than CT. As a consequence, less water will be stored in the micro-depressions on the soil surface and water run-off will increase (Boiffin and Monnier, 1991). In situations with very small amounts of crop residues (as after potato harvesting), this increased surface roughness may counterbalance the positive effect of no-tillage on water and soil run-off (Boiffin and Monnier, 1991). The amount of water that percolates through the soil profile in no-till systems is increased by the greater degree of water infiltration and less soil

evaporation, although this increase may be partly compensated by enhanced transpiration of the crop (Shipitalo *et al.*, 2000).

1.4 Effects of CT and NT on the soil climate

No-till systems induce fewer fluctuations in soil temperature and the soils have a tendency to warm up more slowly in spring due to the presence of a surface mulch, denser soil and/or larger water contents (Germon *et al.*, 1994; Balesdent *et al.*, 2000). During summer, Fortin *et al.* (1996), Dao (1998), Kessavalou *et al.* (1998b) and Alvarez *et al.* (2001) measured lower temperatures in NT than CT whereas no large differences were observed in colder periods (Al-Kaisi and Yin, 2005).

Volumetric water content is often higher in NT systems (Fortin *et al.*, 1996; Dao, 1998; Alvarez *et al.*, 2001; Al-Kaisi and Yin, 2005), particularly in the 0-5 cm soil layer. This larger water content in NT is mainly the result of reduced evaporation and run-off due to the presence of the surface mulch and/or greater water retention (Shukla, 2003).

Most authors report a higher percentage of water filled pore space (WFPS) in NT than in CT (Doran, 1987; Kessavalou *et al.*, 1998a; Kessavalou *et al.*, 1998b). However, Dharmakeerthi *et al.* (2004) measured similar WFPS for the 0-30 cm layer of both NT and CT. The percentage of WFPS is determined by the total porosity, the pore size distribution and the antecedent soil water content, and determines the aeration of the soil profile. This suggests that anaerobic conditions will more frequently exist in NT than in CT.

1.5 Effects of CT and NT on chemical and biological soil properties

1.5.1 Soil pH

When the plots are not limed, the surface layer of NT has a significantly lower pH than CT due to the acidifying effect of the more pronounced nitrification (Blevins *et al.*, 1983). In contrast, the surface layer of a limed NT soil presents a slightly larger pH than CT due to the absence of mixing of the applied lime with the soil (Blevins *et al.*, 1983; Doran, 1987). No pronounced differences in pH are found between CT and NT for the deeper soil layers in either limed or unlimed plots.

1.5.2 Soil C and N concentrations and stocks

The concentration of soil organic C and N is highly stratified in NT: C and N concentrations decrease rapidly with increasing soil depth (Carter and Rennie, 1982; Doran, 1987; Wander *et al.*, 1998; Tebrügge and Düring, 1999). As a consequence, C and N stocks in the surface 0-5 cm soil layer are generally significantly larger in NT than CT (Doran, 1987; Wander *et al.*, 1998; Tebrügge and Düring, 1999; Kristensen *et al.*, 2000; Denef *et al.*, 2004). The often observed higher C to N ratio of soil organic matter (SOM) in this surface layer of NT is due to the presence of larger amounts of particulate organic matter (Stockfisch *et al.*, 1999; Kristensen *et al.*, 2000). Considering the soil layer corresponding to the plough layer of CT (i.e. equivalent plough layer), NT tends to have larger C and N stocks expressed in kg per hectare, although there is no consensus about the magnitude of the differences between CT and NT: some authors reported negligible differences, others found considerable differences (Wander *et al.*, 1998; Six *et al.*, 1999a; Tebrügge and Düring, 1999; Yang and Wander, 1999; Balesdent *et al.*, 2000; Yang and Kay, 2001; Denef *et al.*, 2004). Anken *et al.* (2004) reported no differences at all in C and N stocks between CT and NT.

In tillage systems cultivated for many generations before differentiation, Balesdent *et al.* (1990) attributed the larger total C stocks in NT to the larger proportion of initial C stocks remaining. Stocks of new C (i.e. C from crop residues added after tillage differentiation) were even found to be slightly smaller in NT than CT. Initial C stocks were spread homogeneously throughout the equivalent plough layer both for CT as NT, whereas the new C accumulated mainly near the soil surface in NT and was homogeneously distributed in CT. It has been observed that the surplus of SOM accumulated in a minimum tillage system over 20 years may disappear within one year after a return to mouldboard ploughing (Stockfisch *et al.*, 1999).

1.5.3 Soil microbial biomass

Most microbial biomass is concentrated near the soil surface of NT, in line with the stratification of organic matter (Carter and Rennie, 1982; Doran, 1987; Kessavalou *et al.*, 1998b). This suggests larger fluxes of C and N mineralisation and immobilisation near the soil surface for NT. In CT, the microbial biomass is homogeneously distributed throughout the plough layer (Doran, 1987; Kessavalou *et al.*, 1998b; Kristensen *et al.*, 2000). Considering the entire equivalent depth of this plough layer, Carter and Rennie (1982) and Kessavalou *et al.* (1998b) observed no major differences in the amount of microbial biomass C and N between either tillage system. However, the microbial biomass C and N in the surface layer of NT is significantly larger than in CT.

The composition of the community of crop residue decomposers is affected by the lack of incorporation of the residues in NT: NT contains a larger amount of fungi and a larger proportion of fungi over bacteria than CT (Holland and Coleman, 1987; Frey *et al.*, 1999; Six *et al.*, 2002). These mulch-associated fungi carry out a simultaneous bidirectional translocation of soil-derived N and mulch-derived C. Most of this mulch-derived C is afterwards located in soil macroaggregates (Frey *et al.*, 2003). In CT, bacteria are the predominant composers of incorporated crop residues (Holland and Coleman, 1987; Frey *et al.*, 1999).

1.5.4 Earthworms

Long-term no-tillage or reduced tillage systems increase the number and weight of earthworms, leading to the creation of a larger number of biopores (Elliott and Coleman, 1988; Tebrügge and Düring, 1999; Anken *et al.*, 2004; Balabane *et al.*, 2005). De Vleeschauwer *et al.* (1980) observed that the earthworm activity increased with increasing mulch rate. The number of vertically deep-drilling species shows the greatest increase, because these species particularly appreciate undisturbed soils with lots of surface residues. Continuous vertical biopores favour preferential flow and are highly resistant to pressure loading.

1.6 Physical, chemical and biological drivers of the C and N cycle in soil in CT and NT

The modified physical and chemical properties have an important effect on the performance of the microbial biomass which, in turn, determines to a large extent the C and N transformations in the soil. Figure 1.1 gives a schematic representation of the C and N cycles in the soil.

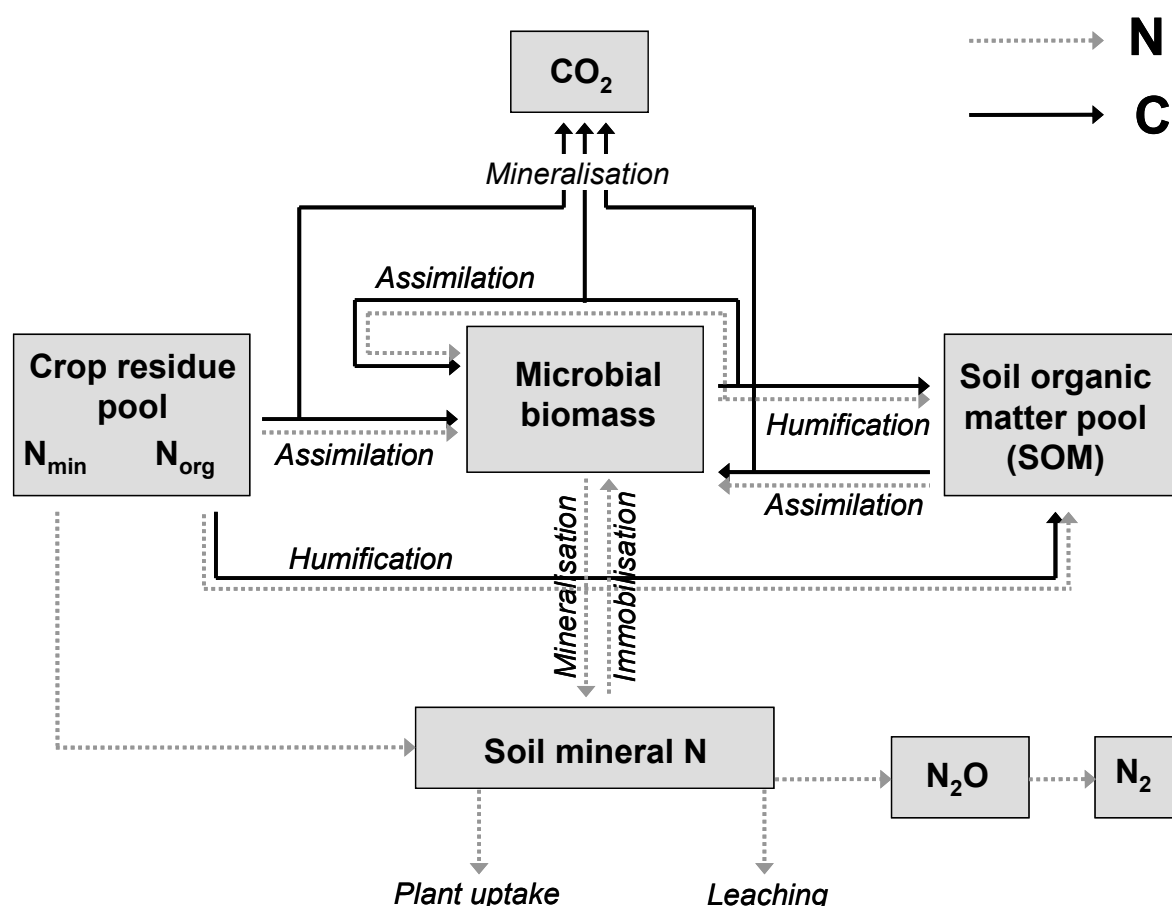


Figure 1.1: Schematic representation of the C and N cycles in the soil

1.6.1 C and N mineralisation, C assimilation, N immobilisation and C and N humification

The decomposition of SOM and crop residues is determined by various processes: C and N mineralisation (i.e. transformation of organic C and N into CO₂ and mineral N), C assimilation and N immobilisation by the microbial biomass and, finally, C and N humification (Figure 1.1).

The decomposition of SOM under controlled conditions (i.e. at constant temperature and soil water pressure) is greatly determined by the chemical and physical properties of SOM and the physical protection of SOM. The physical protection of SOM against decomposition is often reported to be more pronounced in NT than CT due to a larger number of water stable aggregates, a higher aggregate stability and a larger proportion of micropores (Balesdent *et al.*, 2000). Indeed, the aggregates decrease the decomposition rate of the occluded SOM (Wander *et al.*, 1998; Christensen, 2001). The soil porosity may limit SOM decomposition by the localization of SOM in pores not accessible to microorganisms, the limitation of nutrient

supply to microorganisms and the restriction of predation of those microorganisms (Elliott and Coleman, 1988; Hassink, 1992; Ladd *et al.*, 1993; Strong *et al.*, 2004). However, the chemical and physical properties of SOM can counterbalance the effect of the physical SOM protection. For example, the surface layer of NT often has a larger proportion of particulate organic matter (Wander *et al.*, 1998) which, in turn, presents a larger specific C and N mineralisation rate compared to mineral associated organic matter (Alvarez *et al.*, 1995). Under controlled conditions, several authors reported more net C and N mineralisation for the 0-5 cm soil layer of NT than of CT (Beare *et al.*, 1994; Kandeler *et al.*, 1999; Kristensen *et al.*, 2000). However, within the ploughing depth as a whole, most authors did not observe any difference between CT and NT, either in total potential net C and N mineralisation, or in the amount of mineralisable N (Doran, 1987; Beare *et al.*, 1994).

Especially the effect of no-tillage on N supply by the soil under farming systems is yet unclear. Farmers need more information on soil N supply in order to optimise their fertilisation management. The soil climate plays an important role on the decomposition of organic matter (soil organic matter and crop residues) under field conditions. On the one hand, decomposition increases with increasing temperature and soil water pressure, while on the other hand, it is limited by anaerobic conditions when the soil moisture content is too high (Rodrigo *et al.*, 1997). The above mentioned differences in both soil climate and organic matter location will induce differences in the decomposition of organic matter between CT and NT. In CT, the crop residues are incorporated by tillage whereas in NT, they always remain on the soil surface, leading to less contact with the soil inorganic N and the clay matrix. This reduced contact with the soil inorganic N may induce a slower decomposition whereas the reduced contact with the clay matrix may generate a faster residue decomposition in NT than CT (Balesdent *et al.*, 2000). In addition, surface residues in NT experience larger fluctuations in temperature and water content, often resulting in water contents below the optimum for residue decomposition. These differences in soil climate will certainly cause temporal differences in the decomposition of soil organic matter and crop residues between CT and NT, although the annual decomposition may still be similar.

1.6.2 CO₂ emissions as product of decomposition

The emission of the greenhouse gas CO₂ from agricultural soils is a product of organic matter decomposition and represents the largest source of C loss from the soil system. In contrast to the generally acknowledged tendency of larger C stocks in NT than in CT, there is no consensus about the effect of tillage systems on CO₂ emissions. The CO₂ fluxes under NT compared to CT may be larger (Hendrix *et al.*, 1988), similar (Fortin *et al.*, 1996; Kessavalou *et al.*, 1998b; Aslam *et al.*, 2000) or larger for some periods and smaller for others (Ball *et al.*,

1999; Vinten *et al.*, 2002). For the short period after tillage, some authors report significantly smaller CO₂ emissions for NT compared to CT systems (Reicosky and Lindstrom, 1993; Dao, 1998; Kessavalou *et al.*, 1998b; Alvarez *et al.*, 2001; Al-Kaisi and Yin, 2005). The large research gap concerning the effect of tillage systems on the emission of CO₂ mainly consists in the lack of continuous CO₂ measurements for long-term trials in periods other than directly after tillage. In addition, the reported differences in CO₂ emissions between NT and CT are often influenced by different annual C and N inputs due to different crop yields in the two tillage systems.

1.6.3 N₂O emissions as product of denitrification and nitrification

The N₂O emissions from soil are produced both by the denitrification of nitrate and the nitrification of ammonium (Hénault *et al.*, 2005). N₂O emissions are generally larger in no-tillage systems due to a higher prevalence of anaerobic conditions, locally higher C availability and/or locally larger mineral N content (MacKenzie *et al.*, 1998; Ball *et al.*, 1999; Vinten *et al.*, 2002; Baggs *et al.*, 2003). However, Kessavalou *et al.* (1998b) and Choudhary *et al.* (2002) measured similar N₂O emissions in NT and CT. Although N₂O emissions are negligible from an agricultural point of view (due to the small quantities of N lost through N₂O emission), they are important from an environmental point of view because N₂O is a greenhouse gas with a large warming potential (IPPC, 2001).

1.6.4 Dynamics of soil inorganic N

The amount of mineral N is influenced by the decomposition of SOM and crop residues, the addition of N fertilisers, N uptake by crops and N losses by leaching and gaseous emissions. Langlet and Rémy (1976) generally did not observe any difference between CT and NT in the dynamics of the amount of mineral N in the 0-90 cm soil profile. Germon *et al.* (1994) concluded that CT may show greater mineralisation after tillage and in spring (due to the higher temperatures in CT), but that the annual N mineralisation in CT and NT is similar. Finally, the acidification of the surface horizon in NT may affect the activity of nitrifying bacteria (i.e. the transformation of NH₄⁺ into NO₃⁻), which are the most sensitive to soil pH. However, there is no consensus in the literature on the effect of reduced tillage on nitrification (Germon *et al.*, 1994).

1.6.5 N leaching

There is also no consensus on the effect of tillage systems on N leaching (Elliott and Coleman, 1988; Germon *et al.*, 1994; Shipitalo *et al.*, 2000). Reduced tillage systems present

increased water infiltration and preferential flow (see above). This preferential flow induces a considerable amount of N leaching when large amounts of inorganic N are present and when rain intensity is high and the soil profile wet. The effect of preferential flow on N leaching depends on the timing of the nitrogen fertiliser application as well as on the time course of N mineralisation in relation to rain events, because diffusion, adsorption, volatilisation, immobilisation and plant uptake of the mineral N decrease its availability for N leaching. The leaching of solutes within the small pores of the soil matrix depends on the exchange rate with macropore water and on piston flow (Elliott and Coleman, 1988; Shipitalo *et al.*, 2000). Shipitalo *et al.* (2000) reported similar annual amounts of leached NO_3^- in CT and NT because the larger amount of NO_3^- loss by piston flow in CT compensated the earlier larger loss by preferential flow in NT. However, the leaching of NH_4^+ was greater in NT than CT because nearly all NH_4^+ was adsorbed after the first rains and the surplus of NH_4^+ loss by piston flow in CT was smaller than the surplus by preferential flow in NT. The differences in N losses through leaching to the groundwater depend both on the groundwater depth and on the parent material below the ploughing depth (Shipitalo *et al.*, 2000). In addition to N leaching, more N may be lost in CT by surface run-off.

1.6.6 Crop yields

Germon *et al.* (1994) reviewed the effect of different tillage systems on crop yield and fertiliser use efficiency. They found that crop yields are determined by the interaction between the soil mineral N supply and the physical and climatic conditions for crop growth. In some cases, crop yields in NT and CT are similar regardless of the N fertilisation rate. In most cases, crop yields are lower in NT at low fertiliser doses and similar in both tillage systems at high fertiliser rates. The former reduction in crop yield in NT is probably a result of N deficiency during the early stages of crop growth due to a delayed N mineralisation in spring. In other cases, similar crop yields can only be obtained by adding a surplus of 10-40 kg N ha⁻¹ N fertilisation in NT. The need for more N fertilisation may be due to reduced N mineralisation, more elevated denitrification or to reduced root penetration because of the compacted soil structure. In dry regions, the better water status of the soil in NT may result in larger crop yields than in CT at high fertiliser rates, while the N deficiency at low fertiliser rates will result in lower yields in NT. In contrast, under wet climate conditions and on poorly drained soils, NT may cause severe yield reductions due to anaerobic conditions (Anken *et al.*, 2004). Finally, acidification of the surface soil layer of NT may also cause yield reductions compared to CT (Blevins *et al.*, 1983).

1.7 Variability in responses to different tillage systems

There is no consensus on the effect of the different tillage systems on most soil properties, soil processes and/or organic matter pools. Above all, it is the regional climate (temperature conditions, amount of rainfall, rainfall pattern and evapotranspiration) that has most impact on the variability of short-term and long-term responses to different tillage systems (Doran, 1987; Balesdent *et al.*, 2000). For example, it has a large impact on the magnitude of the differences in soil temperature and soil water content between CT and NT.

Secondly, the soil history (prairie, forest, cultivated field) as well as the soil properties plays an important role. In fields cultivated for many generations before differentiation, tillage effects may be less pronounced due to a lower initial microbial biomass and nutrient pools and/or a relatively resilient microbial biomass (Frey *et al.*, 1999; Calderon *et al.*, 2000). Yang and Kay (2001) attributed the differences in C stocks in recently cultivated fields to a slower loss of OM in NT than CT (stocks decreasing both in CT and NT). The differences in fields cultivated for many generations were mainly due to C sequestration (stocks increasing in NT). Differences between tillage systems are less pronounced in fine-textured soils containing montmorillonite because of the SOM protection by sorption on these clay surfaces with large surface and high charge density (Craswell and Waring, 1972; Wander *et al.*, 1998; Collins *et al.*, 2000). Soil physical properties (e.g. sensitivity to structure deterioration) may also play a determining role (Balesdent *et al.*, 2000).

Thirdly, the two tillage systems may not receive equal annual C and N inputs due to different crop yields or to different cropping systems. Under tropical conditions, NT is often combined with a cover crop. In tillage systems with similar C and N inputs, the quantity of C and N restitution has a pronounced effect. Thus a small amount of crop residue will protect the soil surface less in NT than a large amount. So, the cropping systems have a great impact on the differences between CT and NT due to their effects on the quantity, quality and timing of the crop residue restitutions. Other determining factors are the degree of soil cover by living plants, their timing of nutrient requirement and the sensitivity of their root system to soil compaction (Doran, 1987; Benjamin, 1993).

Fourthly, tillage systems have both short- and long-term effects. This means that tillage effects will present temporal variations within a single year as well as over several years (Balesdent *et al.*, 2000). In addition, a new soil equilibrium may be reached after a long period of differentiation.

The large impact of all these factors implies that it is unreliable to extrapolate literature data to agricultural fields with different soil properties, soil history, regional climate or

cropping systems. However, data about the underlying processes of these different responses of C and N dynamics on tillage differentiation between different soils are scarce.

Finally, a remark regarding methodological differences seems warranted. Due to the generally larger bulk densities in NT than CT, C and N stocks have to be calculated for equivalent soil masses. However, many C and N stocks in the literature have been calculated for equivalent soil volumes, which usually overestimate the difference between CT and NT (Balesdent *et al.*, 2000). In addition, the compared soil masses must represent soil volumes that are deep enough to take the whole sphere of influence of the different tillage systems into account. Nutrients and soil processes in NT are more concentrated near the surface so very shallow soil volumes will, once again, overestimate the difference between CT and NT (Wander *et al.*, 1998; Balesdent *et al.*, 2000).

1.8 General objectives questions and hypotheses

The overall objective of this work is:

- (i) to quantify the differences in C and N pools as well as in C and N fluxes in farming conditions between different tillage systems imposed over a long-term (32 years) in Northern France, and
- (ii) to study the effects of climatic conditions, soil structure and biological and physical properties of the soil on the differences in the C and N cycles between those tillage systems.

The work focuses on two extreme tillage systems, i.e. mouldboard ploughing to 20 cm depth (CT) and no-tillage (NT). For some investigations, an intermediate tillage system was also included, namely non-inversion superficial tillage (ST) to 10 cm depth.

The main research questions are indicated on the schematic representation of the C and N cycle in Figure 1.2. Work was focused mainly on the soil C and N stocks, the crop residue decomposition, the dynamics of soil mineral N and the CO₂ and N₂O emissions as these have important agronomical or environmental impacts. Farmers as well as governments require data on the magnitude and determining factors of these pools and fluxes in farming conditions with different tillage systems. One of the originalities of this research lies in the simultaneous analysis of biological and physical characteristics of the soil and in the coupling of water, solute, temperature and C and N dynamics.

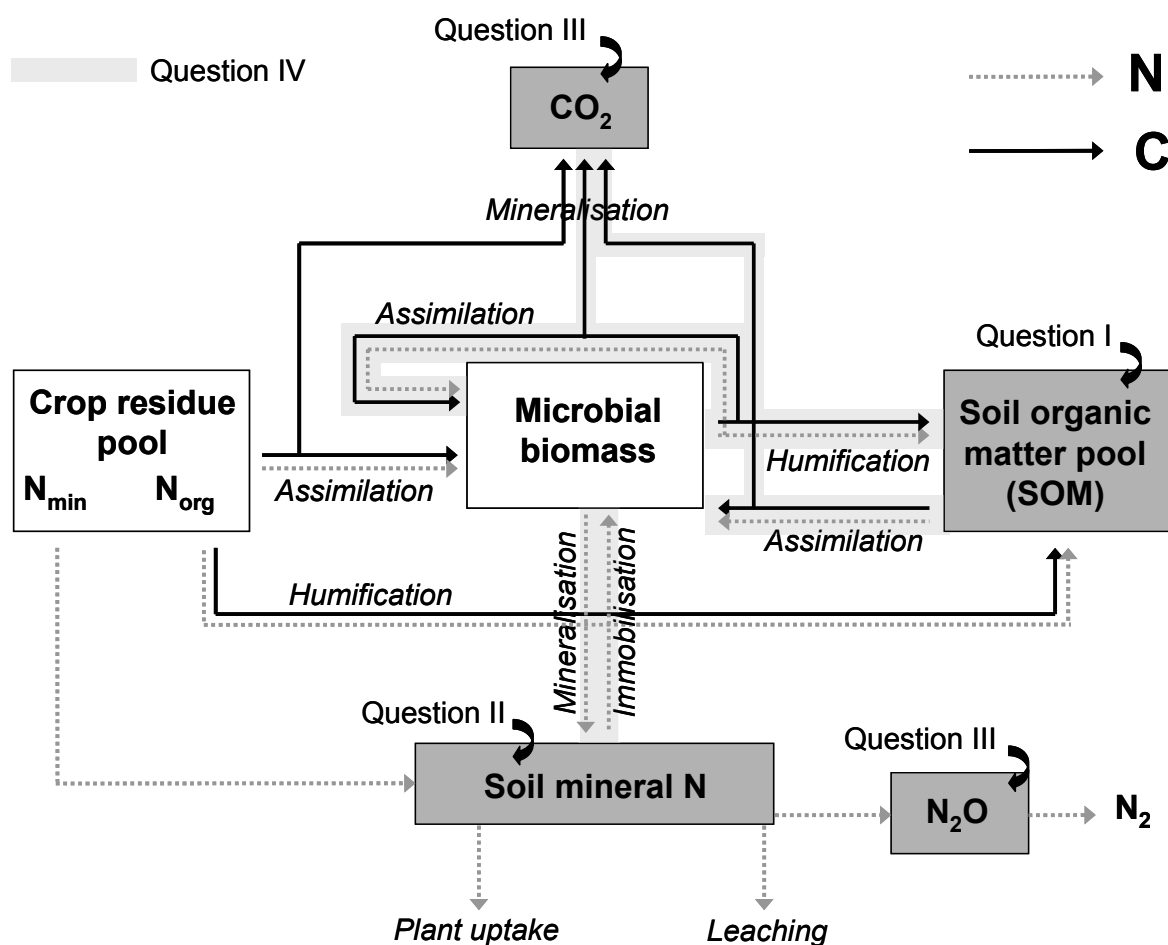


Figure 1.2: Indication of the main research questions on the schematic representation of the C and N cycle. Research question V covers the entire C and N cycles.

1.8.1 General research questions

- I) Is there a difference in the quantity and quality of SOM stocks between conventional tillage (CT) and no-tillage (NT) systems after 32 years of differentiation? How is the SOM distributed in the soil profile?
- II) Does the tillage system (CT, ST or NT) modify the soil mineral N supply (net N mineralisation) and the amount of N leaching?
- III) What are the effects of CT and NT on CO₂ emissions in the absence of plants? Are the N₂O emissions influenced by these tillage systems?
- IV) Does the total and specific potential C and N mineralisation rate of SOM, determined under constant temperature and soil water pressure, differ between CT and NT? At what level of soil structure is organic matter protected against C and N mineralisation in the two tillage systems?

- V) What are the main determining factors governing the difference in C and N fluxes in the two tillage systems in the absence of plants (e.g. SOM localization, SOM potential C and N mineralisation, soil climatic conditions, crop residues decomposition or soil properties)?

1.8.2 Main hypotheses

- I) In NT, the SOM stocks are larger than in CT. In NT, a large proportion of these stocks is located near the soil surface, whereas in CT, they are relatively homogenously distributed throughout the plough layer.
- II) In NT, the soil supplies less mineral N due to less net N mineralisation.
- III) In NT, CO₂ emissions are smaller due to slower C mineralisation. However, N₂O emissions are larger in NT due to the more pronounced anaerobic conditions resulting from a larger percentage of water filled pore space and larger water stable aggregates.
- IV) Differences in C and N stocks and fluxes in CT and NT result from the potential decomposability of the SOM.
- a. The larger C and N stocks and the smaller C and N fluxes in NT are the result of reduced specific potential SOM mineralisation.
 - b. In NT, more SOM is located in water stable aggregates which protect SOM against C and N mineralisation.
 - c. In CT, the protection of SOM differs between the loose and dense structural zones created by tillage.
- V) Differences in C and N stocks and fluxes result from changes in the location and the amounts of SOM and crop residues and from modifications in the soil properties and the soil climatic conditions.
- a. Crop residues are located on the soil surface in NT and incorporated in CT. Surface residues decompose more slowly than incorporated residues due to larger fluctuations of their water content and reduced contact with the soil.
 - b. A larger amount of SOM or crop residues in NT than in CT can counterbalance the effect of their reduced specific potential C and N mineralisation.
 - c. In NT, water evaporation is reduced due to the presence of a surface mulch which modifies the water and solute dynamics. During dry periods, reduced evaporation may induce more favourable conditions for the decomposition of SOM compared to CT.

- d. In CT, the summer soil temperature is more favourable to SOM decomposition due to the absence of a surface mulch.
- e. Long-term tillage systems induce changes in the soil physical, chemical and biological properties which in turn may alter the C and N fluxes in the soil.

1.9 Thesis outline

The introduction chapter presents the reasons for using no-till in cropping systems and the effect of tillage systems on soil physical, chemical and biological properties, based on the existing literature. The state of the art regarding the effects of soil properties on the C and N cycle in soil as affected by tillage is also reviewed. Finally, the objectives and hypotheses are formulated and the thesis outline is described.

Chapter 2 describes the quantification of C and N stocks in CT and NT after 32 years of tillage differentiation. These stock differences are further explored by examining the changes at different levels of structural complexity: (i) primary organic matter fractions (particulate organic matter and 'silt + clay' associated organic matter), (ii) water stable aggregates and (iii) pore size distribution in different structural zones.

Chapter 3 reports on the potential C and N mineralisation in CT and NT and the physical SOM protection induced by both tillage systems. First, total and specific C and N mineralisation in undisrupted fresh soils from long-term CT and NT systems are measured under controlled temperature and moisture conditions. Then, the role of soil structure (ranging from microstructures to the level of the 'in situ' arrangement of different macrostructures) in protecting organic matter against C and N mineralisation is evaluated.

Chapter 4 presents the inorganic N dynamics in farming conditions with different tillage systems. The N mineralisation and N leaching are calculated with the LIXIM model. These processes determine the available mineral N in soil which is important from the agronomical point for the management of N in agrosystems (e.g. N fertilisation, management of the intercrop period).

Chapter 5 focuses on the CO₂ and N₂O emissions occurring in farming conditions under different tillage systems. The dynamics and magnitude of the CO₂ emissions are explained using measurements of soil organic C and mineral N, soil potential for CO₂ emission, climatic data and the decomposition of crop residues.

The C and N fluxes are simulated with the PASTIS model in Chapter 6. Modelling provides more insight into the isolated effects and interactions of the determining factors on C and N dynamics. First, the feasibility of modelling the C and N fluxes in the different tillage

systems is evaluated. Then, the effect of the determining factors on these fluxes (in particular the CO₂ fluxes) is discussed.

Finally, Chapter 7 presents the general conclusions and some recommendations for future work.

1.10 Sampling site

The experimental site is situated at Boigneville in the Parisian Basin in Northern France. This site was set up by the ARVALIS-Institut du Végétal to obtain data about the long-term effects of different tillage systems on crop yield, development of weeds, diseases and parasites, fertilisation requirements and the evolution of organic matter. Three tillage systems were imposed in 1970: conventional tillage (CT), superficial tillage (ST) and no-tillage (NT). The CT system consists of an annual inversion of the 0-20 cm soil layer by mouldboard ploughing. The ST is tilled with a rotary cultivator down to 10 cm depth. No tillage operations are performed in the NT system except for slight soil disturbance (< 5 cm) along the sowing line during sowing. Before 1970, all plots had been mouldboard ploughed to 30 cm depth every year. The site has a maize-wheat rotation. Both crops are present every year. This means that the experimental plots are defined by the present crop and the tillage system.

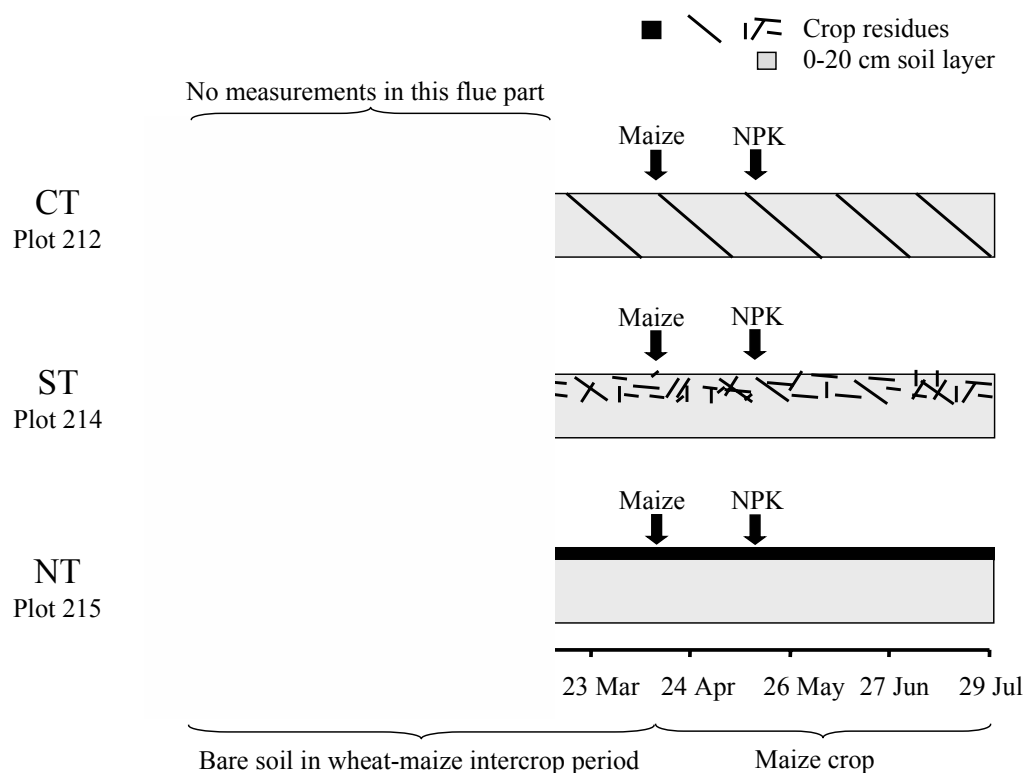
The soil was sampled and measured during two different periods on two different series of plots (Figure 1.3). When the plots followed during period I were in the wheat year, the plots followed during period II were in the maize year. The period I field plots were monitored from March 2003 until August 2003. Maize was sown on April 17th and received 158 kg N ha⁻¹ ammonium-nitrate on May 12th 2003. The plots were irrigated three times with 25 mm of water between the end of June and the end of July. The period II plots were followed from July 31st 2003 until April 13th 2004. Period II covers the entire-maize intercrop period during which the soil remained bare. During period II, the soil was mouldboard ploughed on November 19th in CT and tilled with a rotary cultivator on November 24th in ST. The structural soil profiles of the different plots are presented in Appendix A. The soil properties are presented in the different chapters of this work.

According to the FAO classification, the soil is a Haplic Luvisol. The site is situated on a topsoil derived from loess (70-100 cm deep) with a clayey loam texture overlying a calcareous substrate. The slope is negligible. The soils drain very rapidly and their sensitivity to soil compaction by mechanical forces is very low. In addition, the soils tend to fissure under frost action and drying/wetting cycles. There are no problems with water and wind erosion. Boigneville is characterised by a climate with oceanic influences. The long-term average temperature is 10.8 °C and the annual precipitation and potential evapotranspiration

(PET) are 650 mm and 730 mm, respectively. The late autumn is wet, the early spring rather dry whereas the summer is characterised by thunderstorms. The amount of PET surpasses that of rainfall from March until September with the highest deficit in July.

After 4 and 8 years of differentiation respectively, Langlet and Rémy (1976) and Boissgonnier (1982) observed that, despite only slight changes in the total C and N stocks in the three tillage systems, the distribution of these stocks within the ancient plough layer had clearly been changed. After 28 years of differentiation, the C stocks calculated for the soil mass corresponding to the old plough layer before differentiation (3900 kg dry soil ha⁻¹) were 42.71 ± 0.62 tons C ha⁻¹ for CT, 45.56 ± 1.13 tons C ha⁻¹ for ST and 44.88 ± 2.27 tons C ha⁻¹ for NT (B. Mary, personal communication). Regarding crops, Langlet and Rémy (1976) observed after 2-4 years of differentiation similar cumulated N uptake in NT and CT at wheat harvest despite the smaller N uptake in spring in NT. Between 1976 and 2003, maize plants developed more slowly in the early growth stages in NT but the final crop yields of maize and wheat were similar in all three tillage systems (Labreuche *et al.*, 2003).

Plots followed during period I (March 2003 - July 2003)



Plots followed during period II (August 2003 - April 2004)

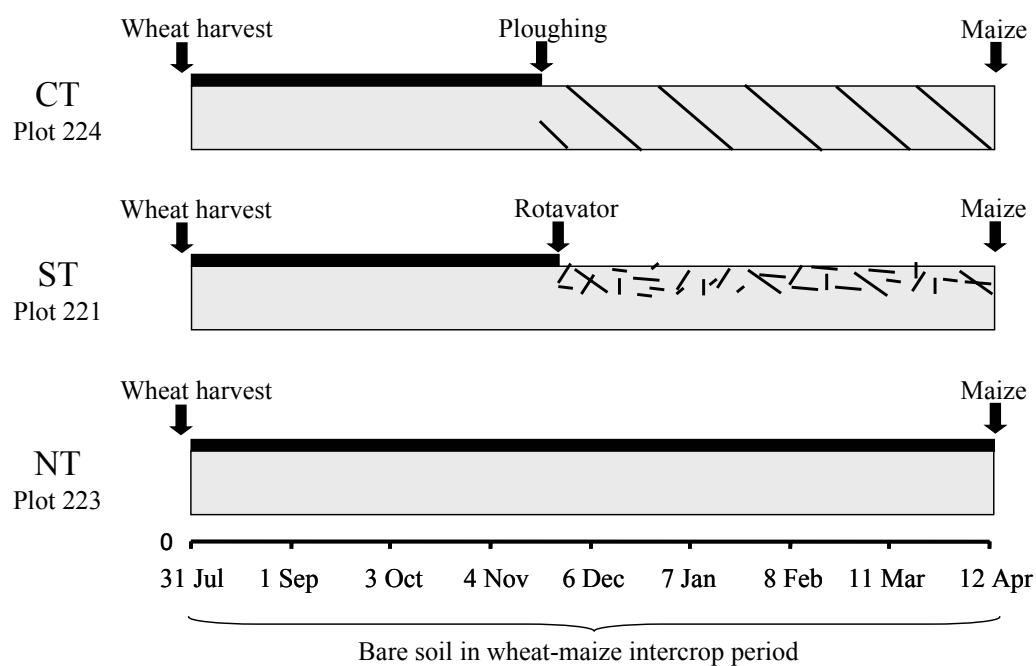


Figure 1.3: Schematic representation of the field plots followed during periods I and II.

CHAPTER 2: CARBON AND NITROGEN STOCKS IN RELATION TO ORGANIC MATTER FRACTIONS, AGGREGATION AND PORE SIZE DISTRIBUTION IN NO-TILLAGE AND CONVENTIONAL TILLAGE

2.1 Introduction

The long-term practice of no-tillage (NT) generally entails larger C and N stocks than conventional tillage (CT) due to better preservation of the organic matter originally present in the soil and/or less mineralisation of that recently added. However, the magnitude of the difference in organic matter (OM) stocks between the two systems depends on factors such as texture, parent material, regional climate, SOM content, soil heritage, cropping system and physical properties (Doran, 1987; Balesdent *et al.*, 2000; Collins *et al.*, 2000).

Physical protection of OM by the soil structure is often mentioned as an important contributor to the slower mineralisation of OM in NT compared to CT (Balesdent *et al.*, 2000). However, it is important to consider all levels of structural complexity in studies of the effect of physical protection on C and N storage in different tillage systems (Christensen, 2001). The primary level of structural complexity comprises soil minerals and uncomplexed OM. This uncomplexed OM is either mineral-associated OM or particulate organic matter (POM). The latter is uncomplexed organic matter larger than 53 μm and has an intermediate turnover time between residues (> 2 mm) and mineral-associated OM.

Mineral-associated OM and POM, together with micro-organisms and microbial products, are the constituents of aggregates (i.e. the secondary level of structural complexity). These aggregates are reported to slow down the decomposition of mineral-associated OM and POM (Wander *et al.*, 1998; Christensen, 2001). No-tillage has been shown to increase the amount of macroaggregates (Balesdent *et al.*, 2000; Six *et al.*, 2004) due to a myriad of factors that have been reviewed exhaustively in these publications. On the other hand, aggregate formation under CT may be increased by the flush of microbial activity after the disruption of aggregates or incorporation of residues.

The intact soil structure forms the highest level of structural complexity (porosity, pore size distribution and localization of organic matter). This soil structure is the result of the

balance between compaction (by machine traffic and soil weight), agglomeration of aggregates (by moderate compaction, climate and/or fauna), fragmentation (by climate, fauna and/or tillage) and displacement (by tillage) of soil (Roger-Estrade *et al.*, 2000). The plough layer of mouldboard ploughing systems has an extremely heterogeneous soil structure with large soil clods with dense or loose structure, fine soil, clusters of crop residues situated between the furrows and large voids (Staricka *et al.*, 1991; Roger-Estrade *et al.*, 2004). In no-tillage systems, residues are not incorporated into the soil and soil is not displaced and fragmented during tillage which results in a less heterogeneous soil structure compared to CT. In addition to aggregation, soil porosity may also influence the turnover of organic matter by controlling the activity of micro-organisms (Elliott and Coleman, 1988; Hassink, 1992; Ladd *et al.*, 1993; Strong *et al.*, 2004). The location of fresh organic matter in the soil profile also has a strong influence on its decomposition. Fresh organic matter will probably decompose more rapidly when incorporated in the soil (as is the case in conventionally tilled systems) due to fewer fluctuations in water content and better contact with the soil leading to a better inorganic N availability (Balesdent *et al.*, 2000). However, the magnitude of this effect depends on the local climate (Wander and Yang, 2000). Conversely, the decomposition of incorporated fresh OM may also be slowed down by a more intimate contact with silt and clay, in comparison with residues left at the soil surface. The interpretation of these effects of residue localization and the action of tillage may be complicated by differences in the annual restitutions of crop residues between the different tillage systems in many experiments.

Our aim was to quantify differences in total C and N stocks after 32 years of tillage differentiation (i.e. NT and CT). The study was specifically confined to CT and NT plots that had been cultivated for many generations before differentiation and in which the annual amounts of returned crop residues were similar (around 3 ± 1 ton C ha⁻¹). The second aim of the paper was to test the hypothesis that the differences in C and N stocks observed were related to physical protection induced at different levels of structural complexity. Besides the distribution and composition of uncomplexed OM (mineral-associated organic matter, particulate organic matter and free residues (> 2 mm)), we studied the distribution and composition of slaking-resistant aggregates and the porosity and pore size distribution. The soil sampling was done in different structural zones in CT and NT in order to consider homogenous structural zones in the study of physical protection by soil structure in CT and NT.

2.2 Materials and methods

2.2.1 Study site

The experimental site was located at Boigneville (Northern France, 48°33'N, 2°33'E) on soil developed on loess (Haplic Luvisol). The average annual temperature and precipitation were 10.8°C and 650 mm, respectively. Before 1970, the fields had been under cultivation for many generations and had been tilled to 30 cm depth. The soil management systems were differentiated in 1970, and the experimental plots (8 x 50 m) either received no tillage (NT) or conventional tillage (CT). Over the last 32 years, the ploughing depth in the tilled plot has been approximately 20 cm (mouldboard ploughing with three blades) whereas tillage was totally suppressed in NT. Since 1970 these plots have been cultivated with a maize-wheat rotation with similar restitutions of residues in both management systems. The residues in CT remain on the soil surface after harvest until incorporation by tillage whereas those in NT always remain on the soil surface. The timing of the tillage, sowing and harvesting operations are presented in Table 2.1.

Table 2.1: Timing of tillage, harvesting and sowing operations

Timing	Conventional tillage	No-tillage
October	Maize harvest	Maize harvest
October	Mouldboard plough (1 pass)	-
October	Rotary harrow combined with wheat sowing	Direct sowing of wheat
July	Wheat harvest	Wheat harvest
November	Mouldboard plough (1 pass)	-
April	2 passes of rotating harrow	-
April	Maize sowing	Direct sowing of maize

During the fallow period between maize and wheat in March 2003 (about 4 months after mouldboard ploughing in CT), a structural soil profile was described (see Appendix A) according to the method published by Roger-Estrade *et al.* (2004). The former plough layer of the CT treatment consisted of a recent plough layer (0-20 cm layer) with an old plough layer beneath it. Two structural zones were distinguished within the recent plough layer: zones with loose structure and eye-visible structural porosity (CT_{Loose}) and zones with dense clods of

compacted structure (CT_{Dense}). The former plough layer of the NT treatment was horizontally homogeneous and contained two distinct vertical layers: a 0-5 cm layer with a large amount of organic matter and a highly fragmented structure (NT₀₋₅), and a 5-20 cm layer with a massive structure (NT₅₋₂₀). A mulch layer was present on the NT soil surface whereas in CT the residues were located between adjacent furrows in the recent plough layer.

2.2.2 Determination of C and N stocks

The bulk density of the 0-5 cm soil layer of NT was measured with a rubber balloon density meter (Nardeaux-Humisol, Saint Avertin, France) at 4 different locations within the plot. At deeper soil layers in the NT plot, the bulk density was measured on undisturbed soil cores sampled with 252 cm³ steel cylinders (6.1 cm high and 7.2 cm dia.) in the 5-11, 11-17, 17-23 and 23-27 cm soil layers (3 replicates per soil layer). The same procedure was followed for the 20-28 cm soil layer of CT whereas undisturbed soil cores were taken from both the loose and dense soil zones in the 0-20 cm soil layer. Any surface residues were removed before soil sampling and residues between furrows were avoided. The dry weight of each soil sample (cores and soil density meter) was measured after drying at 105°C. The samples were then crushed and their C and N concentrations determined using an NA 1500 elemental analyser (Fisons, Milan, Italy).

2.2.3 Distribution of soil pore classes

Pore size distributions were calculated using water retention curves and the law of Jurin. This law calculates the largest effective pore neck diameter (μm) of the pores filled with water at a given soil water pressure. The volume of a pore class is then obtained from the difference in volumetric water contents at the two different soil water pressures corresponding to the upper and lower effective pore neck diameter of the pore class. The water retention of an undisturbed soil sample was determined by measuring the soil water content at equilibrium conditions under a succession of known pressure heads: 0.1, 1, 10 and 100 kPa. The ‘hanging water-column’ and ‘low pressure membrane’ were used for the pressure ranges 0-10 kPa and 10-300 kPa respectively (Kabat and Beekma, 1994). Hassink (1992) observed that pore size distribution was correlated with soil texture. The texture of the four soil zones in our study was similar (Table 2.2), so the differences in pore size distribution were attributed to different tillage management practices.

Table 2.2: Main soil characteristics of the different structural soil zones in the conventional tillage (CT) and no-tillage (NT) plots at Boigneville.

		Conventional tillage		No-tillage	
		CT _{Loose}	CT _{Dense}	NT ₀₋₅	NT ₅₋₂₀
Clay (< 2 μm)	g 100 g ⁻¹	21.6	22.1	18.8	20.8
Silt (2-50 μm)	g 100 g ⁻¹	69.5	69.7	69.9	67.9
Sand (> 50 μm)	g 100 g ⁻¹	8.9	8.3	11.3	11.3
pH (water)	-	6.1	5.9	4.8	6.0
CaCO ₃	g 100 g ⁻¹	0.1	0.1	< 0.1	0.1
CEC	cmol _c kg ⁻¹	11.6	11.8	12.3	11.0
Organic C	g 100 g ⁻¹	1.10	1.09	2.46	0.94
Organic N	g 100 g ⁻¹	0.115	0.115	0.212	0.103
C to N ratio	-	9.6	9.5	11.6	9.1

2.2.4 Separation of slaking-resistant aggregate classes and different primary soil fractions

Soil was sampled from the above described different structural zones: i.e. the loose (CT_{Loose}) and dense (CT_{Dense}) structural zones of the 0-20 cm recent plough layer of CT and the 0-5 cm (NT₀₋₅) and 5-20 cm (NT₅₋₂₀) cm layers of NT. Soil material was collected at different places from the structural soil profile and a composite sample was constituted for each structural zone. Measurements during previous years at other locations within the experimental plots (results not shown) enabled us to verify that the soil collected for this experiment was representative of the different structural zones of the entire experimental plots. Residues in the surface mulch of NT or in the furrows of CT were avoided while sampling. Fresh soil samples were gently passed through a 12.5 mm sieve, stored for a short period at 4°C and then air dried at 30°C. The characteristics of the four soil structural zones are presented in Table 2.2.

The slaking-resistant aggregates were separated according to size using a method adapted from Elliott (1986) (see also Figure 2.1). The dry soil (4 x 30 g) was first disposed on a 2 mm sieve and submerged in water for 5 minutes to allow slaking and floating material was removed (free residues > 2 mm). Large macroaggregates (> 2000 μm), small macroaggregates (250 to 2000 μm), microaggregates (53 to 250 μm) and a ‘silt + clay’ sized fraction (< 53 μm) were separated by wet sieving. For each wet-sieving step, the sieve was moved up and down 50 times for 2 minutes. The soil material and water passing through the sieve was poured on a smaller-sized sieve and the sieving procedure was repeated. A subsample was taken from the

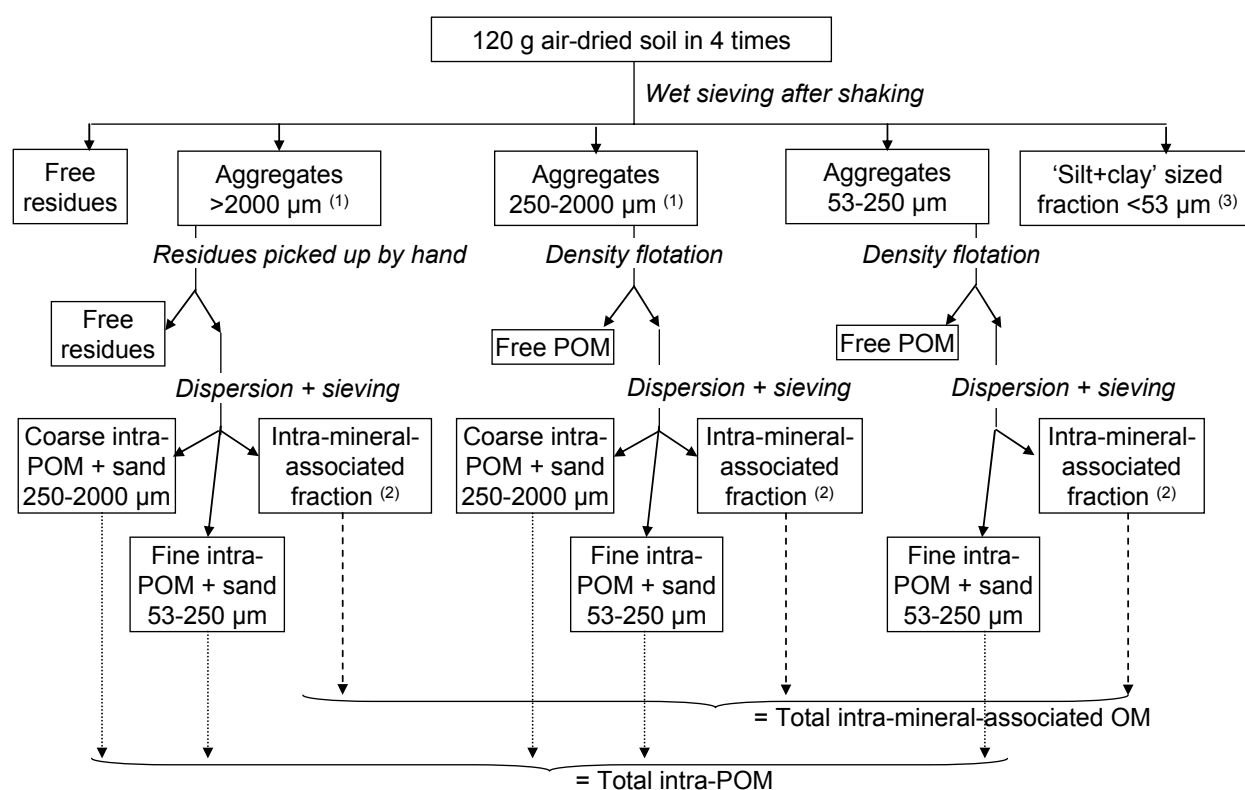
collected soil solution that passed through the 53 μm sieve ('silt + clay' sized fraction). This subsample, the soil material retained on the different sieves and the floating OM were oven-dried at 65°C, weighed and stored at room temperature.

Microaggregates protected within the small and large macroaggregates were isolated by a method adapted from Six *et al.* (2000a) using a microaggregate isolator that completely breaks up the macroaggregates but minimised the disintegration of microaggregates within these macroaggregates. The macroaggregates (5-8 g) were placed on a 250 μm sieve, immersed in de-ionised water and shaken with 125 2-mm glass beads until complete disruption of all the macroaggregates while a continuous, steady flow of water flushed the microaggregates directly through the 250 μm onto a 53 μm sieve in order to avoid further break-up of the microaggregates by the glass beads. The material present on the 53 μm sieve after complete disruption of the macroaggregates, was manually wet-sieved following the procedure used for aggregate separation. The material remaining on the 250 μm (sand and coarse POM) and 53 μm sieves (microaggregates and fine POM) was dried at 65°C and stored at room temperature. The quantities of large macroaggregates (> 2000 μm) produced by wet-sieving of the CT and NT₅₋₂₀ samples were too small to permit microaggregate isolation.

Free particulate organic matter (free-POM) and aggregate-associated OM (Figure 2.1) of the small macroaggregates, microaggregates and microaggregates within macroaggregates, were separated by density flotation with 1.85 g cm⁻³ sodium polytungstate using the procedure proposed by Six *et al.* (1998). Sodium polytungstate was recycled as described by Six *et al.* (1999b) to avoid cross contamination of C. Free residues (> 2 mm) recovered with the large macroaggregates were removed by hand.

The coarse (250 to 2000 μm) and fine (53 to 250 μm) POM plus sand occluded within aggregates (i.e. coarse and fine intra-POM) were isolated from the mineral-associated fraction within aggregates by shaking soil samples of all aggregate classes > 53 μm for 18 hours with 0.5% hexametaphosphate (Figure 2.1). After shaking, the dispersed samples were poured through 250 and 53 μm sieves. Sand plus intra-POM fractions were rinsed with de-ionised water, dried at 65°C and stored at room temperature. Balesdent (1996) observed that the amount of C and N associated with the mineral sand fraction is negligible at the Boigneville site as the mineral sand fraction consists mainly of quartz. This means that all the C and N of the POM plus sand fractions can be considered to be totally associated with POM and that the 'silt + clay' associated OM (< 53 μm) can be called mineral-associated OM.

All separation procedures were repeated 3 times for each sample of the 4 soil zones. Total C and N concentrations of aggregates, POM, residues and the mineral-associated fractions were determined by dry combustion using an NA 1500 elemental analyser (Fisons, Milan, Italy).



(1) For these aggregate classes, also a separation of microaggregates within macroaggregates was done.

(2) This material was not fractionated but rather calculated by difference between total C content and C content of subfractions

(3) This fraction is also called the free mineral-associated fraction.

Figure 2.1: Graphical representation of the fractionation scheme (adapted from De Gryze et al. (2004))

2.2.5 Calculations

The weights and C (N) concentrations of the different slaking-resistant aggregate classes and the C (N) concentrations of the free residues, free POM and intra-POM were used to calculate the following values:

Mean weight diameter (MWD) of the slaking-resistant aggregates:

$$\text{MWD} = \sum (\% \text{ weight aggregate class} * \text{mean diameter aggregate class})$$

Amount of free mineral-associated C (N):

$$\text{Free mineral-associated C (N)} = \text{C (N) in free 'silt + clay' sized fraction (< 53 } \mu\text{m)}$$

Amount of mineral-associated C (N) occluded within aggregates:

$$\text{Intra-mineral-associated C (N)} = [\text{Aggregate-C (-N)}] - [\text{intra-POM-C (-N)}]$$

C (N) concentrations expressed on a sandfree basis:

$$C_{\text{Sandfree}} (N) = \frac{C (N)}{[1 - (\text{weight proportion} > 53 \mu\text{m})_{\text{aggregate class}}]}$$

Proportion of microaggregates within macroaggregates per sandfree aggregate:

$$\% \text{ mM} = \frac{[[\text{microaggregate weight}] - [53\text{-}250 \mu\text{m sand weight}]] \times 100}{[\text{macroaggregate weight}] - [> 250 \mu\text{m sand weight}]}$$

2.2.6 Statistical analysis

The multiple means comparisons of variables obtained for the four structural soil zones (CT_{Loose} , CT_{Dense} , NT_{0-5} and NT_{5-20}) were done with the ANOVA procedure and the Student-Newman-Keuls test (SAS Institute, 2001). On the other hand, the means comparisons between CT and NT (variables expressed for the equivalent plough layer) were performed using student t tests (Bowker & Liberman, 1965). For both analyses, the level of significance was fixed at $\alpha = 0.05$. The statistical results are not indicated in the figures and tables because of the large amount of data. However, differences mentioned in the text are all significant.

2.3 Results

2.3.1 C and N stocks

The C and N stocks were calculated for the portion of the soil profile corresponding to the old plough layer before differentiation and considering 3900 t dry soil per hectare, which corresponds to the average mass of soil in this layer. This was done by taking the C and N concentrations, bulk density and the width of the different soil layers into account. Note that correct measurements of the bulk density of the different soil layers were extremely important to provide good estimates of the C and N stocks in the CT and NT systems. Due to a larger mean bulk density for NT than CT, the mass of 3900 t dry soil per hectare corresponded to a 0-27.7 cm layer in CT versus a 0-26.8 cm layer in NT. Total C and N stocks were significantly larger for the NT plot than for the CT plot (Table 2.3) after 32 years of differentiation, but the differences were not large.

Because the soil below 20 cm (depth of the recent plough layer) had not been tilled for 32 years, the C and N concentrations in that soil layer were similar under both CT and NT (Table 2.3). Therefore, we concentrated on the recent plough layer (0-20 cm layer) to explain the differences in total C and N stocks observed between the two tillage systems. The calculations

for this layer were based on an equivalent soil mass for both tillage systems and took into account the bulk density and volume of the respective structural soil zones (CT_{Loose} , CT_{Dense} , NT_{0-5} and NT_{5-20}). CT_{Loose} and CT_{Dense} constituted 54.3% and 45.7% of the volume of the CT plough layer respectively. The soil mass of the 0-20 cm recent plough layer of CT (2640 t dry soil per hectare) corresponded to a 0-18.7 cm soil layer in NT. The C and N stocks for the whole equivalent plough layer (0-20 cm) were 29.0 ± 0.4 and 33.4 ± 0.7 tons C ha⁻¹ and 3.03 ± 0.04 and 3.33 ± 0.07 tons N ha⁻¹ for CT and NT respectively.

Table 2.3: C and N stocks in the soil calculated for an equivalent mass of 3900 t dry soil ha⁻¹ (0-27.7 cm soil layer in CT and 0-26.8 cm soil layer in NT) and C and N concentrations in the different soil layers of the conventional tillage (CT) and no-tillage (NT) plots at Boigneville.

Treatment	Soil layer	Bulk density	Organic C	Organic N
	cm	g cm ⁻³	tons C ha ⁻¹	tons N ha ⁻¹
CT	0-27.7	1.41 ¹	37.9 ± 2.7 ^{2,3}	4.01 ± 0.26 ³
NT	0-26.8	1.46 ¹	43.5 ± 1.8 ³	4.40 ± 0.12 ³
	cm	g cm ⁻³	g 100 g ⁻¹	g 100 g ⁻¹
CT	0-20	1.32 ± 0.08	1.09 ± 0.05	0.111 ± 0.004
	20-28	1.64 ± 0.02	0.73 ± 0.07	0.086 ± 0.007
NT	0-5	1.18 ± 0.05	2.37 ± 0.29	0.211 ± 0.022
	5-11	1.43 ± 0.03	1.18 ± 0.10	0.121 ± 0.006
	11-17	1.55 ± 0.03	0.84 ± 0.07	0.091 ± 0.005
	17-23	1.57 ± 0.01	0.76 ± 0.03	0.084 ± 0.001
	23-27	1.52 ± 0.02	0.75 ± 0.02	0.082 ± 0.002

¹ recalculated taking into account bulk density and thickness of the different soil layers

² $\pm$ standard error

³ The C and N stocks are significant different between CT and NT ($P < 0.05$)

2.3.2 C and N stocks in the different soil fractions

Most of the difference in C and N stocks between NT and CT was due to the larger amounts of mineral-associated OM and POM occluded within aggregates under NT (Figure 2.2). In contrast to larger total C and N stocks in NT, lower amounts of C and N were found in the free mineral-associated OM of NT than in CT. More C and N were present in the aggregates

in NT but less C and N in the total free OM. Although most C was present as total mineral-associated C, 58% of the difference in C stock between CT and NT was due to a difference in total POM-C, and only 34% to mineral-associated C and 8% to residue-C. Mineral-associated N and POM-N each accounted for about 50% of the observed difference in total N stock between NT and CT whereas only 4 % was due to residue-N.

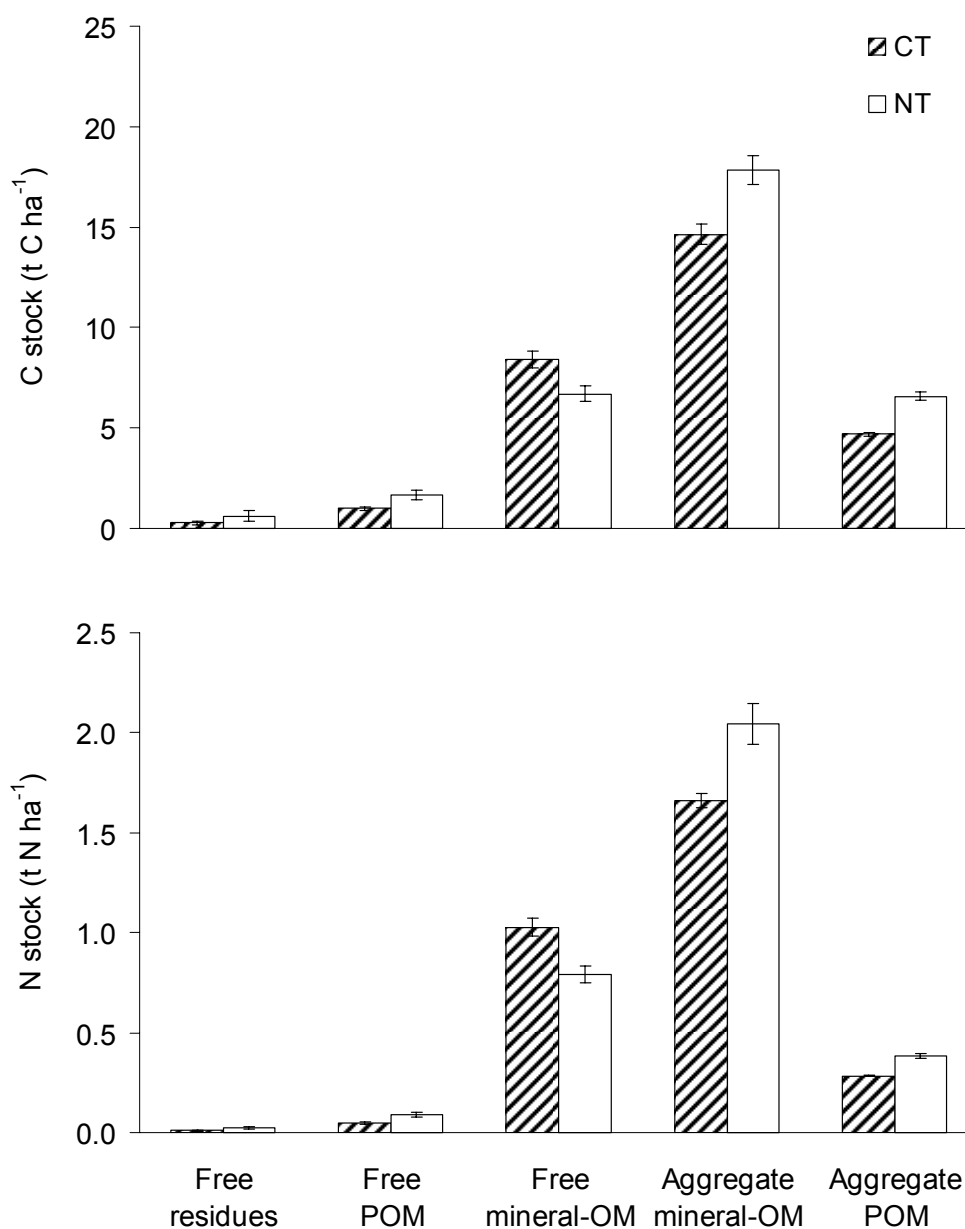


Figure 2.2: C and N stocks in the different soil fractions for the equivalent plough layer of the conventionally tilled (CT) and non-tilled (NT) plots (bars correspond to standard errors)

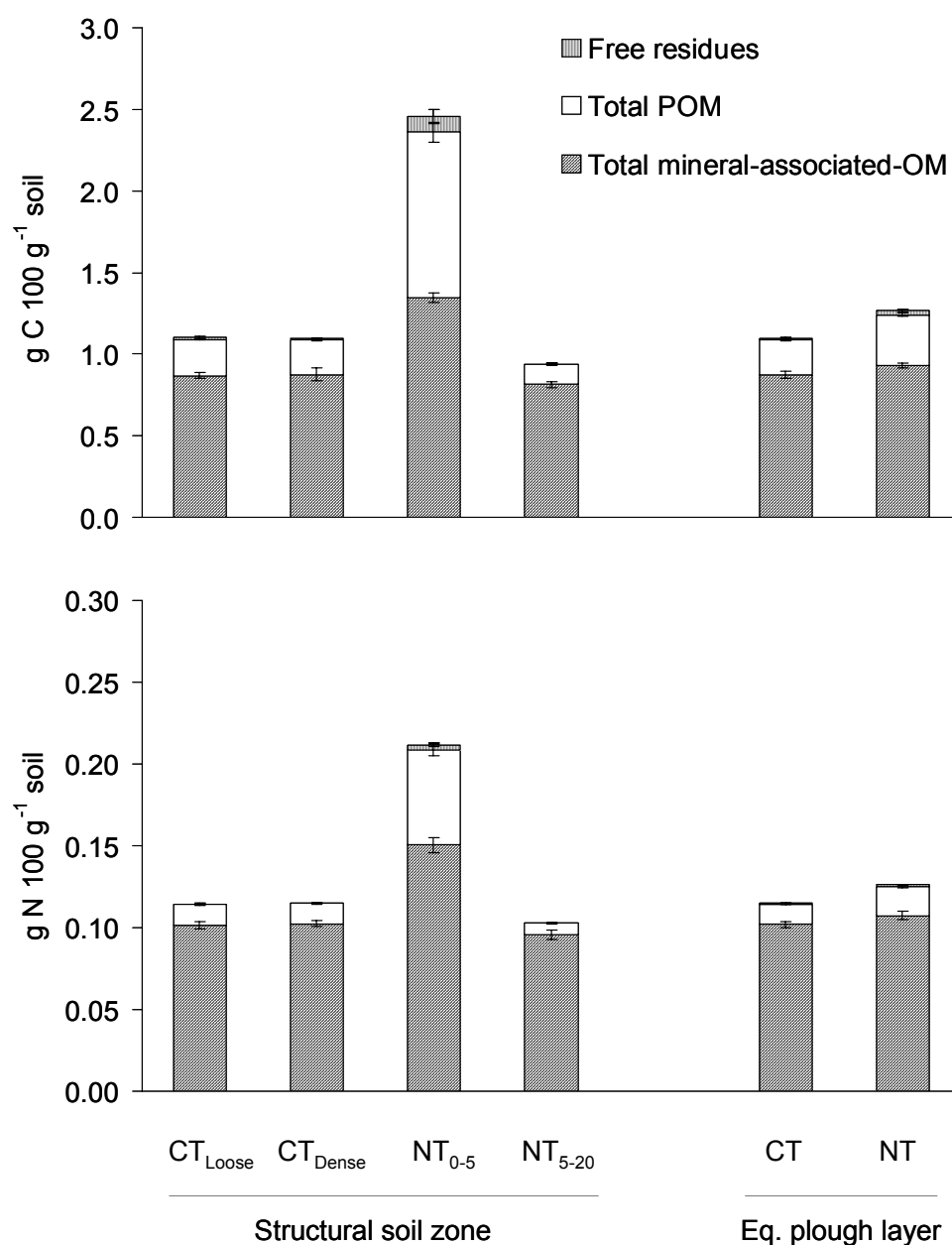


Figure 2.3: C and N concentrations in free residues, particulate OM (POM) and mineral-associated OM of the soil from the different soil zones and the equivalent plough layer of conventionally tilled (CT) and non-tilled (NT) plots (bars correspond to standard errors)

2.3.3 Mineral-associated OM and POM in the different soil zones

The NT₀₋₅ soil zone presented considerably larger amounts of C and N in mineral-associated OM, POM and free residues than the loose and dense soil zones under CT. However, the smallest amounts of C and N in POM were found in the NT₅₋₂₀ soil zone (Figure 2.3). The

differences in total C concentration between the four soil zones were mainly due to differences in total POM concentrations and less to differences in mineral-associated OM whereas the differences in mineral-associated N between the soil zones were comparable to those for POM-N. In general, the proportion of total N due to mineral-associated N in all four soil zones was larger than the proportion of total C due to mineral-associated C.

2.3.4 Distribution and composition of slaking-resistant aggregates in the different soil zones

The mean diameter of slaking-resistant aggregates was notably larger in the equivalent plough layer under NT (0.501 mm) than under CT (0.281 mm). The mean weight diameter was largest for the NT₀₋₅ soil zone (0.730 ± 0.034 mm), intermediate for the NT₅₋₂₀ soil zone (0.439 ± 0.013 mm) and smallest for loose (0.268 ± 0.004 mm) and dense soil zones (0.297 ± 0.025 mm) in the ploughed plot. A substantial amount of large slaking-resistant macroaggregates (> 2000 μm) was only present in NT₀₋₅ (Figure 2.4). Both layers of the no-till plot contained similar amounts of small macroaggregates (250-2000 μm) and more than in the conventionally tilled plot.

Large (> 2000 μm) and small (250-2000 μm) macroaggregates held considerably more C and N in the NT₀₋₅ zone compared to the three other soil zones (Figure 2.4). Differences in C and N in the microaggregates (53-250 μm) and free mineral-associated OM (< 53 μm) were less pronounced. The amounts of C and N in the free POM and free residues were significantly larger under NT₀₋₅ than under the three other soil zones. The most interesting result is that the larger amount of C and N in the equivalent plough layer of NT was almost exclusively due to the larger amounts of C and N in aggregates > 250 μm .

The C and N concentrations calculated on a sand-free aggregates basis increased with aggregate size (Figure 2.5). The concentrations of C and N in intra-POM accounted for most of this increase. For all aggregate size classes, the total C and N concentrations and the C and N concentrations in both the fine and coarse intra-POM of the sandfree aggregates were much larger under NT₀₋₅ compared to CT and NT₅₋₂₀. In contrast, the C and N concentrations of the intra-POM under NT₅₋₂₀ were lower than those under CT but the differences were considerably smaller between NT₅₋₂₀ and CT than between NT₀₋₅ and CT. The differences in C and N concentrations of the intra-mineral-associated C and N between the different soil zones were not significant for the large macroaggregates but followed the same trends as intra-POM for the small macroaggregates and microaggregates.

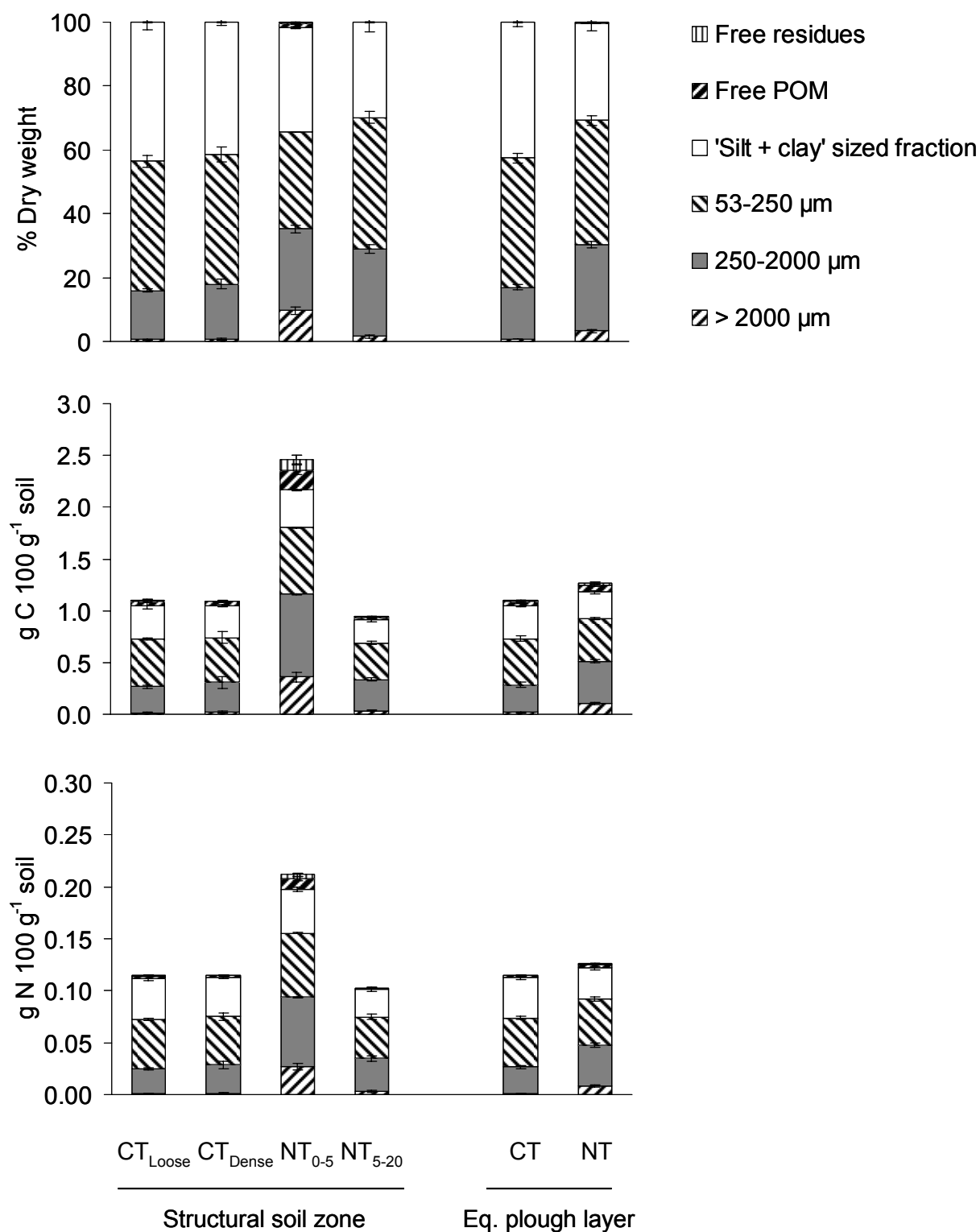


Figure 2.4: Aggregate-size distribution and C and N concentrations in the different aggregate classes in the different structural soil zones and the equivalent plough layer of the conventionally tilled (CT) and non-tilled (NT) plots (bars correspond to standard errors).

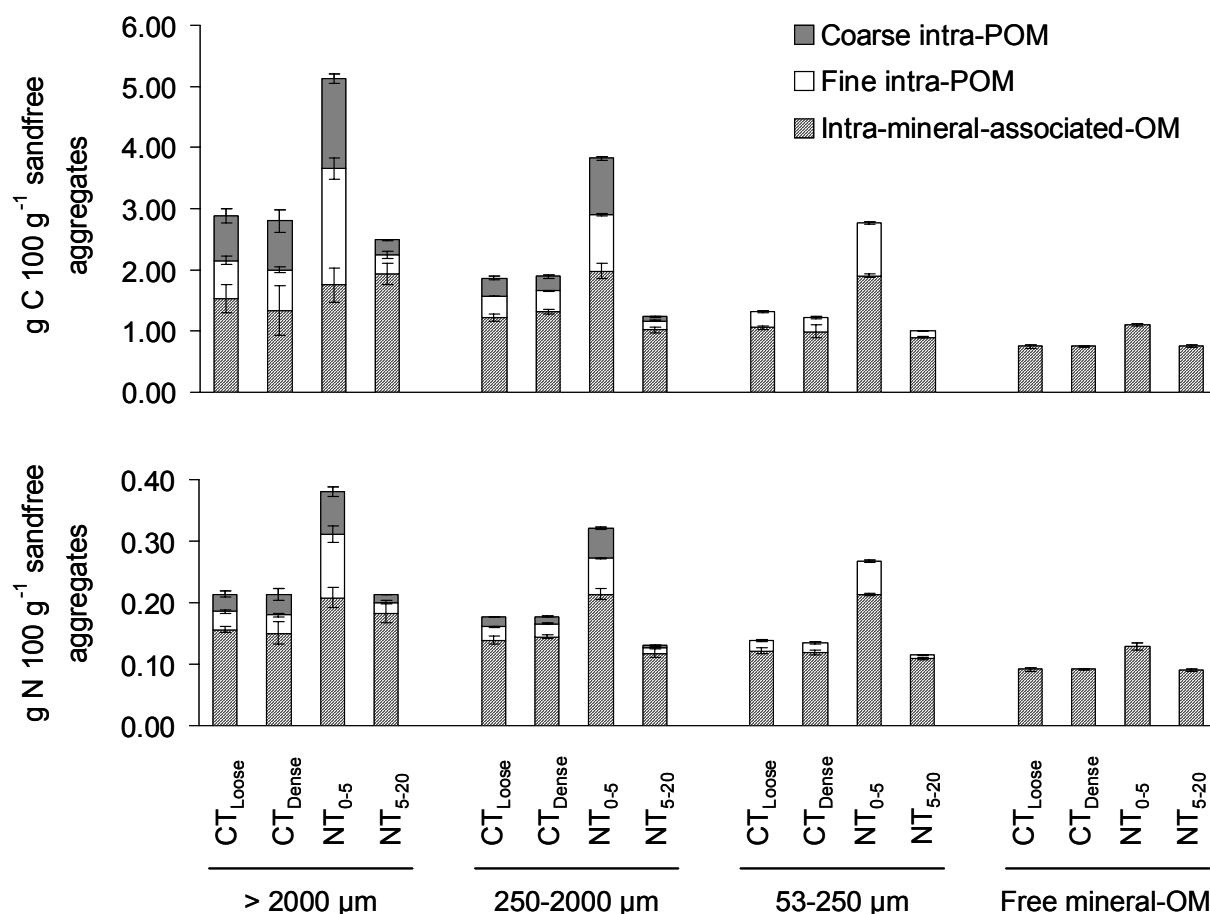


Figure 2.5: C and N concentrations of fine and coarse intra-particulate organic matter (fine and coarse intra-POM) and mineral-associated OM (intra-mineral-associated-OM) for the different sandfree aggregate classes separated from the different soil zones of conventionally tilled (CT) and non-tilled (NT) plots (bars correspond to standard errors)

Table 2.4: Fine intra-POM-C to coarse intra-POM-C ratios in the small (250-2000 μm) and large (> 2000 μm) macroaggregates of the different soil zones in the conventionally tilled (CT) and non-tilled (NT) plots.

	CT _{Loose}	CT _{Dense}	NT ₀₋₅	NT ₅₋₂₀
250-2000 μm	1.21 ± 0.13 ¹	1.51 ± 0.15	1.02 ± 0.02	2.34 ± 0.35
> 2000 μm	0.87 ± 0.12	0.86 ± 0.21	1.29 ± 0.14	1.22 ± 0.16
All (> 250 μm)	1.18 ± 0.11	1.40 ± 0.06	1.11 ± 0.06	2.14 ± 0.30

¹ ± standard error

The ratios between fine intra-POM-C and coarse intra-POM-C in the large macroaggregates were higher in the two NT zones than in the CT soil zones (Table 2.4). The largest value for small macroaggregates was found in the NT₅₋₂₀ samples. When all macroaggregates (> 250 µm) were considered, the ratio of fine intra-POM-C to coarse intra-POM-C was significantly higher in NT₅₋₂₀ than in NT₀₋₅, CT_{Loose} and CT_{Dense} while the latter three zones presented similar values. The same trends were observed for intra-POM-N (results not presented).

Table 2.5: Weight percentage of microaggregates within the small (250-2000 µm) and large (> 2000 µm) macroaggregates and amounts of C and N in macroaggregates found within the small (250-2000 µm) and large (> 2000 µm) macroaggregates of the different soil zones in the conventionally tilled (CT) and non-tilled (NT) plots.

	CT _{Loose}	CT _{Dense}	NT ₀₋₅	NT ₅₋₂₀
	% weight of sandfree aggregate			
250-2000 µm	56.0 ± 1.7 ¹	56.9 ± 0.4	46.2 ± 0.4	60.5 ± 2.0
> 2000 µm	ND	ND	36.3 ± 1.6	ND
	% weight of soil			
250-2000 µm	8.9 ± 0.6	10.1 ± 0.1	12.7 ± 0.7	17.4 ± 0.7
> 2000 µm	ND	ND	4.7 ± 0.5	ND
	g C 100 g ⁻¹ soil			
250-2000 µm	0.16 ± 0.00 ¹	0.18 ± 0.02	0.43 ± 0.04	0.21 ± 0.01
> 2000 µm	ND	ND	0.19 ± 0.02	ND
	g N 100 g ⁻¹ soil			
250-2000 µm	0.015 ± 0.000	0.017 ± 0.002	0.036 ± 0.003	0.022 ± 0.000
> 2000 µm	ND	ND	0.014 ± 0.001	ND

¹ ± standard error

ND: not determined

The proportion of microaggregates within sandfree macroaggregates was highest for NT₅₋₂₀, intermediate for CT and lowest for NT₀₋₅ (Table 2.5). However, both NT soil zones contained more microaggregates within macroaggregates than CT on a whole soil basis. The amounts of micro- within large macroaggregates were presumed negligible for NT₅₋₂₀ and CT due to the extremely small amounts of large macroaggregates in these soil zones. The amounts of C and N in the microaggregates within macroaggregates on a whole soil basis were threefold higher for NT₀₋₅ than for the other soil zones (Table 2.5). The highest and

lowest C and N concentrations per weight of sandfree macroaggregate were found in the microaggregates within small macroaggregates of NT₀₋₅ and NT₅₋₂₀ respectively (Figure 2.6). The C and N concentrations of POM in micro- within macroaggregates were respectively 2 and 4 times larger for NT₀₋₅ compared to CT and NT₅₋₂₀ whereas smaller differences were observed between NT₀₋₅ and the 3 other soil zones for the mineral-associated C and N within the microaggregates within macroaggregates. This mineral-associated fraction contained 53-83% of the C and 67-90% of the N of microaggregates within macroaggregates, the lowest and highest values being obtained for NT₀₋₅ and NT₅₋₂₀ respectively. The C and N concentrations in the microaggregates within macroaggregates of the large macroaggregates in NT₀₋₅ (results not shown) were similar to those of the small macroaggregates.

2.3.5 Porosity and distribution of pore sizes

The above mentioned different soil fractions and aggregates are components of the ‘in situ’ soil structure. The porosity determined by this ‘in situ’ soil structure is an important controlling factor of the activity of micro-organisms in soil. The total porosities of the 0-20 cm layer under CT and of the 0-5 and 5-20 cm layers under NT were $51.2 \pm 3.9\%$, $58.0 \pm 4.4\%$ and $43.6 \pm 4.3\%$ respectively. The pore size distribution (Figure 2.7) indicates that the order of proportion of small pores was $NT_{5-20} > CT > NT_{0-5}$.

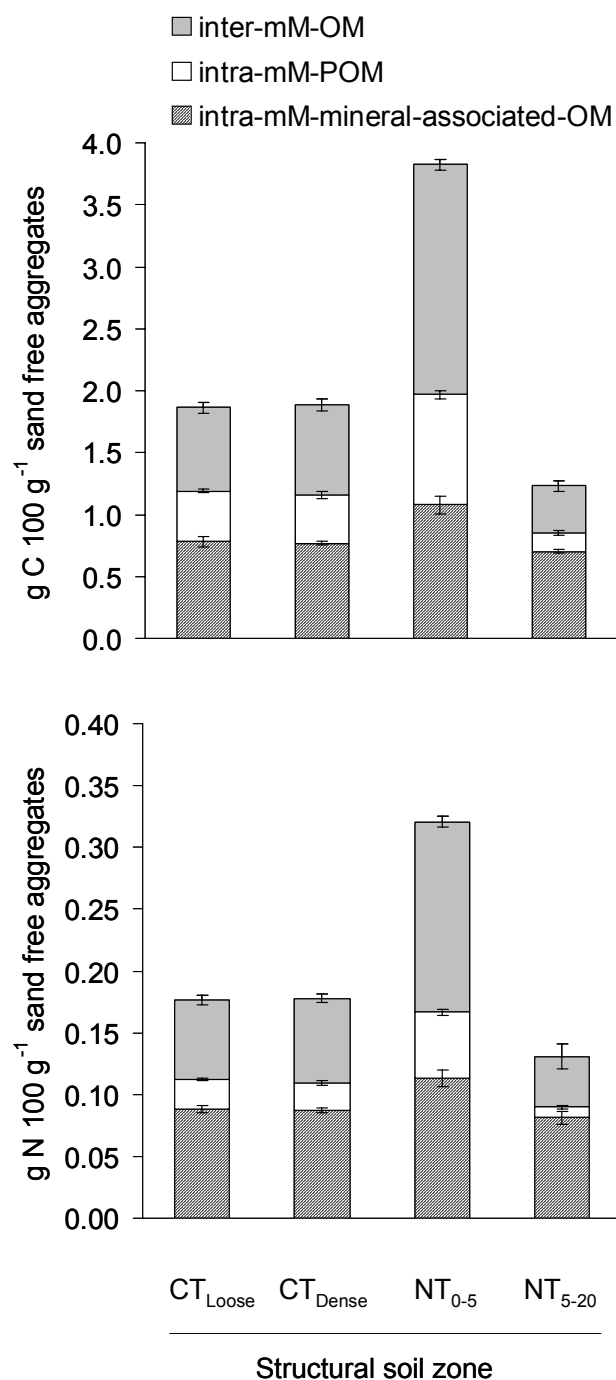


Figure 2.6: C and N concentrations of microaggregates within macroaggregates for the different soil zones of CT and NT (inter-mM-OM = OM inside macroaggregates but outside microaggregates; intra-mM-POM = POM in microaggregates within macroaggregates; intra-mM-mineral-associated-OM = mineral-associated OM in microaggregates within macroaggregates) (bars indicate standard errors).

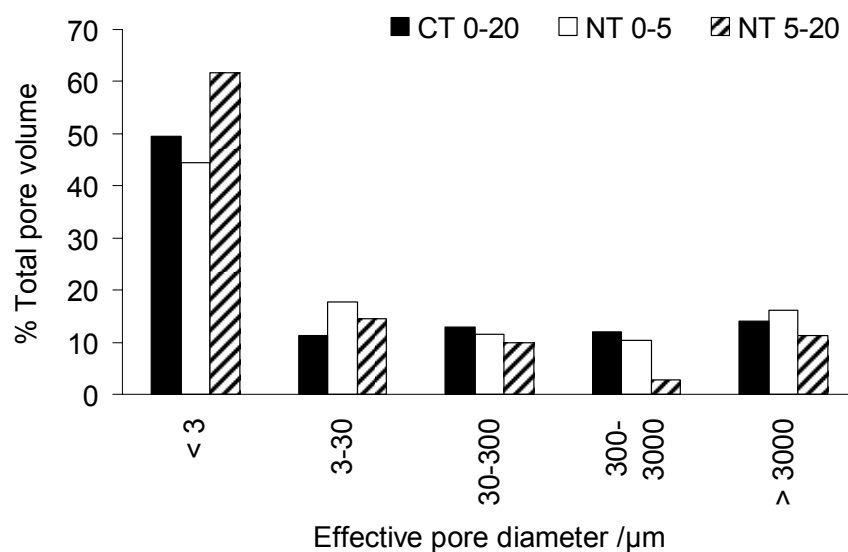


Figure 2.7: Pore size distribution (% of total pore volume occupied by each pore class) of the 0-20 cm plough layer of the conventionally tilled plot (CT) and the 0-5 and 5-20 cm soil layers of the non-tilled plot (NT).

2.4 Discussion

2.4.1 C and N stocks

The differences in C and N stocks between CT and NT in our study were smaller than the ones often reported in literature. However, considering the equivalent plough layer of soils that had been tilled for many generations before differentiation, Balesdent *et al.* (1990), Wander *et al.* (1998) and Yang and Kay (2001) reported — as in our study — only slightly larger C stocks for NT compared to CT soils. In addition, Six *et al.* (1999a) and Yang and Kay (2001) noted significantly smaller stock differences, on a soil volume basis, between tillage systems in fields cultivated for many generations compared to non- or recently cultivated fields before differentiation. Yang and Kay (2001) attributed the differences in C stocks in recently cultivated fields to a slower loss of OM in NT than CT (stocks decreasing in both CT and NT) whereas the differences in fields cultivated for many generations were mainly due to C sequestration (stocks increasing in NT). A second explanation of the small stock differences between CT and NT at Boigneville is that these differences were solely the result of tillage practices, whereas in the CT and NT fields described in the literature, the yearly C and N restitutions are often different. A third explanation emerges from the fact that

the difference in mineralisation of initial C might be higher in soils with a higher C content or in recently cultivated soils due to a larger amount of OM and physically protected OM which results in larger stock differences between CT and NT (Balesdent *et al.*, 1990). Finally, the calculations of C and N stocks, reported in the literature, are often based on volume and this would have led to an overestimation of 18% of the stock differences between CT and NT at Boigneville since the average bulk density was larger in NT.

2.4.2 Mineral-associated OM and POM

As in our study, Franzluebbbers and Arshad (1997a) found a larger percentage of soil C in POM in the 0-5 cm layer and a slightly smaller percentage in the 5-20 cm soil layer under NT compared to CT. However, when expressed on an equivalent volume basis of the plough layer, only small or no differences between NT and CT have been reported in POM-C (Angers *et al.*, 1993; Franzluebbbers and Arshad, 1997a; Mrabet *et al.*, 2001) and POM-N (Angers *et al.*, 1993; Mrabet *et al.*, 2001). In contrast to our study, Jastrow (1996) and Frey *et al.* (1999) observed larger effects on the mineral-associated OM in NT or restored grassland compared to CT. The relative contribution of the free POM, occluded POM and the mineral-associated fraction to C sequestration is affected by soil type, climate, faunal activity, vegetation type and cultivation history (Christensen, 2001; Yang and Kay, 2001). These factors influence the balance between litter input and decomposition. The absence of tillage can provoke a change of this relative distribution of OM over primary organomineral complexes and POM by protection of the labile POM in aggregates and/or lack of incorporation of crop residues. Jastrow (1996) suggested that POM was relatively quickly broken down to mineral-associated C due to the high activity of soil fauna and microbes under favorable moisture and temperature conditions. According to Franzluebbbers and Arshad (1997a) and Christensen (2001) the accumulation of POM is enhanced in a cold and dry climate. This means that the slightly larger proportions of C sequestration as POM-C (58%) compared to mineral-associated C (34%) in the NT plot at Boigneville suggest low microbial activity and/or a considerable physical protection of labile POM. The fact that — as in Wander and Bidart (2000) — our soils contained much more occluded than free POM particularly provides evidence that the POM was physically protected in aggregates.

Note that free residues outside the soil complex (in the surface mulch of NT and the furrows in CT) were not taken into account in our study. Other studies at Boigneville reported a difference about 1.8 tons C ha⁻¹ and 0.03 tons N ha⁻¹ between surface crop residues in NT and incorporated crop residues in CT (Chapter 5).

2.4.3 Slaking-resistant aggregates

The distribution of aggregate size classes suggests that, under NT, an additional portion of the free mineral-associated OM and the microaggregates is occluded in macroaggregates. The significantly larger mean weight diameter (MWD) in the surface layer of NT is in accordance with studies performed on soil cultivated for more than 50 years under similar moisture and climatic conditions (Puget *et al.*, 1995; Frey *et al.*, 1999; Mikha and Rice, 2004). Larger differences in MWD for NT vs. CT were measured on recently, compared with long-term, cultivated fields before tillage differentiation (Frey *et al.*, 1999; Six *et al.*, 2000b). Chan *et al.* (1994) reported some seasonal fluctuations in the distribution of slaking resistant aggregates but differences between the different tillage systems remained steady over time. In our study, soil was only sampled at the end of the bare period between wheat and maize in order to minimise the effect of fresh organic matter present in the soil and the presence of living roots on the total C and N stocks quantification.

Most of the surplus of the C and N stocks under NT compared to CT was situated in aggregates larger than 250 μm (Figure 2.4). This supports the initial hypothesis of physical protection of OM by soil aggregates. The C concentration of the sandfree aggregates in our study increased with aggregate size (Figure 2.5), as also reported in numerous earlier studies (Elliott and Coleman, 1988; Puget *et al.*, 1995; Angers *et al.*, 1997; Six *et al.*, 2000c; Mikha and Rice, 2004). We found that the increase in C concentration was mainly due to considerably higher POM concentrations. Elliott and Coleman (1988) and Six *et al.* (2000c) did not find any differences, per aggregate class, in C concentrations across management treatments. They linked aggregate stability to aggregate C concentration. However, our results can probably be explained by other mechanisms of aggregate stability, besides C concentration or POM concentration, because different C concentrations were observed between the different soil zones, within a given aggregate class. This was also observed by Puget *et al.* (1995) and Mikha and Rice (2004). For the 0-5 cm layer of NT, we corroborate the conclusion of Six *et al.* (2000c) that NT increases the amount of C-rich macroaggregates and decreases the amount of C-depleted microaggregates. However, this conclusion is not valid for the 5-20 cm layer of the NT in our study. Although more macroaggregates were present in NT₅₋₂₀ compared to CT, the C concentrations measured in the small macroaggregates in NT₅₋₂₀ and the microaggregates in CT were similar.

Macroaggregate formation was probably enhanced in NT₀₋₅ at the Boigneville site due to the presence of a larger microbial biomass and numerous fungi (Balabane *et al.*, 2005) producing microbial binding agents and fungal hyphae. Several researchers have reported (Bossuyt *et al.*, 2001; Six *et al.*, 2004) the importance of fungi in the formation and

stabilisation of macroaggregates while bacterial products enhance the formation of microaggregates.

Six *et al.* (1999a) attributed C sequestration to a slower turnover of macroaggregates in NT than in CT. Enhanced macroaggregate disruption by tillage may inhibit the decomposition and fragmentation of coarse intra-POM into fine intra-POM and may hinder further encrustation of this fine intra-POM with clay particles and microbial by-products thereby forming stable microaggregates within macroaggregates. In order to test the former hypothesis, Six *et al.* (2000a) proposed the fine intra-POM to coarse intra-POM ratio as a relative measure of macroaggregates turnover based on ^{13}C measurements. They found that the ratio for NT₀₋₅ was twice the value obtained for CT, whereas in our study it was similar in NT₀₋₅ and higher in NT₅₋₂₀ compared to CT (Table 2.4). We could not control the validity of the fine intra-POM to coarse intra-POM ratio as a relative measure of macroaggregates turnover with ^{13}C measurements but our results of micro- within macroaggregates also provided evidence for the absence of reduced macroaggregate turnover in NT₀₋₅ compared to CT. Unlike Six *et al.* (2000a) and Denef *et al.* (2004), NT₀₋₅ had the smallest weight proportion of micro- within macroaggregates in the macroaggregates (Table 2.5). However, their theory of increased formation of micro- within macroaggregates due to the absence of macroaggregates disruption is probably valid for NT₅₋₂₀.

In literature, a multitude of aggregate isolation techniques exists. However, results on aggregate distribution and composition of aggregates highly depend on the sample pre-treatment (soil water content, drying/rewetting, initial clod sizes, method of rewetting, etc) and the aggregate isolation technique (duration of oscillation time, wet or dry sieving, sieving or shaking, etc) used (Beare and Bruce, 1993; Haynes, 1993; Puget *et al.*, 1995; Ashman *et al.*, 2003). Ashman *et al.* (2003), for example, measured completely opposite compositions of aggregates whether they were obtained by slaking dry soil or shaking wet soil. Therefore, the results on aggregate distribution and composition in our study were only compared with similar studies about slaking resistant aggregates.

2.4.4 The effect of the ‘in situ’ soil structure and porosity

Strong *et al.* (2004) found that microbial activity and the decomposition of added material were less pronounced in soils with large volumes of pores with neck diameters $< 4\ \mu\text{m}$. According to Hassink (1992), the percentage of pores $< 1.2\ \mu\text{m}$ provides a good estimate of physically protected OM. Reverting to our study, the physical protection of organic matter in small pores should be most pronounced in NT₅₋₂₀, intermediate in the plough layer of CT and less present in the surface layer of NT. Considering their low OM concentration, the macroaggregates in NT₅₋₂₀ should be more prone to aggregate disruption compared to CT due

to the low microbial production of binding agents and the use of binding agents as substrate in OM depleted aggregates (Six *et al.*, 2004). However, this is in contrast with the observed distribution of aggregate size classes (Figure 2.4). These findings suggest that OM is physically stabilised in the NT₅₋₂₀ soil layer. The small number of large pores in NT₅₋₂₀ may inhibit the contact between substrate and microbes and also between microbes and predators (with less grazing pressure and thus no stimulation of microbial growth), making it impossible for microorganisms to decompose the intra-POM (Hassink, 1992; Ladd *et al.*, 1993; Strong *et al.*, 2004). In addition, microaggregates may be more resistant to slaking when the small pores become obstructed with microbial products or blocked by compaction (Caron *et al.*, 1996) especially when disruption of the aggregates is infrequent. The NT₅₋₂₀ soil zone had not been tilled for 32 years and had remained practically undisturbed by climatic action. The slower turnover is confirmed by the high ratio of fine over coarse iPOM and the large amount of microaggregates within macroaggregates. The bulk density of the NT₅₋₂₀ zone was also high and the soil under NT was often wetter than under CT. Both phenomena result in larger percentages of water-filled pore space, leading – in turn – to more anaerobic conditions that limit the decomposition of OM. According to Li *et al.* (2004) aggregation by contact may also be promoted by gravitational forces in the deeper soil layers. The cumulative weight on the aggregates together with swelling during wetting of the aggregates in small pore spaces will both reinforce the contact between particles.

No differences in the amounts of primary organomineral complexes and POM, or in the size distribution of aggregates and composition of aggregates were observed between the loose (CT_{Loose}) and dense (CT_{Dense}) structural zones of the CT plough layer in our study. This means that the differences in structure between the loose and dense soil zones had no effect on organic C and N turnover in CT. The latter may be explained by the fact that the plough layer was mixed every year by tillage, leading each time to fragmentation and compaction of certain soil zones and the exposure of new aggregates to dry/wet cycles and raindrop impacts at the soil surface.

2.5 Conclusions

Differences in total C and N stocks were quantified after 32 years of no-tillage (NT) and conventional tillage (CT) in plots with a long-term cultivation history before differentiation and with the same yearly C and N restitutions. The role of physical protection of organic matter on these stock differences was further investigated by examining the changes at different levels of structural complexity. Four structural zones were sampled: loose and dense

soil zones under CT and the 0-5 cm (rich in OM) and 5-20 cm (massive structure) soil layers under NT.

The no-tillage field plot (NT) presented only slightly larger C (15%) and N (10%) stocks than the conventionally tilled plot (CT) after 32 years of tillage differentiation. Mineral-associated OM accounted for only 34% of the difference in C stock, whereas 58% of this difference was due to POM-C. Mineral-associated N and POM-N both contributed about 50% of the difference in N stock. The differences in C and N stocks between CT and NT were almost exclusively due to larger amounts of C and N occluded within slaking resistant aggregates > 250 μm in NT. Regarding the different structural soil zones in CT and NT, no differences in mineral-associated OM, POM, aggregate size distribution and the composition of aggregates were found between the loose and dense structural soil zones in CT, whereas large differences were found between the 0-5 and 5-20 cm layers of NT. The C and N concentrations in the 0-5 and 5-20 cm layers of NT were respectively considerably higher and slightly lower than those of CT. The soil aggregation and OM protection observed in the 5-20 cm soil layer of NT was a result of physical factors (small pores) and the lack of soil disturbance by tillage and climate. Enhanced macro-aggregation due to high microbial activity and high SOM content and the presence of a surface mulch were probably responsible for the preservation of OM in the surface layer of NT. In contrast to the 5-20 cm soil layer of NT, no evidence has been found for decreased macroaggregate turnover in the surface layer of NT. The results suggest that physical protection is an important reason for the larger C and N stocks in NT but that the mechanisms of physical protection are not similar in the surface and 5-20 cm layer of NT.

CHAPTER 3: C AND N MINERALISATION OF UNDISRUPTED AND DISRUPTED SOIL FROM DIFFERENT STRUCTURAL ZONES OF CONVENTIONAL TILLAGE AND NO-TILLAGE SYSTEMS

3.1 Introduction

Converting conventional tillage (CT) to no-tillage (NT) has several important effects on soil functioning. These include increased bulk density leading to less aeration, more fungi, no incorporation of crop residues, larger water stable aggregates, fewer fluctuations in soil temperature and increased soil moisture content (Balesdent *et al.*, 2000; Six *et al.*, 2002). Their combination has a pronounced impact on the biogeochemical C and N cycles in soil including C and N mineralisation. Soil C and N mineralisation determines the decomposition of soil organic matter and has important agronomical and environmental impacts as it affects soil mineral N supply and CO₂ emissions. In the long term, C and N mineralisation has, in addition to C and N inputs, a pronounced influence on total soil C and N stocks.

Numerous authors have reported, for controlled temperature and moisture conditions, larger C and N mineralisation (Beare *et al.*, 1994; Kandeler *et al.*, 1999; Kristensen *et al.*, 2000) and larger amounts of mineralisable N (Doran, 1987; Balesdent *et al.*, 2000) in the 0-5 cm surface layer of NT or minimum tillage systems compared to CT. These authors observed smaller or no such difference in mineralisation in the deeper soil layers of the same fields. Some authors did not observe any difference in C and N mineralisation or amount of mineralisable N within the entire ploughing depth (Doran, 1987; Beare *et al.*, 1994), whereas Franzluebbers *et al.* (1995) reported larger C and N mineralisation for NT. The C and N mineralisation under controlled conditions is greatly determined by the nature and the physical protection of soil organic matter (SOM). It is believed that the physical protection of SOM is more pronounced in NT than CT due to a larger number of water stable aggregates, a higher aggregate stability and a larger proportion of micropores (Balesdent *et al.*, 2000). The pore size distribution may limit SOM decomposition by the localization of SOM in pores not accessible to microorganisms, the limitation of nutrient supply to microorganisms and the restriction of predation of those microorganisms (Elliott and Coleman, 1988; Hassink, 1992; Ladd *et al.*, 1993; Strong *et al.*, 2004).

The C and N mineralisation in different tillage systems is frequently evaluated by incubating sieved soil samples, soil fractions or dried and rewetted soil samples (Elliott, 1986; Beare *et al.*, 1994; Kandeler *et al.*, 1999; Collins *et al.*, 2000). The ubiquitous disruption of soil structure in most studies (e.g. sieving through 2 mm) may have profound effects on soil C and N mineralisation and perhaps differently in soil from CT and NT systems (Cabrera and Kissel, 1988; Gupta and Germida, 1988; Franzluebbers, 1999; Kristensen *et al.*, 2000). Similarly, air drying and rewetting often causes a flush of C and N mineralisation depending on several factors that may differ between CT and NT soils (Cabrera and Kissel, 1988; Hassink, 1992; Van Gestel *et al.*, 1993; Franzluebbers, 1999; Balesdent *et al.*, 2000). Nevertheless, the C and N mineralisation of soil samples with different levels of soil structure disruption may provide information on the protection of organic matter in structures of different sizes. This approach has been used to compare soil disruption effects in native and cultivated soils (Balesdent *et al.*, 2000; Six *et al.*, 2002). Concerning no-tillage and conventional plough tillage, crushing the soil aggregates in subtropical kaolinitic soils generally had greater effects on C or N mineralisation in NT than in CT soils (Beare *et al.*, 1994; Bossuyt *et al.*, 2002), but few data are available for temperate soils. Some were obtained for CT and NT systems by Kristensen *et al.* (2000) but their study only concerned the 0-2 cm layer and the soil samples were not broken down to less than 2 mm. Finally, Franzluebbers and Arshad (1997b) found a slightly larger amount of macroaggregate protected-C (i.e. > 250 µm) for NT compared to chisel tillage systems in a cold and semiarid climate.

Tillage systems affect soil structure at different scales ranging from several centimetres to a few micrometers. In a tilled field, the plough layer consists of compacted clods, porous clods, zones with loose structure, residues located between two furrows and voids (Roger-Estrade *et al.*, 2004). This heterogeneity of structure is, however, usually ignored in C and N mineralisation studies and this is one reason why laboratory studies are often criticised for their lack of reality. The aim of our work was therefore to determine the C and N mineralisation in undisrupted moist soil sampled from various structural zones from long-term no-tillage and conventional tillage plots. The impact of physical protection of organic matter on this C and N mineralisation was also studied. This was done by measuring C and N mineralisation of soil samples after disruption of soil structures of different sizes ranging from microstructures to the level of the 'in situ' arrangement of different macrostructures.

3.2 Materials and methods

3.2.1 Study site

The experimental site is situated at Boigneville (Northern France, 48°33'N, 2°33'E) in soils developed on loess (Haplic Luvisol). The annual temperature and precipitation averages are 10.8°C and 650 mm, respectively. In 1970, the plots were differentiated according to soil management, i.e. no-till (NT) where tillage was absent and conventional till (CT) with annual mouldboard ploughing to approximately 20 cm depth. Before 1970, the fields had been cultivated for many generations and ploughed to 30 cm depth. The plots were cultivated under a maize-wheat rotation and the crop residues left in the field.

3.2.2 Soil sampling and soil processing

One experimental plot (8 x 50 m) was available for each experimental treatment (no-tillage, conventional tillage) taken into account in our study. Soil sampling was based on soil structure profiles (Figure 3.1 and Appendix A), using the method published by Roger-Estrade *et al.* (2004), and was carried out at the end of the fallow period between wheat and maize (March, 2003). Soil trenches (4m large) were dug perpendicular to the tillage, sowing and harvesting directions. We verified with observations at other points in time that each soil profile was fully representative of each experimental plot. Two megastructure zones could be distinguished within the ploughed layer (0-20 cm) of CT (plot 212): one with fine soil and clods with loose structure and eye-visible porosity (CT_{Loose}) and the other with clods of dense and compacted structure without eye-visible porosity (CT_{Dense}). The structural zones in NT (plot 215) were clearly stratified. Soil was sampled from the 0-5 cm layer (NT₀₋₅) which had a highly cracked and loose structure with a large amount of organic matter and from the 5-20 cm layer (NT₅₋₂₀) which had a massive structure with few cracks. We collected soil material at different places from each soil profile to constitute a composite sample of each identified structure that was representative of the whole soil profile because for some soil zones (e.g. the zone with loose structure in conventional tillage) soil had to be collected at numerous places within the profile to obtain enough material. The soil water content was around field capacity at the time of sampling. Any surface residues were removed before sampling the no-till plot and similarly those soil zones with buried crop residues were avoided when sampling CT. The soil characteristics are presented in Table 3.1.

The four composite soil samples were stored fresh at 4°C until further processing. Soil structure was then progressively destroyed by forcing the soil through a series of sieves with decreasing mesh sizes (12.5 mm, 2 mm, 250 µm and 50 µm) following the procedure

described in Figure 3.2. Primary particles (mainly sand) larger than the respective mesh size were discarded (non-destructible fractions I to IV).

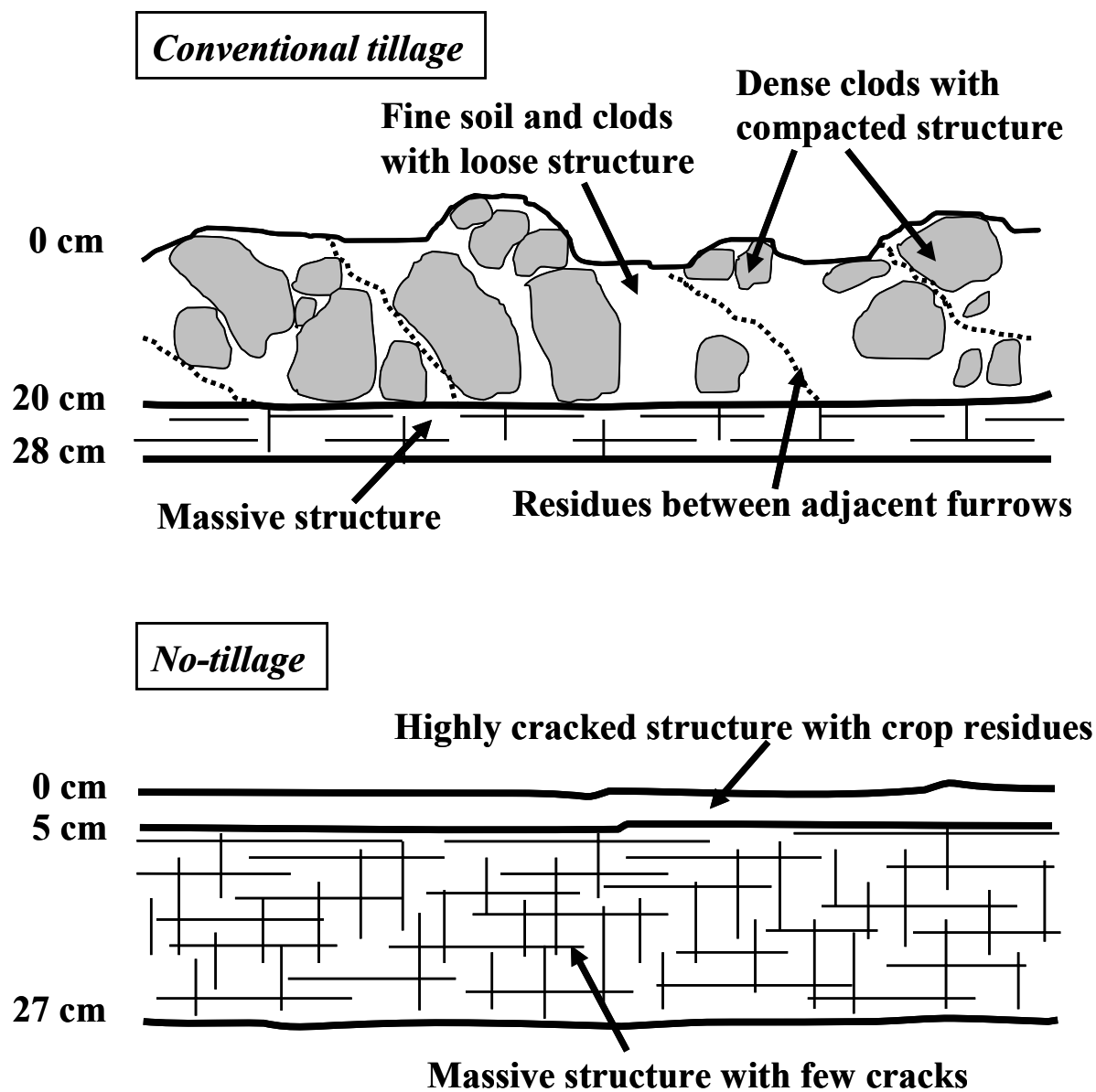


Figure 3.1: Soil structure profiles indicating the four structural zones.

Table 3.1: Main soil characteristics of the different soil structural zones in the conventional tillage (CT) and no-tillage (NT) plots at Boigneville.

		Conventional tillage		No-tillage	
		CT _{Loose}	CT _{Dense}	NT ₀₋₅	NT ₅₋₂₀
Clay (< 2 µm)	(g 100 g ⁻¹ soil)	21.6	22.1	18.8	20.8
Silt (2-50 µm)	(g 100 g ⁻¹ soil)	69.5	69.7	69.9	67.9
Sand (> 50 µm)	(g 100 g ⁻¹ soil)	8.9	8.3	11.3	11.3
pH (water)	(-)	6.1	5.9	4.8	6.0
CaCO ₃	(g 100 g ⁻¹ soil)	0.1	0.1	< 0.1	0.1
CEC	(cmol _c kg ⁻¹ soil)	11.6	11.8	12.3	11.0
Organic C ¹	(g 100 g ⁻¹ soil)	1.09	1.10	2.21	1.00
Organic N ¹	(g 100 g ⁻¹ soil)	0.111	0.113	0.194	0.105
C to N ratio	(-)	9.8	9.7	11.4	9.5
Bulk density	(g cm ⁻³)	1.32 ²	1.32 ²	1.13	1.52
Microbial biomass C ³	(mg C kg ⁻¹ soil)	165 ± 10	161 ± 8	335 ± 10	154 ± 6

¹ The 20-27 cm soil layer contained 0.76 and 0.73 g C 100 g⁻¹ soil and 0.083 and 0.086 g N 100 g⁻¹ soil for CT and NT, respectively

² measured for the 0-20 cm soil layer without distinction between loose soil and dense clods

³ mean values followed by standard errors

3.2.3 Soil incubation

C and N mineralisation was measured in four replicates during incubation of the soil samples at 15°C for 168 days. Soil water pressure was fixed at -63 kPa, based on separately established pF-curves, and the corresponding soil water contents were controlled regularly by weighing and corrected if necessary by adding distilled water. We brought all soil samples to this soil water pressure at which the soil is drier than at field capacity in order to minimise anoxic conditions in the 50 µm soil fractions. Aliquots of soil samples equivalent to 25 g dried soil (soils sieved through 2 mm, 250 µm and 50 µm) or to 100 g dried soil (soils sieved through 12.5 mm) were incubated in polystyrene cups placed in glass jars together with a CO₂ trap (30 ml of 1 M NaOH). The soil inorganic N was measured regularly after shaking (30 min at 20°C) the soil samples with 1 M KCl solution (soil to solution ratio = 1/4). Carbon mineralisation was measured by placing comparable amounts of soil in glass plasma vials in

the presence of a CO₂ trap (10 ml 0.25 M NaOH). The NaOH in the traps was replaced regularly (every 1 - 4 weeks) and analysed for the amount of trapped CO₂.

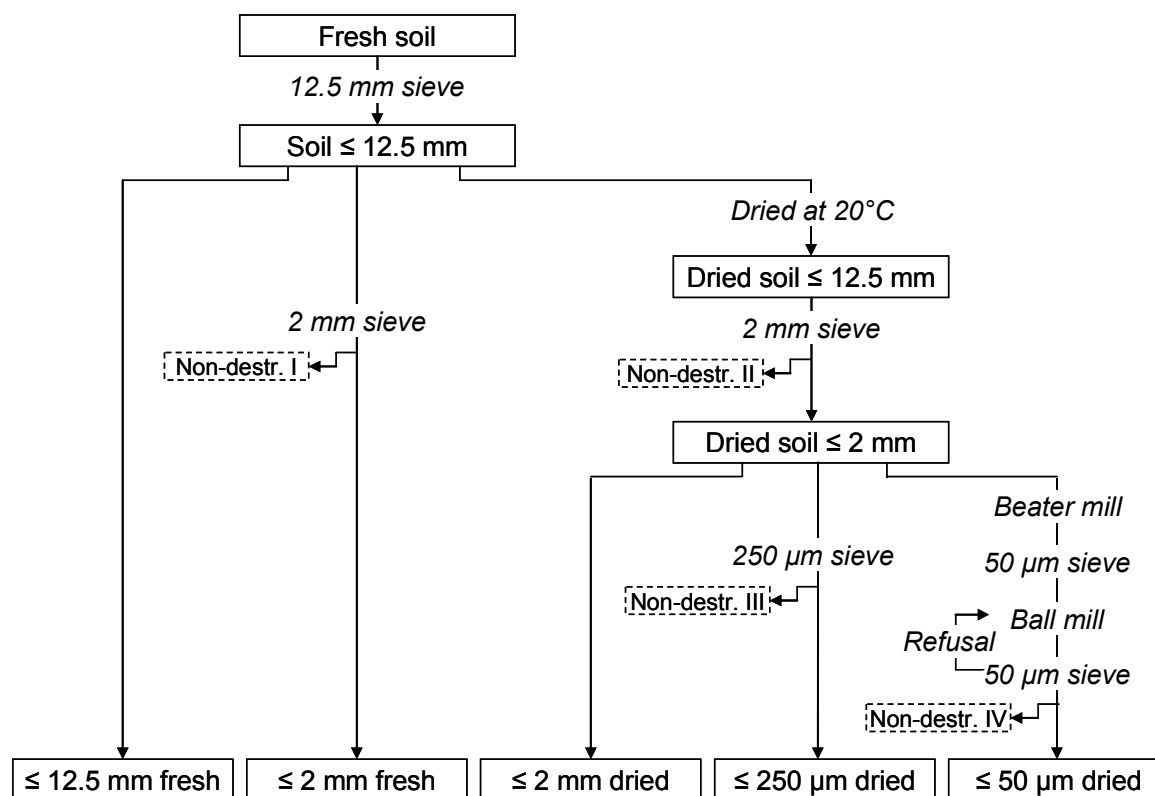


Figure 3.2: Sieving procedures for disrupting soil structure. The soil was forced through the sieves. Non-destr. stands for the non-destructible fraction.

3.2.4 Analytical determinations

The CO₂ trapped by NaOH was measured with a continuous flow colorimeter (TRAACS 2000, Bran+Luebbe, Norderstedt, Germany) as described by Alavoine and Nicolardot (2002). The NO₃⁻ and NH₄⁺ in the soil extracts were measured by a continuous flow colorimeter (TRAACS 2000, Bran+Luebbe, Norderstedt, Germany) using an adaptation of the method proposed by Kamphake *et al.* (1967) for nitrate and nitrite and a method derived from Krom (1980) for ammonium. Total C and N contents of the different sieved soils were determined by dry combustion using an NA 1500 elemental analyser (Fisons, Milan, Italy).

3.2.5 Calculations and statistical analysis

The amounts of absolute C and N mineralised in the undisrupted (12.5 mm) and disrupted soils (2 mm, 250 and 50 μm) were expressed as mineralised C (N) kg^{-1} soil. The calculations for the disrupted soils, took the weight of sand in the removed “non-destructible fractions” into account. The non-destructible fractions I to IV (Figure 3.2) contained up to 8% and 5% of the C and N in the soil before sieving, respectively. The largest values were found for the non-destructible fractions IV. The impact of removing non-destructible fractions I, III and IV on C and N mineralisation was studied by incubating the sieved soil, with or without the respective non-destructible fraction, in the same way as for the main incubation. Removing the non-destructible fractions had no effect on N mineralisation, but did induce a slight decrease in C mineralisation. Therefore, for the main incubation, the C mineralisation value for the disrupted soils was corrected according to the difference between C mineralisation with and without the non-destructible fractions. The fresh soils sieved through 12.5 mm and the dried soils sieved through 2 mm were not corrected because they were the reference samples for the fresh and dried soils, respectively. The fresh soils sieved through 2 mm were corrected for the impact of the removal of the non-destructible fraction I. The dried soils sieved through 250 and 50 μm were corrected for the impact of the removal of the non-destructible fractions III and IV, respectively.

Specific C and N mineralisation was expressed in g mineralised C (N) 100 g^{-1} soil C (N). The mineralisation of protected organic matter was calculated as the difference in C (or N) mineralisation between the disrupted soil and the less disrupted soil (i.e. 2 and 12.5 mm, 250 μm and 2 mm, 50 and 250 μm , 50 μm and 2 mm) over the entire incubation period (168 days). The results were either expressed in mg mineralised C (N) kg^{-1} soil or in g mineralised C (N) 100 g^{-1} soil C (N).

Calculations for the equivalent depths of the plough layer were based on an equivalent soil mass and took into account the bulk densities and respective importance of the different structural zones (CT_{Loose} , CT_{Dense} , NT_{0-5} and NT_{5-20}). The equivalent plough layers for CT and NT corresponded to the 0-20 cm and 0-18.7 cm soil layers, respectively. CT_{Loose} and CT_{Dense} made up 54.3% and 45.7% respectively of the mass of the plough layer in CT whereas the 0-5 cm layer accounted for 21.4% of the mass of the equivalent plough layer in NT.

Data analysis was performed on the different structural zones and on the different disruption treatments within these zones. The fresh and dried soils were analysed separately. The analysis of variance was carried out using the ANOVA procedure and SAS software (SAS Institute, 2001) and the means classification was based on Student-Newman-Keuls tests. Differences were considered significant at $P \leq 0.05$.

3.3 Results

3.3.1 C and N mineralisation of undisturbed soils

C and N showed considerably greater mineralisation in the undisturbed soil samples (12.5 mm) from NT₀₋₅ than from CT_{Loose}, CT_{Dense} and NT₅₋₂₀ (Figure 3.3 and Figure 3.4). These latter three structural zones did not exhibit any significant difference in N mineralisation throughout the entire incubation, whereas the observed order for C mineralisation was: CT_{Loose} > CT_{Dense} = NT₅₋₂₀. Comparable trends were obtained for specific C and N mineralisation (Table 3.2). However, the relative differences between specific C and N mineralisation obtained for NT₀₋₅ and the three other structural zones were much smaller than the differences in C and N mineralisation with the same treatments expressed per 100 g of soil (Figure 3.3 and Figure 3.4). The absolute C and N mineralisation calculated for the equivalent plough layer was significantly larger for NT (1081 ± 60 kg C ha⁻¹, 71.9 ± 5.7 kg N ha⁻¹) than CT (765 ± 21 kg C ha⁻¹, 55.4 ± 1.4 kg N ha⁻¹). The specific C and N mineralisation was equally significantly larger for NT (3.25 ± 0.18 g C 100 g⁻¹ C, 2.19 ± 0.17 g N 100 g⁻¹ N) than CT (2.65 ± 0.07 g C 100 g⁻¹ C, 1.87 ± 0.05 g N 100 g⁻¹ N).

Table 3.2: Specific cumulative C and N mineralisation after 168 days of incubation for the different soil disruption treatments of the structural zones from conventional tillage (CT) and no-tillage (NT) plots.

	Specific C mineralisation (g C 100 g ⁻¹ C)				Specific N mineralisation (g N 100 g ⁻¹ N)			
	CT _{Loose}	CT _{Dense}	NT ₀₋₅	NT ₅₋₂₀	CT _{Loose}	CT _{Dense}	NT ₀₋₅	NT ₅₋₂₀
Fresh soils								
12.5 mm	2.94 ^{b,A}	2.31 ^{c,A}	5.20 ^{a,A}	2.08 ^{c,A}	1.98 ^{b,A}	1.75 ^{b,A}	3.22 ^{a,A}	1.68 ^{b,A}
2 mm	2.72 ^{b,B}	2.35 ^{c,A}	5.17 ^{a,A}	1.97 ^{d,A}	1.90 ^{b,A}	1.78 ^{c,A}	3.10 ^{a,A}	1.56 ^{d,A}
Dried soils								
2 mm	2.99 ^{b,C}	3.14 ^{b,B}	3.89 ^{a,A}	3.03 ^{b,C}	2.68 ^{b,B}	2.66 ^{b,B}	2.94 ^{a,C}	2.88 ^{ab,C}
250 µm	3.11 ^{c,B}	3.20 ^{c,B}	3.70 ^{a,AB}	3.53 ^{b,B}	2.78 ^{c,B}	2.87 ^{c,B}	3.16 ^{b,B}	3.65 ^{a,B}
50 µm	4.26 ^{a,A}	4.22 ^{a,A}	3.50 ^{b,B}	4.52 ^{a,A}	4.36 ^{b,A}	4.23 ^{b,A}	3.56 ^{c,A}	5.07 ^{a,A}

Numbers followed by the same letter are not statistically different ($P < 0.05$): a-d: comparison between sampling zones, A-C: comparison between disruption treatments. Statistical analysis was performed separately for the fresh and dried soils.

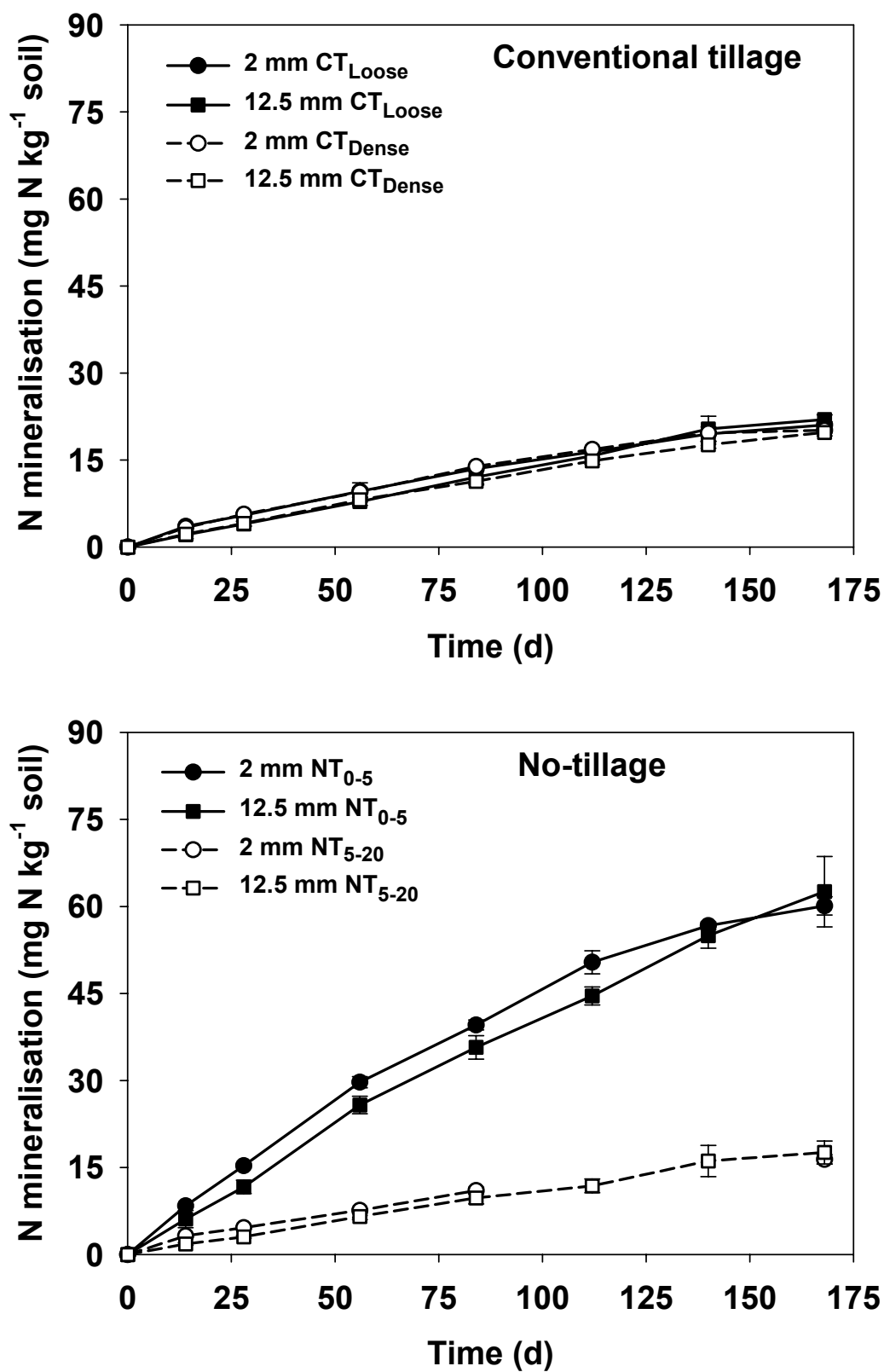


Figure 3.3: Cumulative N mineralisation of fresh soils sieved down to 12.5 and 2 mm from conventional tillage and no-tillage plots and their respective structural zones (error bars indicate standard errors).

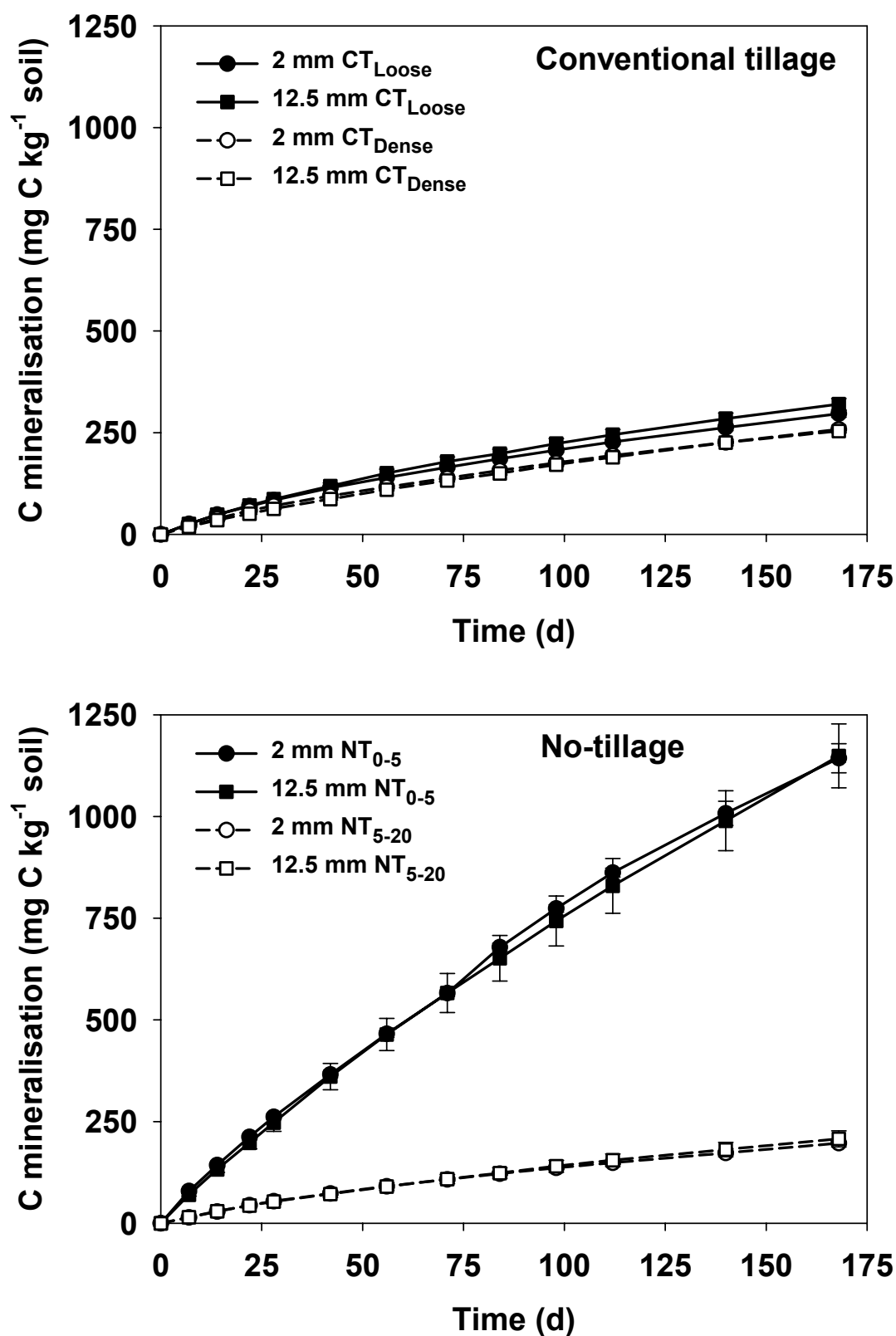


Figure 3.4: Cumulative C mineralisation of fresh soils sieved down to 12.5 and 2 mm from conventional tillage and no-tillage plots and their respective structural zones (error bars indicate standard errors).

3.3.2 Effect of soil drying on C and N mineralisation

The amounts of C and N mineralised in the fresh sieved 2 mm samples were corrected for the removal of the non-destructible fractions I in order to be able to compare this disruption treatment with the fresh soils sieved through 12.5 mm (Figure 3.2). The dried soils sieved through 2 mm were not corrected because these samples were the reference samples for the dried soils. Therefore, in the comparison between the fresh and dried soils sieved through 2 mm, the fresh soil was not corrected for the non-destructible fraction I. Comparison of the C and N mineralisation of 2-mm sieved fresh and dried soils showed that drying and rewetting the soil resulted in considerably higher C (20-50%) and N (39-79%) mineralisation in the CT_{Loose}, CT_{Dense} and NT₅₋₂₀ samples. The largest increase was observed for NT₅₋₂₀. In contrast, similar N and lower C mineralisation were observed in the dried NT₀₋₅ soil compared to the corresponding fresh soil.

3.3.3 Effect of protection of organic matter (OM)

As drying/rewetting had different effects on C and N mineralisation in the four structural zones, the results for the fresh and dried soils were handled separately. For most samples, there were no differences in C and N mineralisation between fresh soils sieved at 12.5 mm and 2 mm, irrespective of the tillage treatment or structural zone (Figure 3.3 and Figure 3.4). A slight but significant drop in C mineralisation was only noted for CT_{Loose} when the 12.5 mm samples were sieved down to 2 mm (Figure 3.4).

Nitrogen mineralisation in dried soils sieved through 50 μ m, was 21% to 76% greater than in soils sieved through 2 mm (Figure 3.5). CT_{Loose}, CT_{Dense} and NT₅₋₂₀ soil samples also showed an increase in C mineralisation after disruption of the soil structures between 50 μ m and 2 mm (Figure 3.6). The same trend was observed for the NT₀₋₅ samples at the beginning of the incubation whereas the cumulative C mineralisation at the end of the incubation period was lower for the 50 μ m than for the 2 mm sieved samples. The C and N mineralisation of the most disrupted soil samples (sieved through 50 μ m), was in the following order: NT₀₋₅ > NT₅₋₂₀ = CT_{Loose} = CT_{Dense} (see Figure 3.5 and Figure 3.6). In contrast, the specific C and N mineralisation values of the most disrupted soils were lower for NT₀₋₅ than for NT₅₋₂₀ and the CT structural zones (Table 3.2).

The increase in absolute and specific C and N mineralisation after the disruption of soil structure (i.e. mineralisation of protected OM) is shown in Table 3.3. The C and N mineralisation of protected OM generally increased with decreasing size of the soil structures. In the 50 to 250 μ m soil structures, this mineralisation of protected OM was significantly greater than zero for all the structural zones except for C mineralisation in NT₀₋₅. In the 250

μm to 2 mm soil structures, a significant mineralisation of protected N was observed for NT_{0-5} and NT_{5-20} and significant mineralisation of protected C for CT_{Loose} and NT_{5-20} . No C and N mineralisation of protected organic matter occurred for structures between 2 and 12.5 mm. When the effects of structure disruption in soil structures between 50 μm and 12.5 mm were combined, NT_{5-20} displayed the largest increase in C and N mineralisation after structure disruption, the differences being greatest for specific mineralisation. The smallest overall effect of structure disruption was observed for NT_{0-5} .

3.3.4 Effect of megastructure in CT

No effect of megastructure (CT_{Loose} vs. CT_{Dense}) was observed on the N mineralisation of undisturbed and disrupted soil (Figure 3.3 and Figure 3.5) which clearly suggests that the organic N was similarly protected in loose soil and in compacted clods. No major effects of megastructure were observed on C mineralisation either (Figure 3.4 and Figure 3.6). A significantly larger C mineralisation in 12.5 mm fresh soils and an effect of disruption of the 250 μm - 2 mm structures was apparent in CT_{Loose} . However, these differences were negligible.

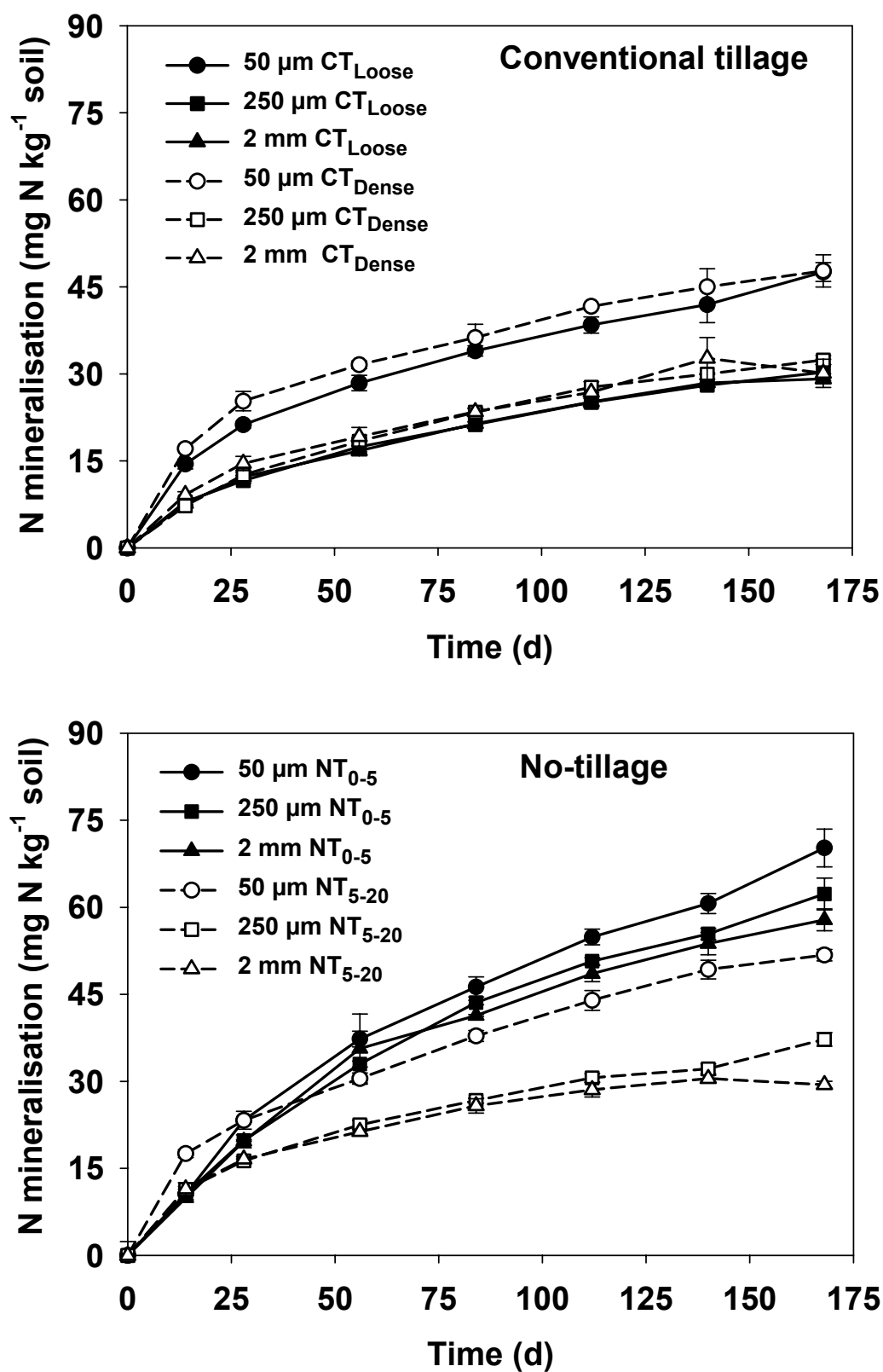


Figure 3.5: Cumulative N mineralisation of dried soils sieved down to 2 mm, 250 µm and 50 µm from conventional tillage and no-tillage plots and their respective structural zones (error bars indicate standard errors).

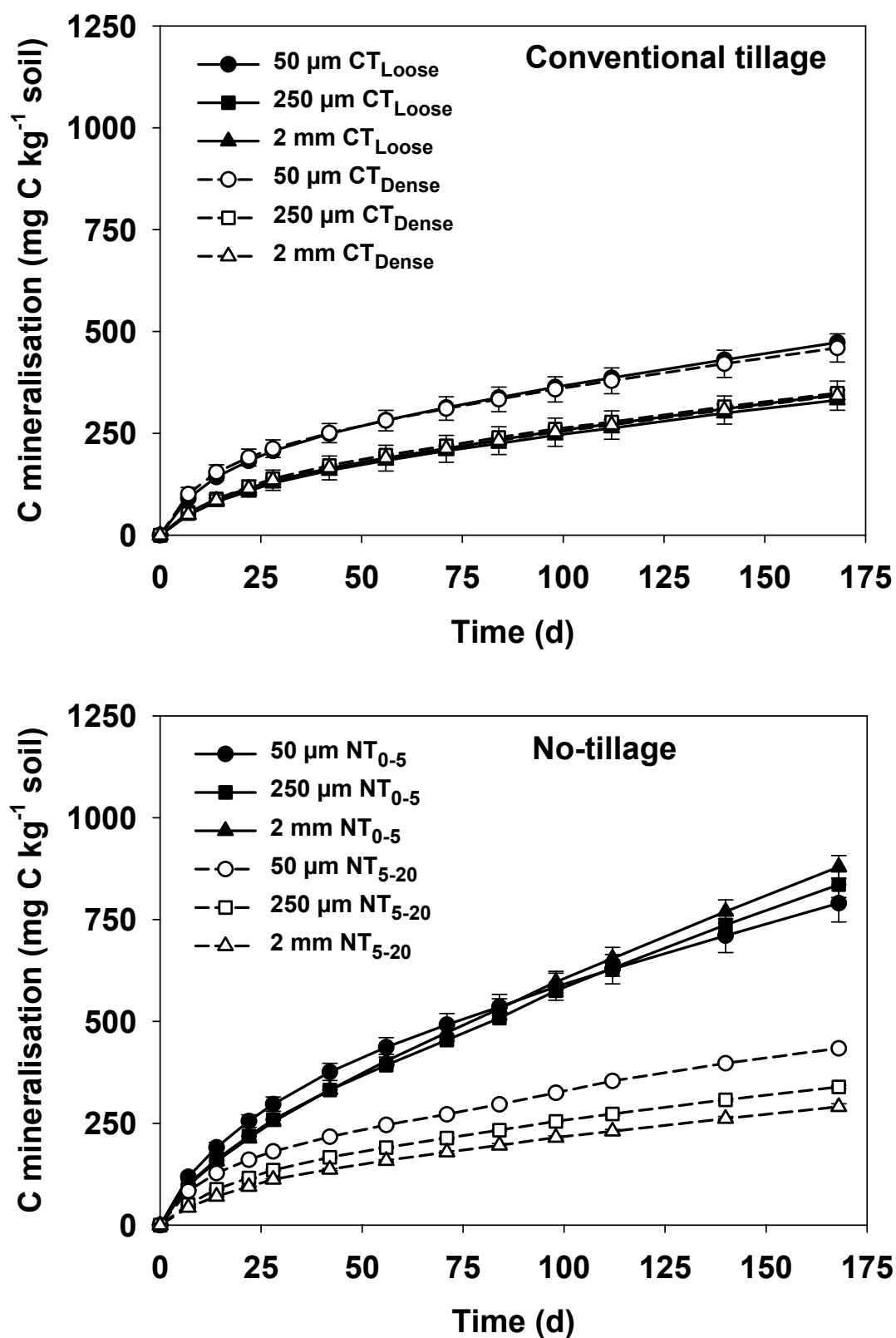


Figure 3.6: Cumulative C mineralisation of dried soils sieved down to 2 mm, 250 µm and 50 µm from conventional tillage and no-tillage plots and their respective structural zones (error bars indicate standard errors).

Table 3.3: Absolute and specific C and N mineralisation of protected OM calculated for a 168-day period in structures of different sizes for the different structural zones of conventionally tilled (CT) and non-tilled (NT) plots.

Structure size	C mineralisation (mg C kg ⁻¹ soil)				Specific C mineralisation (g C 100 g ⁻¹ C)			
	CT _{Loose}	CT _{Dense}	NT ₀₋₅	NT ₅₋₂₀	CT _{Loose}	CT _{Dense}	NT ₀₋₅	NT ₅₋₂₀
2 mm - 12.5 mm	-24*	4	-6	-10	-0.22*	0.04	-0.03	-0.10
250 µm - 2 mm	13*	6	-44	49*	0.12*	0.06	-0.19	0.51*
50 µm - 250 µm	128*	111*	-46	95*	1.16*	1.02*	-0.20	0.99*
50 µm - 2 mm	142*	117*	-90*	143*	1.27*	1.07*	-0.40*	1.49*

Structure size	N mineralisation (mg N kg ⁻¹ soil)				Specific N mineralisation (g N 100 g ⁻¹ N)			
	CT _{Loose}	CT _{Dense}	NT ₀₋₅	NT ₅₋₂₀	CT _{Loose}	CT _{Dense}	NT ₀₋₅	NT ₅₋₂₀
2 mm - 12.5 mm	-0.9	0.4	-2.5	-1.2	-0.09	0.03	-0.13	-0.11
250 µm - 2 mm	1.1	2.3	4.5*	7.8*	0.10	0.20	0.23*	0.77*
50 µm - 250 µm	17.3*	15.4*	7.9*	14.5*	1.59*	1.36*	0.40*	1.42*
50 µm - 2 mm	18.4*	17.7*	12.4*	22.3*	1.69*	1.56*	0.63*	2.19*

Mineralisation of protected OM was calculated as the difference in cumulative C (or N) mineralisation between disrupted and less disrupted soil after 168 days. Negative values indicate larger C or N mineralisation in the less disrupted soil.

* indicates significant differences from zero based on the 95% confidence interval

3.4 Discussion

3.4.1 C and N mineralisation of undisrupted samples

The C and N mineralisation of undisrupted soil samples from the equivalent plough layer was greater in NT than in CT due to the considerably larger C and N mineralisation in the 0-5 cm surface layer in NT. The greater C and N mineralisation of the surface layer in NT is in good agreement with the observations of several authors (Beare *et al.*, 1994; Kandeler *et al.*, 1999; Kristensen *et al.*, 2000). However the larger amounts of OM in the surface layer of NT can only partially explain this higher mineralisation as the specific C and N mineralisation values were also higher for this soil layer. This latter is consistent with Alvarez *et al.* (1995) but not with other authors (Beare *et al.*, 1994; Kristensen *et al.*, 2000) who observed significantly larger amounts of mineralised C or N but lower specific mineralisation for the surface layer of NT. In fact, the higher SOM mineralisation rate in NT₀₋₅ is probably due to a larger

percentage of soil C and N in the particulate organic matter (POM) for which the specific mineralisation is higher than in whole soil organic matter (Alvarez *et al.*, 1995). Indeed, we showed in Chapter 2 that the soil organic matter in NT₀₋₅ contained a larger proportion of POM than the 3 other structural zones.

In our study, C and N mineralisation was measured under controlled conditions in soils which had been sampled at the end of the fallow period in order to quantify the long-term response of C and N mineralisation to tillage management in the absence of the easily degradable crop residues and roots of the previous crop. In addition, the crop residues in the surface mulch and between adjacent furrows were removed before soil sampling. C and N mineralisation under field conditions is influenced by a combination of several factors (i.e. decomposition of incorporated and surface residues and roots, rhizodeposition, soil climatic conditions and the presence or absence of soil disturbance) (Franzluebbers *et al.*, 1995; Kandeler *et al.*, 1999). These may result in less ‘in situ’ mineralisation for NT compared to CT despite the greater mineralisation potential observed for NT under controlled conditions.

3.4.2 Effect of drying the soil

The effect of drying and rewetting differed between the structural zones considered in this study. Soils with a high SOM content have a lower wettability than soils with a lower SOM content (Caron *et al.*, 1996). Thus, rapid rewetting of the CT and NT₅₋₂₀ soil samples with low SOM content leads to release of microbial-C (Kieft *et al.*, 1987) and to disruption of soil aggregates (Caron *et al.*, 1996) which partially explains the observed flush of OM mineralisation in these zones. The smaller mineralisation flush after drying and rewetting the NT₀₋₅ sample, in which the SOM content is higher, can partially be attributed to slower rewetting and to protection of the organic materials by absorption onto colloid surfaces during drying (Caron *et al.*, 1996). Secondly, Van Gestel *et al.* (1993) observed a larger relative decline in the microbial biomass-C after drying for zymogenous biomass (associated with nutrients from plant residues) than for autochthonous microbial biomass in soils under a similar climate to Boigneville. After rewetting, they observed a smaller flush of C mineralisation after rewetting for C derived from added plant residues than for autochthonous soil C. These findings corroborate our results since particulate organic matter derived from crop residues made up a considerably larger proportion of the total soil C in NT₀₋₅ compared with the other soil zones (Chapter 2). Finally, Hassink (1992) provided evidence that rewetting dried sieved soils increases C and N mineralisation by providing water and substrates to micro-organisms, situated in the soil micro-pores, which had limited access to these before the soil was dried. This suggests that in our study the greater effect of drying and rewetting on C and N mineralisation in CT and the NT₅₋₂₀ structural zones compared to NT₀₋₅

may also be partially explained by the higher proportion of such small pores in CT and NT₅₋₂₀ (Chapter 2).

3.4.3 Effect of soil megastructure

In general, organic C and N was mineralised and protected in the same way in soil samples from both megastructure zones (loose soil and dense clods) in the conventionally tilled plot. However, mineralisation was measured under nearly optimal moisture conditions (soil water pressure of -63 kPa). The soil megastructure might have an effect 'in situ', due to the higher bulk density in dense clods compared to loose soil. A higher bulk density more often leads to limiting oxygen concentrations within the clods inducing a drop in nitrification (Houot *et al.*, 1990).

3.4.4 Effect of disrupting soil structure

Organic C and N were not protected in the large macrostructures (2 to 12.5 mm) in either the conventionally tilled or non-tilled soils. These results are in agreement with observations by Craswell and Waring (1972) and Franzluebbers (1999) who did not find any protective effect when large macrostructures in soils of varying texture and clay-type were broken down to 2 mm. However in contrast with our results, Kristensen *et al.* (2000) measured a significant increase in N mineralisation when intact soil samples from the 0-2 cm layer of NT were sieved down to 2 mm whereas no increase was observed in the 0-2 cm sample of CT.

Disruption of the small macrostructures (250 µm to 2 mm) had a significant effect on both C and N mineralisation only in the NT₅₋₂₀ soil (Table 3.3). In contrast, Beare *et al.* (1994) observed a significantly larger increase in C and N mineralisation for the 0-5 cm layer of NT, compared to the 5-15 cm layer or the CT soil layers, when small and large macroaggregates were crushed to 250 µm. The mineralisation of protected C and N in the microstructures (50 to 250 µm) was similar for the structural zones of CT and NT₅₋₂₀ but non-existent (C) or significantly lower (N) for NT₀₋₅ (Table 3.3). Disruption of the soil microstructures led to a greater increase in C and N mineralisation than the disruption of large and small macrostructures, as indicated by other authors (Craswell and Waring, 1972; Balesdent *et al.*, 2000; Bossuyt *et al.*, 2002; Six *et al.*, 2002; Pulleman and Marinissen, 2004).

In general, the disruption of soil structures in the 0-5 cm layer of the non-tilled plot in our study induced a decrease in C mineralisation and had little effect on N mineralisation whereas the greatest increase in C and N after structure disruption was observed in NT₅₋₂₀. These observations contrast with most recent reports (see above). Several hypotheses and observations can be put forward to explain our results:

- The “aberrant” results for 0-5 cm layer of no-till did not change drastically whether the correction for the removal of the non-destructible fractions was made or not, proving that the two issues were not related.
- Structure disruption had a more pronounced effect on N mineralisation than on C mineralisation, as already demonstrated by Elliott (1986), Hassink (1992), Aoyama *et al.* (1999) and Six *et al.* (2002). In contrast, Powlson (1980) and Beare *et al.* (1994) did not find any difference in the effect of crushing between C and N mineralisation. In fact, the low C to N ratio (< 8.3) of the additional mineralisation after structure disruption found in our study suggests that the released protected OM is of microbial origin or associated with the silt and clay fraction rather than from particulate organic matter, which has a higher C to N ratio (16.5-17.4). This hypothesis is further supported by the fact that Van Gestel *et al.* (1993) observed a smaller C to N ratio of the mineralisation flush after soil disturbance by drying and rewetting compared to basal soil mineralisation due to a lower relative contribution of mineralised ^{14}C labeled C from added plant residues. POM made up a significantly higher proportion of C and N both within and outside water stable aggregates of NT₀₋₅ compared to the three other structural zones in our study (Chapter 2). This suggests that a substantial part of this occluded POM in water stable aggregates is not highly protected and/or recalcitrant. Indeed, Collins *et al.* (2000) observed an increase in acid-resistant soil C with a high mean residence time after several years of no-till. When it is deprotected, recalcitrant OM does not lead to any increase in C mineralisation (Gregorich *et al.*, 1989). The considerably larger C and N mineralisation of the 12.5 mm sieved soil samples of NT₀₋₅ compared to the other soil zones supports the hypothesis that there is a substantial part of soil organic matter that is not highly protected within structural soil elements smaller than 12.5 mm.
- Bremer *et al.* (1991) and Angers and Recous (1997) measured higher C mineralisation for ground than for whole straw but found no effect or only a small decrease in C mineralisation when residues with low C to N ratios were ground. However, Vestergaard *et al.* (2001) concluded that decomposition, regardless of N content, was not necessarily enhanced by increasing the surface area of residues. Although it seems unlikely that grinding greatly influenced C mineralisation of the partially decomposed organic materials with low C to N ratios in our study, it cannot be excluded that this procedure may have influenced our results, especially in the NT₀₋₅ soil sample where the amount of particulate organic matter was greatest.
- Grinding induces a reduction of microbial biomass (Balesdent *et al.*, 2000). Sorensen (1983) showed that after grinding ($< 150\text{ }\mu\text{m}$), the soil microbial biomass remained smaller than the original amount during a 90-day soil incubation at 20°C. In our study, sieving may have had different effects on the microbial biomass in NT₀₋₅ compared to the other structural zones

because the surface layer of NT contains more fungi (Six *et al.*, 2002). The hyphal network may have been destroyed during disruption of the soil structure.

- The smaller cumulative C mineralisation over 168 days for more disrupted compared with less disrupted soils in NT₀₋₅ is due to the fact that the observed increase in more disrupted soil at the beginning of the incubation period was overcompensated by the decrease at the end of the 168 days period. In contrast, the C mineralisation rates in the CT and NT₅₋₂₀ samples remained higher in the more disrupted soils, throughout the entire incubation period. However, the conclusions drawn for the cumulative results over the entire 168 days period or over only the first week are similar because even at the beginning of the incubation, structure disruption induced smaller increases in C mineralisation in NT₀₋₅ than in the three other structural zones.

Finally, destruction of soil structures may increase the mineralisation of OM previously protected in small pores (Hassink, 1992; Ladd *et al.*, 1993). The proportion of small pore fractions (< 3 µm) in the four structural zones (Chapter 2) followed the same order as the amount of mineralised protected C and N in structures between 50 µm and 2 mm: NT₅₋₂₀ > CT > NT₀₋₅. This result supports the theory of Hassink (1992) who found a significant correlation between the relative importance of micro-porosity (< 1.2 or 0.2 µm) in soil and the amount of physically protected OM.

3.5 Conclusions

Differences in soil structure created by tillage systems are often believed to have large impacts on C and N mineralisation in turn influencing total soil C and N stocks, CO₂ emissions and soil mineral N supply. The objectives of our work were therefore (i) to study C and N mineralisation in undisrupted fresh soils from long-term conventional mouldboard plough (CT) and no-till (NT) systems in the Northern France and (ii) to evaluate at which scale soil structure plays a significant role in protecting organic matter against C and N mineralisation. The 'in situ' heterogeneity of soil structure was taken into account during sampling. Undisrupted soil from the surface layer of NT showed greater absolute and specific C and N mineralisation than the CT soil samples and the samples from the 5-20 cm soil layer of NT. Consequently, C and N mineralisation of undisrupted samples was higher for the NT than for the CT soil. In the CT soil, the megastructure (i.e. loose soil compared to dense clods) had no effect on either the absolute or specific C and N mineralisation. In addition, no apparent protection of soil organic matter in different sized structures could be ascribed to this megastructure.

Carbon and nitrogen mineralisation of protected OM was most pronounced in the microstructures (disruption from 50 to 250 μm). Disruption of the small macrostructures (250 μm to 2 mm) led to a significant increase in both C and N mineralisation only in the 5-20 cm layer of the NT plot. No protective effects were found for the large macrostructures (2 to 12.5 mm). Disrupting the 50 μm to 2 mm soil structures produced the greatest increase in C and N mineralisation for the 5-20 cm layer of NT, differences with the CT structural zones being small. However structure disruption did not increase C mineralisation in the 0-5 cm layer of NT. The 0-5 cm layer of NT also showed the smallest increase in specific N mineralisation, whereas the effect on absolute N mineralisation was similar in the 0-5 cm layer of NT and the plough layer of CT. Our results suggest that the surplus C and N in the 0-5 cm layer of NT after 32 years of differentiation is (i) mainly located in a non-highly protected form and/or (ii) situated in recalcitrant OM (which does not induce an increase in C and N mineralisation when liberated) as neither leads to an increase in C mineralisation when the structural soil elements are broken.

CHAPTER 4: LONG-TERM EFFECT OF TILLAGE SYSTEMS ON SOIL ORGANIC NITROGEN MINERALISATION UNDER FIELD CONDITIONS

4.1 Introduction

The reduction of tillage intensity by suppressing mouldboard ploughing and adopting conservation tillage techniques (e.g. reduced tillage or no-tillage) became increasingly popular in France over the last decade (Trocherie and Rabaud, 2004). The motivations can be economic (reduction of running costs and working time) and/or agronomical or environmental (e.g. reduction of erosion, increase of organic matter at the soil surface to improve soil structure, increase of water holding capacity and reduction of erosion). A modification of the tillage system also changes agricultural practices concerning fertilisation, weed control by herbicides and seeding. Therefore, farmers need information on the soil N supply in order to optimise their fertiliser management.

The effect of reduced tillage on N supply by the soil in farming systems is yet unclear. Soil tillage affects the physical and biological functioning of the soil and consequently C and N dynamics (Blevins and Frye, 1993; Martens, 2001). Residues are situated on the soil surface in no-tillage (NT) whereas they are incorporated in the plough layer in conventional mouldboard ploughing systems (CT). Under field conditions, the decomposition of crop residues at the soil surface is generally slower compared to incorporated residues (Blevins and Frye, 1993; Burgess *et al.*, 2002) and residue placement may affect the balance between gross N immobilisation and gross N mineralisation processes in the soil (Franzluebbers *et al.*, 1995; Burgess *et al.*, 2002). Moreover, the reduction of tillage induces an accumulation of organic matter in the upper layers of the soil profile (Pekrun *et al.*, 2003) and changes the distribution and/or amounts of potentially mineralisable N, microbial biomass as well as microbial activities (Doran, 1987; Franzluebbers *et al.*, 1995; Kandeler *et al.*, 1999). All these modifications in combination with the soil climate (temperature and soil water pressure) will interact on the net mineralisation of organic N, in turn determining the soil N supply. The large variability in climatic conditions, crop rotations, soil types, techniques of tillage and duration of differentiation explains the variation in data on soil N supply published in

literature. Most studies found that the potential N mineralisation, measured under controlled optimal temperature and moisture conditions, is comparable between tillage systems for a soil layer corresponding to the tilled layer (Carter and Rennie, 1982; Rice *et al.*, 1986; Doran, 1987). But Franzluebbers *et al.* (1995) measured larger potential N mineralisation for NT. Mineralisation under field conditions has been lower for non-tilled compared to tilled plots (Brye *et al.*, 2003) or similar for both tillage systems (Rice *et al.*, 1986). Other authors did not find any difference in nitrate content down to 100-150 cm depth between different tillage systems (Zihlmann *et al.*, 2001; Fuentes *et al.*, 2003).

The aim of our work was to determine soil N mineralisation under real farming conditions with different long-term tillage systems in Northern France. The N mineralisation was assessed using two approaches: (i) calculation of N leaching and mineralisation using the simple calculation model LIXIM in combination with the evolution of water and nitrate content in different layers of the soil profile and (ii) estimation of the N mineralisation potential in soil samples incubated under controlled laboratory conditions.

4.2 Materials and methods

4.2.1 Experimental set-up

The experimental site is located at Boigneville (Northern France, 48°33'N, 2°33'E). Average annual temperature and precipitation are 10.8°C and 650 mm, respectively. Climatic conditions during the study period are presented in Figure 4.1. The Boigneville soil is a Haplic Luvisol (FAO) with a deep loam overlaying fissured calcareous material. The calcareous material is found at 70 – 100 cm depth. Before the start of the experiments, the fields had been under cultivation for many generations and had been mouldboard ploughed to 30 cm depth. The soil tillage systems were differentiated in 1970 and the plots were subject to either no-tillage (NT), conventional tillage (CT) or superficial tillage (ST). The site is cultivated with a maize/wheat rotation. There are two plots for each tillage treatment but when the first plot is under maize, the other plot is under winter wheat. Therefore, only one replicate for each experimental treatment could be taken into account. Nevertheless, two additional experimental treatments were included, CT_{rep} and NT_{rep}, which are semi-repetitions of CT and NT, respectively. In CT_{rep} and NT_{rep} crop residues were exported from 1983 to 1994 whereas in CT and NT crop residues were never exported since 1970. The characteristics of the different tillage systems and the soil properties in the different plots are described in Table 4.1 and Table 4.2, respectively. To avoid crop effects on N dynamics (e.g. rhizodeposition and N

accumulation in roots), soil measurements were only done between wheat harvest and maize sowing, i.e. from August 2003 to April 2004. During this measurement period, the soil was kept bare using chemical or manual weeding. Three microplots (10 x 4.5 m) were set up for each experimental plot in order to take the spatial variability into account.

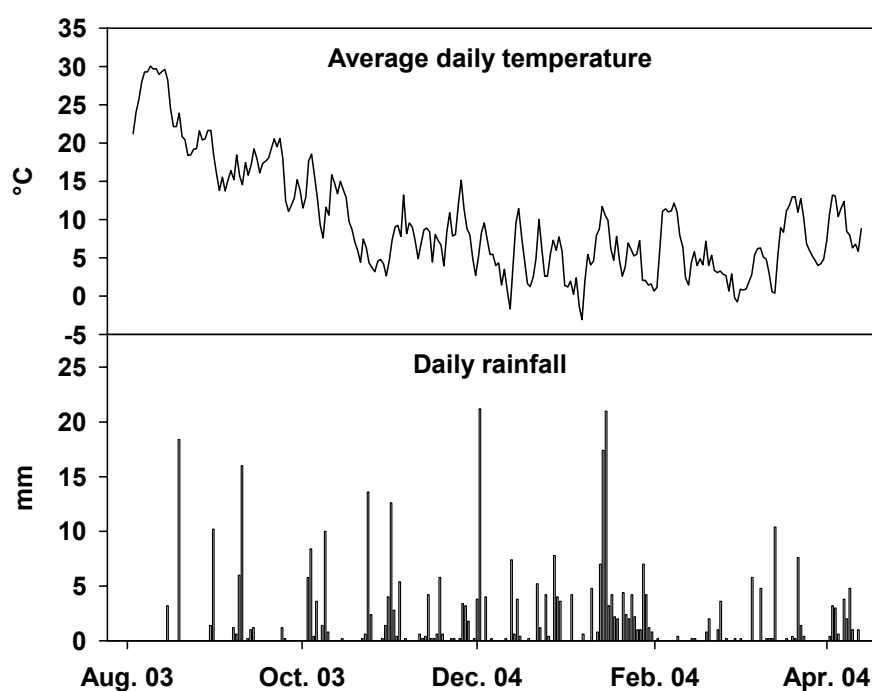


Figure 4.1: Daily climatic conditions during the measurement period at Boigneville.

4.2.2 Soil measurements

Soil was sampled monthly in each microplot using a hydraulic coring device with a 15 mm diameter auger (MCL3, Geonor, Oslo, Norway). At each sampling date, eight soil cores were taken in each microplot. Each core was subsequently divided into 5 soil layers (0-25, 25-50, 50-75, 75-90 and 90-120 cm). Afterwards the eight samples of each soil layer within each microplot were pooled. The samples were frozen until measurement of water content (oven-dry basis, 105°C, 24 h) and mineral-N.

The bulk density of the superficial layer (0-5 or 0-10 cm) was measured with a rubber balloon density meter (Nardeaux-Humisol, Saint Avertin, France). The bulk density of deeper soil layers was measured on undisturbed soil cores sampled with 98 cm³ steel cylinders (5 cm

high and 5 cm dia.) pushed horizontally into the soil using a pneumatic jack (4 replicates per soil layer). Subsequently, the dry weight of the sampled soil was measured after oven-drying at 105°C. Water content at field capacity of the different soil layers was estimated with the largest measured water contents in the field. The water content at the permanent wilting point was determined at a soil water pressure of -1600 kPa using the Richards pressure plate.

Meteorological data were recorded by a meteorological station located close to the experimental plots. In addition, soil temperature was measured at different depths (2.5, 12.5 and 25 cm) in the CT and NT plots.

Table 4.1: Main characteristics of the different tillage systems under a maize-wheat rotation at Boigneville.

Tillage system	Conventional tillage (CT)		No-tillage (NT)		Superficial tillage (ST)
Plots considered ¹	115	224	111	223	221
Treatment code	CT _{rep}	CT	NT _{rep}	NT	ST
Tillage	Mouldboard ploughing down to 20 cm depth (At the 19th of November 2003 during the measurement period)		None		Superficial tillage with Rotavator HR down to 8-10 cm depth (At the 24th of November during the measurement period)
Crop residues	Incorporated in soil during tillage		No incorporation in soil		Incorporated in soil during superficial tillage
Particularities	Crop residues exported from 1983 to 1994, but not exported during the other periods	Crop residues never exported	Crop residues exported from 1983 to 1994, but not exported during the other periods	Crop residues never exported	Crop residues never exported

¹ The plots had a size of 50 x 8 m

Table 4.2: Main characteristics of the 0 to 25 cm soil layer of the different tillage systems at Boigneville.

Treatment	Unit	CT	CT _{rep}	NT	NT _{rep}	ST
Clay (< 2 µm)	g kg ⁻¹	266	224	241	239	234
Silt (2-50 µm)	g kg ⁻¹	651	675	665	683	664
Sand (50-2000 µm)	g kg ⁻¹	83	101	97	88	102
Organic C	g kg ⁻¹	11.0	10.8	11.2	10.8	12.2
Organic N	g kg ⁻¹	1.13	1.10	1.12	1.13	1.21
C to N ratio	-	9.8	9.8	10.0	9.5	10.1
pH water	-	6.5	6.3	6.3	6.0	6.4
CaCO ₃	g kg ⁻¹	< 1	< 1	< 1	< 1	< 1
CEC Metson	cmol _c kg ⁻¹	13.7	11.7	12.9	12.5	12.8

4.2.3 Analytical determinations

Mineral-N (NO₃⁻ and NH₄⁺) content was measured by extracting 100 g fresh soil with 200 ml 1 M KCl during 30 min. at 20°C. Nitrate-N and ammonium-N concentrations in each extract were determined by continuous colorimetry using a TRAACS 2000 autoanalyser (Bran & Luebbe, Norderstedt, Germany) following adaptations of the methods proposed by Kamphake *et al.* (1967) and Krom (1980), respectively.

4.2.4 Calculation of drainage, leaching and mineralisation

The amount of evaporated water, drained water, mineralised N and leached N were calculated using the LIXIM model (Mary *et al.*, 1999). The movement of water and nitrate is defined by the algorithm proposed by Burns (1974). Water and nitrates move downwards from one layer to the next when the water content equals field capacity. Simultaneously, the model takes rain, evaporation and N mineralisation into account. The optimising routine of this simple model calculates the ratio of actual to potential evapotranspiration and the potential rate of mineralisation during each sampling interval [t₁, t₂] by optimising simulations of the measured water and mineral-N profiles in several soil layers at dates t₁ and t₂. The time step of the model is one day. The input variables and fixed and fitted parameters are listed in Table 4.3.

In order to compare N mineralisation in different sites, treatments or periods, the LIXIM model calculates potential N mineralisation (kg N ha⁻¹) expressed in normalised days (nd)

having a soil temperature of 15°C and a soil moisture at field capacity. These normalised days are calculated with temperature and moisture functions for soil C and N mineralisation (Mary *et al.*, 1999). Finally, the 'in situ' potential N mineralisation rate V_p (kg N ha⁻¹ nd⁻¹) was determined by linear regression using calculated cumulated potential N mineralisation data for each treatment:

$$N \text{ mineralized}_t = V_p * t$$

with $N \text{ mineralized}_t$ equal to the cumulative potential N mineralisation (kg N ha⁻¹) over t days with t expressed in normalised days at 15°C and at field capacity.

The calculations with the LIXIM model were done for each microplot separately to allow statistical analysis on the output variables for each treatment.

Table 4.3: Input variables and fixed and fitted parameters used in the LIXIM model

Input variables

Daily rainfall, potential evapotranspiration (Penman), soil temperature

Water content, NH_4^+ and NO_3^- contents (for each layer, at each measurement date)

Bulk density (for each layer)

Water content at field capacity (for each layer)

Water content at field capacity (for each layer)

Fixed parameters

Maximum soil depth contributing to water evaporation

Coefficient of reduction of evaporation through depth

Maximum soil depth contributing to N mineralisation

Temperature coefficient (Van 't Hoff) for N mineralisation

Relative N mineralisation rate at wilting point

Mineral N concentration in rain

Fitted parameters

Actual to potential evapotranspiration ratio

Potential mineralisation rate at reference temperature and optimal water content

4.2.5 Soil incubations

Potential N mineralisation of soil was also measured during soil incubations under controlled laboratory conditions. For the CT and NT treatments, soil samples were taken in April 2004 from the 0-5 cm, 5-20 cm and 20-25 cm soil layers, sieved at 12.5 mm and incubated like the samples of the plots of period I described in Chapter 3. Soil water pressure was maintained at -63 kPa, controlled regularly by weighing and corrected if necessary by adding distilled water. Soil samples were incubated at 15°C for 168 days. Inorganic N of incubated soil was regularly extracted by shaking (30 min at 20°C) the soil samples with 100 ml 1 M KCl solution. The inorganic N in the extracts was then determined as previously described. Finally, the mineralisation potential under controlled laboratory conditions (expressed in kg N ha⁻¹) was established by summing up the values for the different soil layers of the 0-25 cm soil layer, taking their measured bulk density into account and by applying the moisture function of LIXIM to convert the time to normalised days. The incubation results for NT were summed up for a 0-23.7 cm soil layer in stead of a 0-25 cm soil layer in order to take into account a comparable soil mass for CT and NT, namely 3399 tons ha⁻¹.

4.2.6 Statistical analysis

The linear regressions and correlation coefficients were calculated using a spreadsheet. The statistical criteria used to evaluate the model are the mean difference (MD), the absolute root mean square error (RMSE) and the modelling efficiency (EF) (Smith *et al.*, 1996). The variance analysis and mean classification were respectively performed using the ANOVA procedure and the Student-Newman-Keuls test of the SAS software (SAS Institute, 2001). The replicates for each experimental treatment consisted of the three microplots within each experimental plot.

4.3 Results

4.3.1 Measured water and nitrate contents

The observed gravimetric water content and nitrate content in each of the five soil layers of the experimental treatments are shown in Figure 4.2 and Figure 4.3, respectively. The values of both variables for CT_{rep} and NT_{rep} are not presented because they were very similar to CT and NT, respectively.

The evolution of the gravimetric water content in the different soil layers of the three tillage systems was quite similar (Figure 4.2). In the upper layer (0-25 cm), the soil water content increased from August to December, remained at field capacity during winter and decreased from March onwards. For deeper layers, similar patterns were observed but the amplitude of variation attenuated with depth. In the upper soil layer, soil moisture was almost comparable between the three tillage treatments, whereas in the deeper soil layers, differences between the tillage systems were often significant ($p < 0.05$). Compared to the other tillage treatments, the soil water content was larger for NT in the three deepest layers and smaller for ST in the two deepest layers. Considering the whole 0-120 cm soil profile, the evolution of water content was comparable between the three tillage treatments, but NT presented for most of the measurement days significantly larger water contents compared to ST and CT.

The evolution of the nitrate content in the different soil layers was also comparable between the three tillage systems (Figure 4.3). In the upper soil layer, the nitrate content was less than 20 kg N ha^{-1} in August 2003 and increased during autumn up to $32\text{-}48 \text{ kg N ha}^{-1}$, depending on the tillage system. From November 2003 to February 2004, nitrate was transferred to deeper layers by leaching. At the end of this leaching period, the upper soil layer contained less than 10 kg N ha^{-1} . From February to April 2004, the nitrate content in this upper layer increased again due to a more intense soil organic N mineralisation. The evolution of the nitrate content in deeper soil layers was the result of nitrate transfer by leaching. Regarding the difference in the amount of nitrate between the three tillage systems, significant differences ($p < 0.05$) were observed for the upper soil layer: the nitrate content was larger in NT (except for the larger values for CT in spring 2004). Considering the whole 0-120 cm soil profile, the nitrate content was not significantly different between the three tillage systems ($p < 0.05$) except for one date. At the end of the measurement period (April 2004), more than 75 kg N ha^{-1} was present in the 0-120 cm soil profiles. LIXIM also takes the temporal variation in NH_4^+ content into account. However, the NH_4^+ measurements did not show large temporal variations for the different soil layers so that this variation only had a negligible effect on the model calculations.

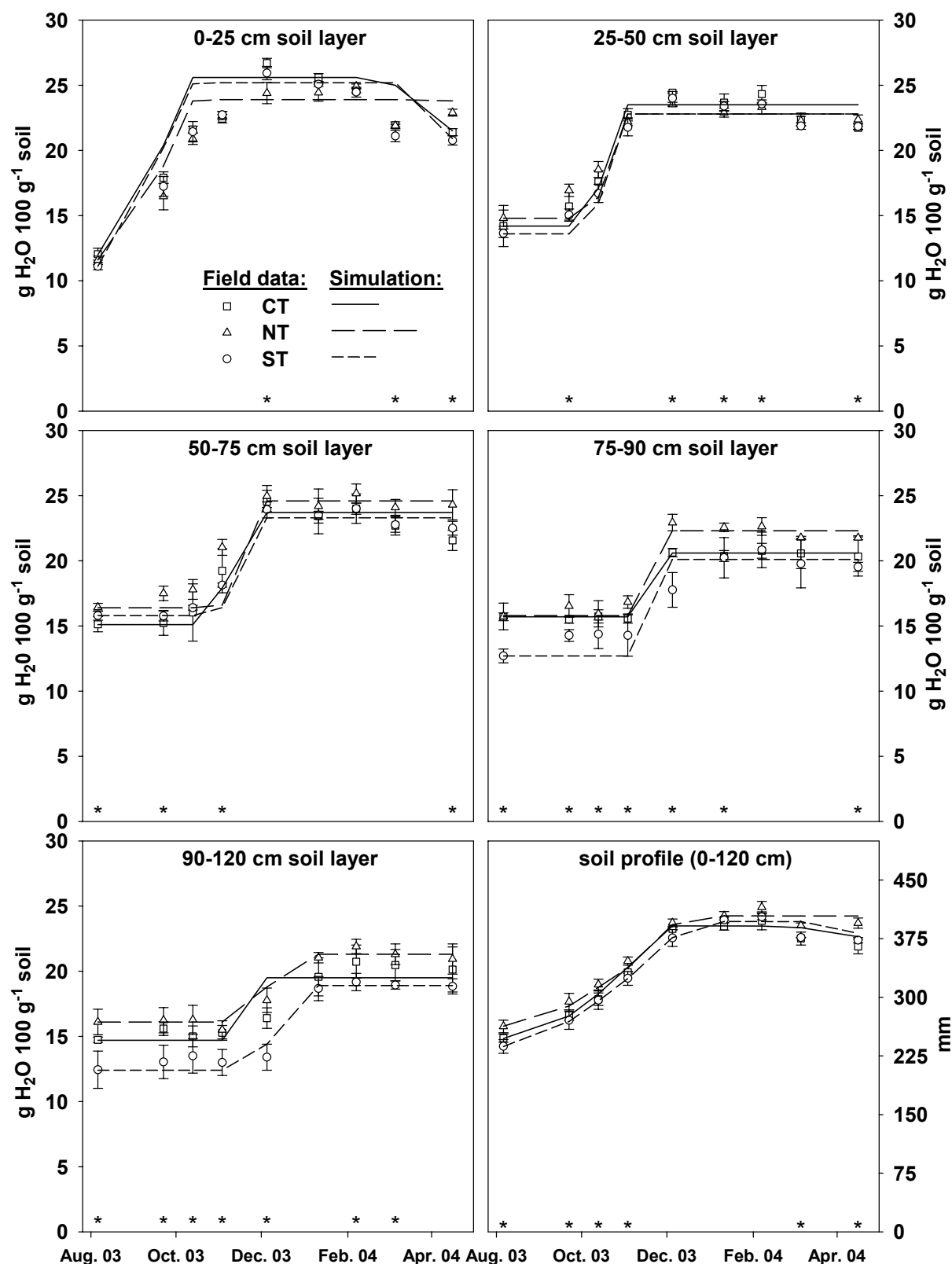


Figure 4.2: Evolution of observed (symbols) and simulated (lines) water contents for the different soil layers and the whole soil profile of the different tillage treatments. Verticals bars represent standard errors. Dates labelled with * indicates significant differences among treatments at the 5 % significance level.

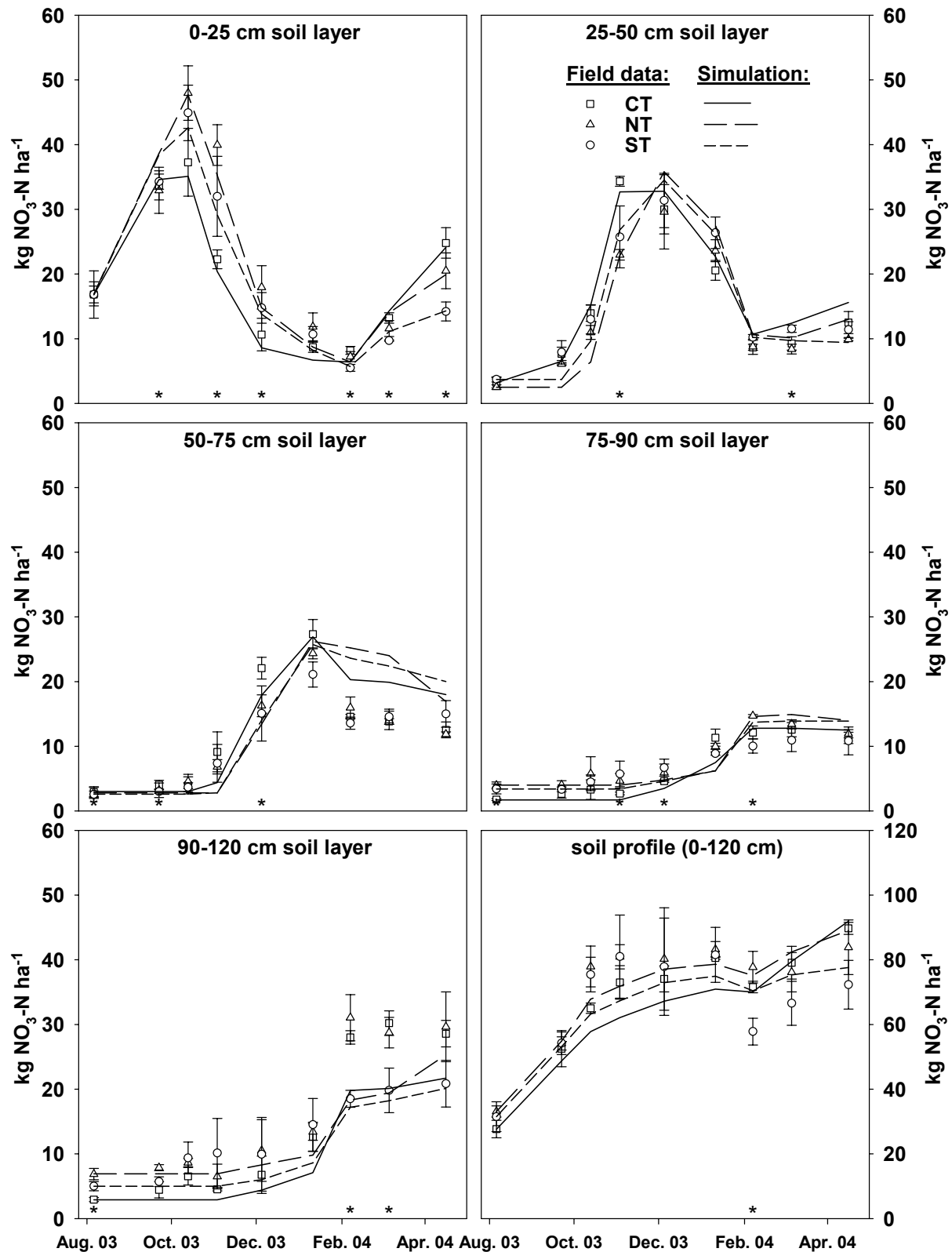


Figure 4.3: Evolution of observed (symbols) and simulated (lines) nitrate contents for the different soil layers and the whole soil profile of the different tillage treatments. Verticals bars represent standard errors. Dates labelled with * indicates significant differences among treatments at the 5 % significance level.

4.3.2 Simulation of water and nitrate contents

The simulation of gravimetric water and nitrate contents are also shown in Figure 4.2 and Figure 4.3. For all tillage treatments, a good agreement is found between the observed and simulated values (Table 4.4). The average mean difference (MD) and mean root square error (RMSE) between measured and simulated water content are, respectively, -0.17 and 1.38 g H₂O 100 g⁻¹ soil. Regarding the nitrate content, the average MD and RMSE values are 0.29 and 3.7 kg N ha⁻¹, respectively. The modelling efficiency (EF) is higher for the nitrate content (0.87 on average) than for the water content (0.82 on average). The latter is probably due to the larger temporal variation of the measured nitrate compared to the water content. Despite the overall good agreement of the simulated and measured values, some discrepancies are observed. Firstly, for the 0-25 cm soil layer, the model tends to overestimate the water content during autumn. Secondly, the overestimation of the nitrate content in the 50-75 cm soil layer and the underestimation for the 90-120 cm soil layer during late winter and spring suggests that the simulated nitrate transfer to the deeper soil layers is too slow.

Table 4.4: Statistical criteria for the global evaluation of the LIXIM model to simulate data for the different experimental treatments.

	Soil water content			Mineral N content		
	EF ¹	MD ¹	RMSE ¹	EF	MD	RMSE
	-	g H ₂ O 100 g ⁻¹ soil		-	kg N ha ⁻¹	
CT	0.777	-0.73	1.61	0.947	-0.59	2.27
CT _{rep}	0.823	-0.37	1.39	0.872	0.94	3.53
NT	0.832	0.29	1.28	0.832	0.38	4.99
NT _{rep}	0.830	0.11	1.23	0.831	0.43	4.30
ST	0.862	-0.13	1.39	0.862	0.29	3.41

¹ EF, modelling efficiency; MD, mean difference, RMSE, root mean square error

4.3.3 Calculation of N mineralisation

The calculated N mineralisation vs. calendar days for the different tillage treatments is presented in Figure 4.4. Cumulative N mineralisation is not significantly different ($p < 0.05$) between the tillage treatments for each sampling date. Averaged over the different tillage

treatments, cumulative mineralisation over the whole measurement period is $67.4 \text{ kg N ha}^{-1}$. LIXIM calculates the largest N mineralisation for autumn and spring. The measured variations of N stocks are in agreement with the simulated variations of N stocks for the period without N leaching at the bottom of the 0-120 cm soil profile (Figure 4.5). Nevertheless, the small bias between the measured and simulated variations of N stocks suggests that the model tends to slightly underestimate soil N mineralisation.

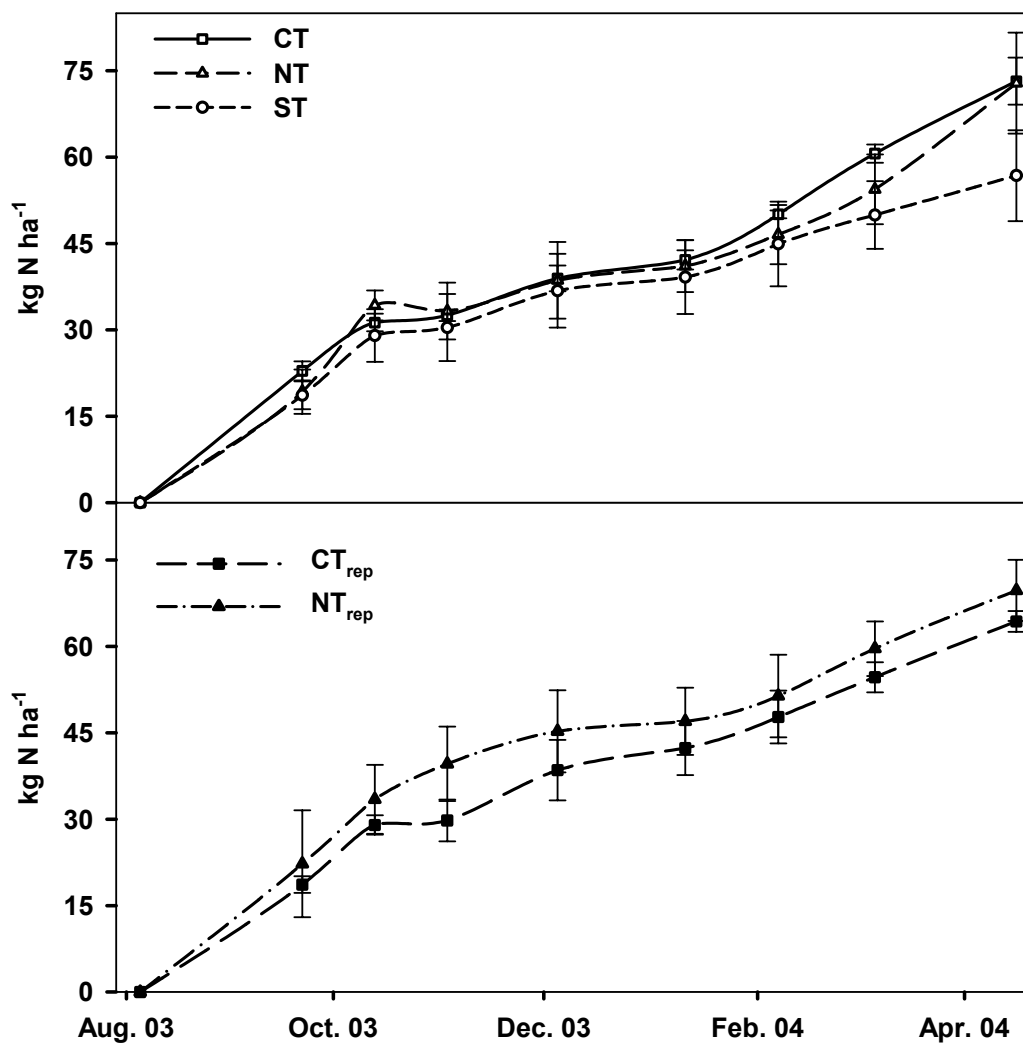


Figure 4.4: Cumulative amounts of mineralised N in the 0-120 cm soil profile calculated by LIXIM expressed in calendar days for the different experimental treatments. Vertical bars represent standard errors. CT_{rep} and NT_{rep} represent semi-repetitions of CT and NT. Crop residues were exported from 1983 to 1994 in CT_{rep} and NT_{rep}, but never exported in CT and NT.

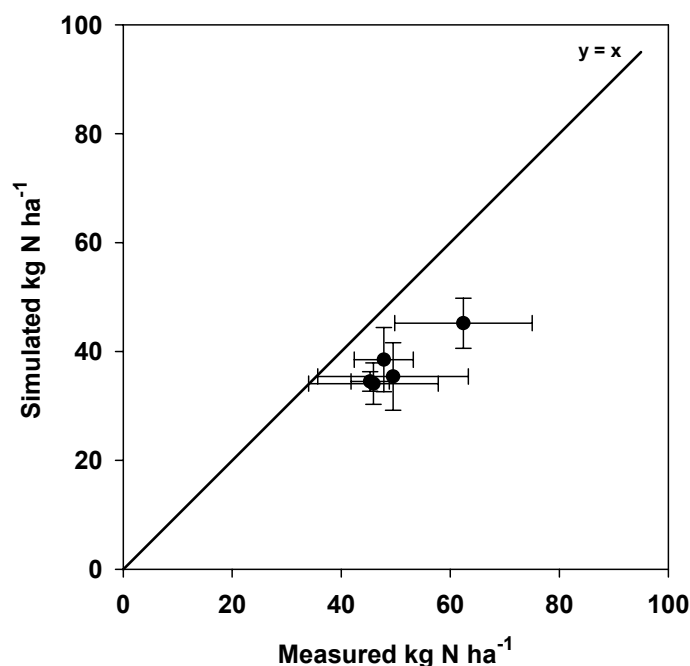


Figure 4.5: Comparison between measured and simulated variation of N stocks in the whole soil profile during period without N leaching at the 120 cm soil depth for the different experimental treatments. Vertical bars represent standard errors.

The cumulative N mineralisation against normalised days (15°C and water content at field capacity) is plotted in Figure 4.6. For each tillage treatment, the ‘in situ’ potential N mineralisation rate (V_p) was calculated as the average slope of this curve. These V_p values are not statistically different between treatments (Table 4.5). Averaged over all tillage treatments, the mean V_p value is $0.582 \text{ kg N ha}^{-1} \text{ nd}^{-1}$. The potential mineralisation rates obtained by incubating soil samples under controlled conditions in the laboratory were also not significantly different between CT ($0.497 \text{ kg N ha}^{-1} \text{ nd}^{-1}$) and NT ($0.494 \text{ kg N ha}^{-1} \text{ nd}^{-1}$) (Figure 4.6). However, the mean potential N mineralisation rate obtained from laboratory incubations ($0.496 \text{ kg N ha}^{-1} \text{ nd}^{-1}$) is slightly smaller than the corresponding value calculated by LIXIM ($0.582 \text{ kg N ha}^{-1} \text{ nd}^{-1}$).

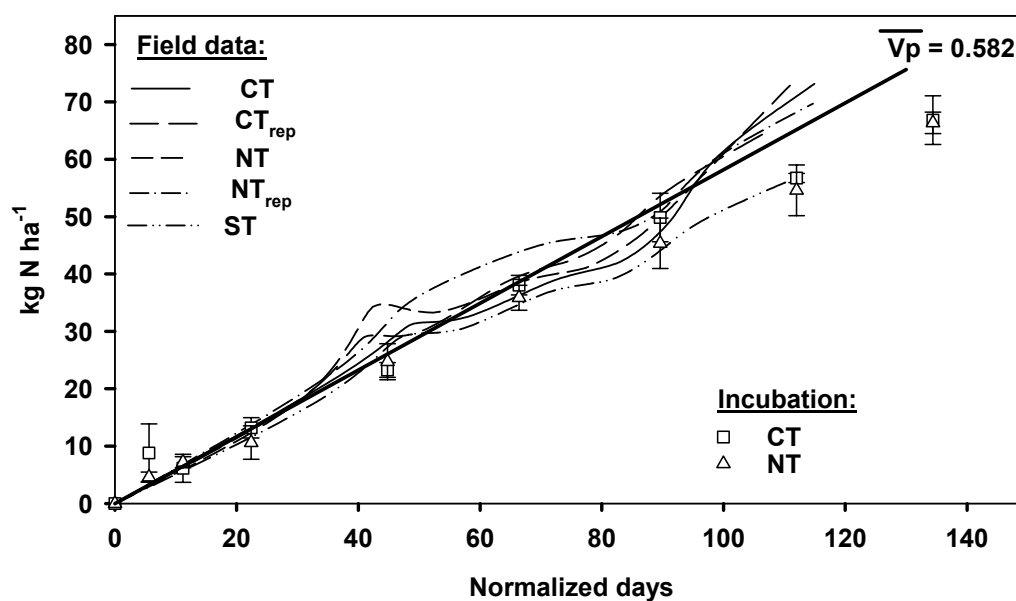


Figure 4.6: Cumulative amounts of mineralised N calculated by LIXIM against 'normalised time' for the different experimental treatments compared with laboratory incubation data. Vertical bars represent standard errors.

Table 4.5: Mean values of the 'in situ' potential N mineralisation rate (V_p) expressed in normalised days and calculated by the LIXIM model for the different experimental treatments between August 2003 and April 2004.

	Vp	SD ¹	Mean classification ²
	kg N ha ⁻¹ nd ⁻¹		
CT	0.596	0.045	a
CT _{rep}	0.583	0.031	a
NT	0.615	0.087	a
NT _{rep}	0.602	0.082	a
ST	0.515	0.102	a

¹ SD, standard deviation, 3 replicates

² means followed by the same letter are not significantly different at 5 % significance level

4.3.4 Calculation of water drainage, N leaching and water evaporation

The cumulative amount of calculated evaporation over the whole measurement period is smaller for NT and ST compared to CT plots (Table 4.6). Only from October 2003 to April 2004, the model calculates water drainage and N leaching at the bottom of the 0-120 cm soil profile. The cumulative amount of water drainage over this period amounts to 145-185 mm and NT presents the largest amount of water drainage. The calculated amount of N leaching is 11 to 24 kg N ha⁻¹. The non-tilled and superficially tilled plots tend to have more N leaching than the conventionally tilled plots.

Table 4.6: Cumulative values of evaporation, drained water and leached N at the 120 cm soil depth calculated by the LIXIM model for the different tillage treatments.

	Evaporation ^{1,2}		Drained water ^{1,2}		Leached N ^{1,2}	
	mm		mm		kg N ha ⁻¹	
CT	92.9 (3.7)	a	163.0 (3.5)	bc	15.4 (0.6)	bc
CT _{rep}	100.6 (1.8)	a	145.0 (6.9)	c	10.8 (1.0)	c
NT	57.8 (1.7)	b	185.0 (7.5)	a	23.6 (0.7)	a
NT _{rep}	62.6 (0.4)	b	166.0 (9.2)	b	19.1 (2.7)	ab
ST	67.3 (25.1)	b	162.7 (8.5)	bc	20.8 (5.1)	ab

¹ mean of 3 microplots, standard deviation in brackets

² means followed by the same letter are not significantly different at 5 % significance level

4.4 Discussion

4.4.1 Water and nitrate content

The tillage systems had little or no effect on the water and nitrate content in the soil profile. The evolution of the water content in bare soils is a result of the balance between rainfall and water transfer through drainage, run-off and evaporation. For NT, LIXIM calculated a smaller cumulative amount of evaporation, a larger amount of water in the 0-120 cm soil profile and a larger quantity of cumulative water drainage at the bottom of the soil profile compared to the CT systems. These calculations are in accordance with literature. Firstly, in NT, water losses

by soil evaporation and runoff (Fuentes *et al.*, 2003; Pekrun *et al.*, 2003) are reduced due to the presence of crop residues on the soil surface. Note that, in our study, runoff was negligible because slopes were almost zero. Secondly, the more continuous macropore network in NT (Shipitalo *et al.*, 2000; Anken *et al.*, 2004) induces a larger amount of preferential flow, causing a faster transfer of water to the deeper soil layers and thus faster drainage at the bottom of the soil profile.

Franzluebbers *et al.* (1995), Zihlmann *et al.* (2002) and Fuentes *et al.* (2003) observed, in accordance with our study, a comparable evolution and magnitude of soil nitrate contents between different tillage systems. Zihlmann *et al.* (2002) observed small seasonal differences in the nitrate content between the tillage treatments like a delayed N mineralisation in spring in NT compared to CT. Papini *et al.* (2002) reported for a loamy soil generally larger nitrate contents for ST compared to CT, which they attributed to larger N mineralisation. They also found that the differences between those tillage systems highly depended on both the specific climate and the cropping system. In fact, inorganic N dynamics in soils are the result of the interaction of different biological processes and physical transfers. The soil supplies mineral N by soil organic N mineralisation (i.e. the difference between gross N mineralisation and gross N immobilisation). Mineral N is lost from the soil system by N leaching, N uptake by plants and gaseous N losses. These gains and losses of mineral N are determined by several factors (e.g. pedoclimatic conditions, management of crop residues, N fertilisation, soil structure, organic N stocks).

4.4.2 N mineralisation

A lot of studies (Carter and Rennie, 1982; Rice *et al.*, 1986; Sainju *et al.*, 2002) have shown that potential N mineralisation, measured during soil incubation under controlled temperature and moisture conditions, is larger for the superficial soil layer of NT than of tilled systems. However, when N potential mineralisation is evaluated or re-computed for the depth of the tilled layer, effects of tillage systems on N mineralisation or the amount of mineralisable N are negligible (Carter and Rennie, 1982; Rice *et al.*, 1986; Doran, 1987; Sainju *et al.*, 2002) except for Franzluebbers *et al.* (1995). For the 0-25 cm soil layer in our study, potential N mineralisation measured under laboratory conditions was also similar between the two tillage systems. A larger potential N mineralisation of the 0-5 cm layer in NT was compensated by a smaller potential N mineralisation of the 5-25 cm soil layer in NT (not shown). Moreover, the 'in situ' potential N mineralisation rate obtained by LIXIM for field conditions was also comparable between the different tillage systems. This 'in situ' potential N mineralisation rate was not a constant over time. It can fluctuate within a year due to the influence of harvest

residues, rhizodeposition and changes in the type, quantity and activity of the microbial biomass (Franzluebbers *et al.*, 1995; Sainju *et al.*, 2002).

Differences in soil temperature and soil moisture between tillage systems may play an important role in N mineralisation observed under field conditions. Germon *et al.* (1994) and Martens (2001) concluded that larger temperatures in CT after tillage and in spring may induce a temporary larger 'in situ' mineralisation in CT compared to NT. During our study period, no large differences in water content and soil temperature were observed between the tillage systems. Only in August, soil temperature in the 0-25 cm layer was higher in CT than NT (Chapter 5).

The reduction of tillage induces changes in the location of crop residues. In our study, crop residues were situated on the soil surface in all of the three tillage treatments from August until November (wheat harvest in July), but, at the end of November, crop residues in CT and ST were incorporated into the soil. No effect was observed of this residue incorporation on the evolution of mineral N and N mineralisation. This is in contradiction with Mary *et al.* (1999) who observed (under similar pedoclimatic conditions) that larger gross immobilisation in systems with incorporated wheat residues, resulted in a lower net N mineralisation compared to systems with surface wheat residues. The gross N immobilisation during surface residue decomposition is indeed limited by the available soil mineral N in the upper few centimetres of the soil (Mary *et al.*, 1996). However, the decomposition of incorporated residues in our study was probably also seriously limited by the amount of available soil mineral N. First, the 0-25 cm layers of CT and ST only had a small amount of mineral N at the moment of residue incorporation at the end of November. Secondly, after soil tillage in CT, residues were situated in groups of residues between adjacent furrows so that the decomposer microbial biomass immobilised only the mineral N near the incorporated residues and not all the soil mineral N (Gaillard *et al.*, 1999).

4.4.3 N losses

In our study, the calculated N losses by N leaching were 53-77 % larger for NT than for CT due to 13-14% larger water drainage and probably more leaching of the mineralised N by preferential flow. There is no consensus in literature on the effect of tillage on N leaching (Elliott and Coleman, 1988; Germon *et al.*, 1994; Shipitalo *et al.*, 2000; Pekrun *et al.*, 2003). Shipitalo *et al.* (2000) and Pekrun *et al.* (2003) reported that N leaching by preferential flow is larger in NT than CT, but also that the leaching of nitrate located within the small pores of the soil matrix may be higher in CT than NT. The effect of preferential flow on N leaching is large when considerable amounts of mineral N are present in large pores and when the rain intensity is high and the soil profile wet.

Regarding gaseous N losses, ammonia volatilisation and nitrous oxide losses are generally more important in no-till systems (Martens, 2001). Indeed, larger N₂O emissions were measured for NT compared to CT plots for the site studied in our work, although the differences were not significant due to large spatial variability (Chapter 5). However, the quantities of N lost as N₂O were negligible from an agronomical point of view. The volatilisation of NH₃ at our study site was probably negligible because the N fertiliser had been applied 4 months before the start of this experiment.

4.4.4 N availability

The N availability under field conditions, in our study, was evaluated by calculating N mineralisation with the LIXIM model for a period with bare soil after wheat harvest. However, in most of the published studies, soil N availability under field conditions is evaluated with data on crop yield or N uptake by plants. Between 1976 and 2004, similar N fertiliser additions resulted in comparable maize and wheat yields between the different tillage systems at our study site (Labreuche *et al.*, 2003), which is in good agreement with the comparable soil N supply calculated by LIXIM. Using the crop yield or N uptake approach, several authors observed — in accordance with our study — no differences in N availability between tillage systems: plant N uptake and/or crop yields were similar in tilled and non-tilled systems (Meek *et al.*, 1994; Kandeler *et al.*, 1999; Salazar-Sosa *et al.*, 2002; Steinbach *et al.*, 2004). However, many authors also report reduced N uptake and/or plant yields in no-till systems (Sainju *et al.*, 2002; Brye *et al.*, 2003; Kushwaha and Singh, 2005). Soon *et al.* (2001) reports higher plant N uptake in NT systems. Germon *et al.* (1994) found that, in most cases, crop yields are lower in NT than CT at low fertiliser doses and similar in both tillage systems at high fertiliser rates due to a delayed N mineralisation in spring in NT. Finally, plant yield and/or N uptake by plants is only partially related to soil N availability because it may also be influenced by water availability, anaerobic conditions, soil temperature, soil structure, plant pathogens or predators and the interactions of all these factors (Germon *et al.*, 1994; Martens, 2001; Fuentes *et al.*, 2003). This means that the differences observed in crop yield and N uptake between tillage systems may not reflect differences in N availability but differences in the other determining factors.

4.5 Conclusions

The effect of soil tillage on the soil mineral N supply under farming conditions was evaluated for three different tillage systems differentiated over 32 years: conventional (CT), superficial (ST) and no-tillage (NT). From August 2003 to spring 2004, dynamics of soil mineral N were measured every three to four weeks in bare field plots between wheat and maize. Soil N mineralisation and N leaching were calculated using the LIXIM model by simulating the water and mineral-N profiles measured in several soil layers at different dates. Afterwards, N mineralisation rates, expressed as a function of calendar days or normalised days at 15°C and soil moisture content at field capacity, were calculated. In addition, potential N mineralisation in CT and NT was determined by soil incubations under controlled laboratory conditions. The evolution of the water and nitrate content in different layers of the 0-120 cm soil profile was comparable between the no-tillage, superficial tillage and conventional tillage systems. The by LIXIM calculated N mineralisation rates expressed in calendar or normalised days were not significantly different between the three tillage systems. Potential mineralisation determined in laboratory incubations of soil sampled from CT and NT plots was also not significant different between the tillage systems. The comparable supply of mineral N by the soil in the different tillage systems suggests that fertiliser management does not need to be differentiated between the long-term tillage systems for the studied combination of soil, climate and cropping rotation.

CHAPTER 5: DETERMINANTS OF CO₂ AND N₂O FLUXES IN LONG-TERM NO-TILLAGE AND CONVENTIONAL TILLAGE SYSTEMS

5.1 Introduction

Fluxes of CO₂ and N₂O from agricultural soils are the result of complex interactions between climate and several biological, chemical and physical soil properties. Tillage systems may affect all these soil properties and therefore influence the release of these greenhouse gases. The implementation of no-tillage (NT), a management technique without soil tillage, has been proposed to reduce CO₂ emissions from the soil as larger C stocks are generally observed in NT than in conventionally mouldboard ploughed soils (CT) (Tebrügge and Düring, 1999; Balesdent *et al.*, 2000; Yang and Kay, 2001; Deneff *et al.*, 2004). The mouldboard ploughing technique ensures soil inversion and full incorporation of residues and provides the most intensive tillage. There is no consensus in the literature on the differences in CO₂ emissions between no-tillage and mouldboard ploughing systems. Some authors report similar CO₂ fluxes under both NT and CT (Fortin *et al.*, 1996; Kessavalou *et al.*, 1998b; Aslam *et al.*, 2000) whereas Hendrix *et al.* (1988) measured generally larger CO₂ emissions in NT. Another group of authors reported larger CO₂ emissions for some periods and smaller emissions for other periods under NT (Ball *et al.*, 1999; Vinten *et al.*, 2002). Finally, significantly smaller CO₂ emissions for NT than in CT were measured during the short period after tillage (Reicosky and Lindstrom, 1993; Dao, 1998; Kessavalou *et al.*, 1998b; Alvarez *et al.*, 2001; Al-Kaisi and Yin, 2005).

Differences in CO₂ emissions between tillage systems may not be the sole result of short-term effects of tillage but instead the combined effect of short- and long-term effects. Short-term effects of tillage result from the physical soil disturbance occurring during ploughing and the location of crop residues. Long-term effects include the effect of changes in physical, chemical and biological soil properties after several years of no-tillage. Most authors only measured the short-term effects, since they studied the tillage effect on CO₂ emissions on sites that had received different tillage treatments for less than 5 years (Fortin *et al.*, 1996; Aslam *et al.*, 2000; Vinten *et al.*, 2002; Al-Kaisi and Yin, 2005) and even on sites that had been

differentiated solely during the year of measurement (Reicosky and Lindstrom, 1993; Ball *et al.*, 1999; Alvarez *et al.*, 2001). Only Hendrix *et al.* (1988), Dao (1998) and Kessavalou *et al.* (1998b) provide information on the combined effect of short- and long-term effects of mouldboard ploughing and no-tillage on CO₂ emission. However, Dao (1998) only obtained measurements during the period directly after tillage and Hendrix *et al.* (1988) and Kessavalou *et al.* (1998b) recorded weekly or monthly CO₂ emissions. We did not find any report of long-term trials in which CO₂ emission was measured continuously in other periods than directly after tillage.

The emission of CO₂ from soil is a product of the decomposition of soil organic matter (SOM) and crop residues and the respiration of plant roots. SOM decomposition depends on several factors including soil temperature, soil moisture content, amounts of organic C and the specific decomposition potential of soil C (Rodrigo *et al.*, 1997). The decomposition of crop residues is affected by additional factors such as the quality and location of the residues and soil nitrate content (Garnier *et al.*, 2003). Under temperate climates, higher temperatures and soil moisture will both result in larger SOM and crop residue decomposition provided that the decomposition is not limited by anaerobic conditions as in excessively wet soils. During summer, CT soils often present higher temperatures with larger diurnal variations than NT soils (Fortin *et al.*, 1996; Dao, 1998; Kessavalou *et al.*, 1998b; Alvarez *et al.*, 2001). Larger water contents are reported for NT especially in the 0-5 cm layer (Fortin *et al.*, 1996; Dao, 1998; Alvarez *et al.*, 2001; Al-Kaisi and Yin, 2005). This means that climatic variables partly control the differences in CO₂ emissions between CT and NT.

The N₂O emissions from agricultural soils are smaller than the CO₂ emissions when expressed in molar quantities, but the warming potential of N₂O is 296 times greater than that of CO₂ (IPPC, 2001) and N₂O contributes to the destruction of stratospheric ozone. Although there is no consensus regarding CO₂ emissions, most authors report generally larger N₂O fluxes under NT than under CT (MacKenzie *et al.*, 1998; Ball *et al.*, 1999; Vinten *et al.*, 2002; Baggs *et al.*, 2003). However, Kessavalou *et al.* (1998b) and Choudhary *et al.* (2002) report similar N₂O emissions under both NT and CT. Nitrous oxide is a product of denitrification and nitrification (Hénault *et al.*, 2005). Increased N₂O emission in NT may be the result of changes in water-filled pore space, available C for denitrifiers, soil temperature and mineral N concentration.

The aim of our work was to quantify CO₂ and N₂O emissions from long-term mouldboard ploughing and no-tillage systems after 32 years of differentiation. Soil temperature, soil moisture, soil mineral N, soil potential for CO₂ emission, rainfall, potential evaporation and amounts of SOM and crop residues were measured simultaneously to explain differences in gas emissions between the two tillage systems.

5.2 Materials and methods

5.2.1 Study site

The study site is located at Boigneville (Northern France, 48°33'N, 2°33'E). The soil is an Haplic Luvisol (FAO *et al.*, 1998). Boigneville has a long-term average temperature of 10.8°C and an annual precipitation and potential evapotranspiration (PET) of 650 mm and 730 mm, respectively. The average temperature, total precipitation and PET were 12.1°C, 572 mm and 808 mm, respectively, during the measurement year, with higher temperatures and PET between mid-May and mid-August compared to the long-term average.

Two tillage systems, i.e. no-tillage (NT) and conventional tillage (CT), had been imposed since 1970. Before differentiation, the two systems had been mouldboard ploughed to 30 cm depth annually. Tillage was then totally suppressed in NT whereas CT was mouldboard ploughed to 20 cm depth every year. The structural soil profiles are described in Chapter 3 and Appendix A. The site has a maize-wheat rotation. Maize is sown at the end of April and usually harvested mid-October, nearly immediately followed by tillage for CT and sowing of wheat. After the wheat has been harvested at the end of July, the soil is left bare, ploughed in November for CT and sown with maize in April. The crop residues are returned to the soil in both systems, remaining on the soil surface after harvest until incorporation by tillage in CT, but always remaining on the soil surface in NT.

The gas measurements were performed during two different periods on four different plots (212, 215, 223 and 224) at the Boigneville site. When plots 212 (CT) and 215 (NT) were in the wheat year, plots 224 (CT) and 223 (NT) were in the maize year of the maize-wheat rotation. Period I refers to field plots 212 and 215 monitored under maize cultivation from the end of April 2003 until July 2003. The maize was sown on April 17th 2003 and received 158 kg N ha⁻¹ ammonium-nitrate on May 12th 2003. The plots were irrigated three times with 25 mm of water between the end of June and July. Period II refers to field plots 223 and 224 which were followed from July 30th 2003 until April 13th 2004 during the period between wheat and maize, with tillage on November 19th 2003 in CT. The soil remained bare throughout period II. Periods II_a and II_b refer respectively to the parts of period II before and after tillage. The general soil properties of the different field plots are presented in Table 5.1.

Table 5.1: Selected soil properties of the conventionally tilled (CT) and no-tillage (NT) field plots for measurement periods I and II at Boigneville.

		Period I April– July 2003			Period II July 2003 – April 2004					
		CT Plot 212	NT Plot 215		CT Plot 224			NT Plot 225		
Soil layer	cm	0-20	0-5	5-20	0-5	5-20	20-25	0-5	5-20	20-25
Clay (< 2 μm)	g 100 g ⁻¹ soil	22.0	18.8	20.8	26.5	26.6	26.7	22.4	24.2	25.2
Silt (2-50 μm)	g 100 g ⁻¹ soil	69.6	69.9	67.9	65.3	65.1	64.7	67.2	66.3	65.5
Sand (> 50 μm)	g 100 g ⁻¹ soil	8.4	11.3	11.3	8.2	8.3	8.6	10.4	9.5	9.3
pH (water)	-	6.0	4.8	6.0	6.6	6.6	6.7	5.1	6.3	7.0
CaCO ₃	g 100 g ⁻¹ soil	0.1	< 0.1	0.1	0.1	< 0.1	< 0.1	0.1	< 0.1	< 0.1
CEC	cmol _c kg ⁻¹ soil	11.7	12.3	11.0	13.5	13.8	13.9	13.3	12.8	13
Organic C ¹	g 100 g ⁻¹ soil	1.10	2.46	0.94	1.10	1.11	1.07	2.26	0.95	0.77
Organic N	g 100 g ⁻¹ soil	0.108	0.212	0.103	0.112	0.113	0.109	0.193	0.101	0.085
C to N ratio	-	10.2	11.6	9.1	9.9	9.8	9.8	11.7	9.4	9.1

¹ The 20-25 cm soil layers of CT and NT for period I contained 0.76 and 0.73 g C 100 g⁻¹ soil and 0.083 and 0.086 g N 100 g⁻¹ soil respectively

The average bulk density over the two periods was 1.132 and 1.517 g cm⁻³ for the 0-5 and 5-25 cm soil layer of NT and 1.320 and 1.517 g cm⁻³ for the 0-20 and 20-25 cm soil layer of CT.

5.2.2 CO₂ and N₂O measurements

CO₂ was measured using automatic chambers (0.49 m², 0.225 m high) connected to an infrared analyzer (Illustration Appendix B). Four chambers for each tillage treatment were pressed 0.10 m into the soil. These gas chambers were placed perpendicular to the direction of the field operations and covered the whole width of the combine harvester in order to take into account the horizontal heterogeneity of both soil structure and the distribution of crop residues induced by tillage and harvest operations. This heterogeneity was greater perpendicular, rather than parallel, to the direction of the field operations. The soil surface in and around the CO₂ chambers was always kept bare in order to avoid plant-emitted CO₂. The chambers closed 4 times a day for 5 min. at 4:00, 10:00, 16:00 and 22:00 GMT. CO₂ emissions were measured during these closure periods and the daily CO₂ emissions were calculated each day as the average of these four measurements. Outside these closure periods, the chambers were left open in order to expose the soil inside the chambers to realistic weather conditions.

N₂O emissions were quantified on average every three weeks with non automatic chambers (0.5 m diameter and 0.15 m high) as described by Hénault *et al.* (2005). Eight chambers per tillage treatment were pushed approximately 0.08 m into the soil between the maize plants for period I and into the bare soil for period II (Illustration Appendix B). The atmosphere inside the chambers was sampled with a needle after 0, 1, 2 and 3 hours of closure and stored in 10 ml Venoject tubes sealed with silicones and wax. The gas samples were analysed using a gas chromatograph equipped with a porapak Q column connected to an electron capture detector (Shimadzu GC8AIE, Kyoto, Japan).

5.2.3 Measurement of determining factors

5.2.3.1 Climatic data

Rainfall, potential evaporation and air temperature were obtained from an automatic weather station on the experimental site. Soil temperature was recorded hourly in both tillage systems with thermocouples at 2.5 and 12.5 cm soil depths.

5.2.3.2 Soil mineral N and gravimetric water content

During period I, the mineral N and gravimetric water content (24 hours at 105°C) of the 0-5 and 5-20 cm soil layers were measured at each N₂O sampling date for both the cropped soil around the N₂O chambers and the bare soil around the CO₂ chambers. During period II, the mineral N of the 0-5 and 0-25 cm soil layers was measured monthly as described in Chapter 4. The gravimetric water content for the 0-5 and 5-20 cm soil layers was measured several times during period II. The mineral-N was measured during both periods after extraction (30 min. at 20°C) of 100 g fresh soil with 200 ml of 1 M KCl. The NO₃⁻ and NH₄⁺ in the soil extracts were analysed by a continuous flow colorimeter (TRAACS 2000, Bran+Luebbe, Norderstedt, Germany) using an adaptation of the method proposed by Kamphake *et al.* (1967) for nitrate and nitrite and a method derived from Krom (1980) for ammonium.

5.2.3.3 Amounts of crop residues at different points in time

The amounts of crop residues at different points in time were only measured during period II. The measurement dates were July 29th 2003 (beginning of period II), November 18th 2003 (day before tillage) and April 13th 2004 (end of period II). The crop residues in CT consisted only of recent wheat residues (harvested on July 16th 2003) whereas the crop residues in NT were a combination of recent wheat residues and older remaining weathered maize and wheat residues. Three microplots were set up within each tillage treatment. Eight open square frames (0.25 m²) were placed on the soil surface next to each other and perpendicular to the direction

of the field operations. The eight frames covered the entire width of the combine harvester to take the heterogeneity of residue localization after harvest into account. The surface residues and wheat stubble in each frame were removed from the field, washed if necessary and dried (80°C). On April 13th 2004, the procedure for CT was adapted so that residues previously incorporated into the 0-20 cm soil layer by the tillage operation on November 19th 2003 could be sampled. Soil from the 0-20 cm soil layer was sieved through 250 µm to extract the residues. C and N concentrations of the residues were determined by dry combustion using an NA 1500 elemental analyser (Fisons Isochrom). The biochemical composition of the residues was determined using a modification of the method proposed by Van Soest (Linères and Djakovitch, 1993). Measurements of ¹³C were obtained using a NA 1500 elemental analyser coupled to a mass spectrometer (Fisons Isochrom) to determine the proportion of residue-C in the collected crop residues derived from maize. Maize and wheat have different natural ¹³C abundance.

5.2.3.4 C and N stocks

The C and N stocks in the different fields during both periods were determined for an equivalent soil mass of 3900 tons dry soil ha⁻¹ following the method described in Chapter 2. This soil mass corresponds to the average soil mass in the plough layer before differentiation of the two tillage treatments in 1971. The C and N in crop residues were not taken into account.

5.2.3.5 Soil CO₂ emission potential

The soil CO₂ emission potentials were measured during incubations under controlled conditions (15°C and -63 kPa) for 168 days. The CO₂ emission potentials for the 0-20 cm soil layer of the field plots for period I have already been reported (Chapter 3). For period II, soil samples were taken from the 0-5 cm, 5-20 cm and 20-25 cm layers inside the gas chambers in April 2004 and the CO₂ emission potentials were measured as in period I. The CO₂ emission potential for each soil layer was expressed in mg C kg⁻¹ soil; whereas the soil specific CO₂ emission potential was expressed in % of soil organic C.

Soil CO₂ potentials for the whole equivalent 0-25 cm soil profile of the plots for period II, were expressed in kg C ha⁻¹ and calculated on an equivalent dry mass basis of 3399 t dry soil ha⁻¹ taking into account the average bulk densities of the different soil layers. This mass corresponds to the 0-25 cm soil layer of CT and the 0-23.7 cm layer of NT. The results for the equivalent 0-20 cm soil layer were computed in the same way so that the results for periods I and II could be compared.

5.2.4 Effect of temperature, water content and nitrate concentration on CO₂ and N₂O emissions

The effect of temperature on CO₂ emissions was evaluated by the factors $f_{T,SOM}$ and $f_{T,RES}$, i.e. the daily temperature factors for the decomposition of SOM and crop residues respectively. These factors were calculated as described by Garnier *et al.* (2001):

$$f_T = \exp[B_T(T - 15)] = \frac{\Delta CO_{2,T}}{\Delta CO_{2,15^\circ C}}$$

with T the actual soil temperature and 15 the reference temperature in °C and B_T derived from Recous (1995) for SOM and crop residues. The higher the values of f_T , the better the temperature conditions for SOM and residue decomposition. The effect of diurnal variation of temperature was considered by calculating daily values for both $f_{T,SOM}$ and $f_{T,RES}$ as the mean of the f_T values for the four daily CO₂ measurement periods (i.e. at 4:00, 10:00, 16:00 and 22:00 GMT). The $f_{T,RES}$ values were calculated for the soil depth where most of the residues were situated, i.e. the 2.5 cm soil depth for surface residues and the 12.5 cm depth for incorporated residues.

The effect of water content on ‘in situ’ CO₂ emissions from SOM was determined using the moisture factor for SOM decomposition ($f_{W,SOM}$) proposed by Mary *et al.* (1999):

$$f_{W,SOM} = 0.8 \frac{\theta - \theta_{wp}}{\theta_{fc} - \theta_{wp}} + 0.2$$

with θ the measured gravimetric water content in the field and θ_{wp} and θ_{fc} the gravimetric water contents at wilting point and field capacity respectively. This equation gives the same results if the volumetric water content is used instead of gravimetric water content, since it is a ratio of water contents. For the decomposition of crop residues, no moisture factor was calculated.

For field denitrification, the response factors to soil temperature ($f_{T,DENIT}$), water-filled pore space ($f_{W,DENIT}$) and soil nitrate concentration ($f_{N,DENIT}$) were calculated using the equations from the NEMIS model developed by Hénault *et al.* (2005). Water-filled pore space (WFPS) was calculated using the gravimetric water contents and average bulk densities of the different soil layers. The above temperature ($f_{T,SOM}$) and moisture factors ($f_{W,SOM}$) for the decomposition of SOM were used to determine the effect of temperature and moisture on the N₂O emitted as a product of nitrification.

5.2.5 Statistical analysis

The analysis of variance was done using the ANOVA procedure of SAS software (SAS Institute, 2001) and the means were compared using Student-Newman-Keuls tests at $P \leq 0.05$.

5.3 Results

5.3.1 CO₂ emissions

The CO₂ emissions in CT during period I, from the end of April to June 10th, were more variable than in NT with high peaks of CO₂ emission observed in CT (Figure 5.1). After the 10th of June, the CO₂ emissions in both systems presented similar dynamics, with CO₂ emission in NT gradually becoming larger than in CT. The CO₂ emissions measured on 61% of the days in period I did not differ significantly between the two tillage systems. NT presented significantly larger emissions on 30% of the days compared to only 9 % for CT. During period II_a, NT emitted significantly more CO₂ than CT on 77 % of the days, but no significant differences were found for the remaining days. The timing of the CO₂ emissions was similar in both tillage systems, with high peaks before mid-October and rather low and constant CO₂ emissions after mid-October. During period II_b, tillage and the incorporation of residues caused an increase in the CO₂ emissions in CT, which attained the same level of CO₂ emissions as in NT. However, the CO₂ emissions did not occur at the same time in the two systems. The CO₂ emissions were not significantly different between the two tillage systems on 45% of the days and were larger in NT than in CT on 47 % of the days. During the whole period II_b after tillage, CO₂ emissions were rather low.

The cumulated CO₂ emissions for period I did not differ significantly between CT (1159 ± 198 kg CO₂-C ha⁻¹) and NT (2213 ± 135 kg CO₂-C ha⁻¹) (Figure 5.2). However, the cumulated CO₂ emissions for period II_a were considerably larger for NT (1912 ± 101 kg CO₂-C ha⁻¹) than for CT (1146 ± 92 kg CO₂-C ha⁻¹). Finally, the cumulated CO₂ emissions for period II_b, were again in the same order of magnitude in both tillage systems, though still significantly larger for NT (1006 ± 18 kg CO₂-C ha⁻¹) than for CT (816 ± 98 kg CO₂-C ha⁻¹). Over the entire measurement period, CT and NT emitted 3160 ± 269 and 4064 ± 138 kg CO₂-C ha⁻¹, respectively. These values correspond with extrapolated yearly CO₂ emissions of 3484 ± 296 and 4481 ± 152 kg CO₂-C ha⁻¹ yr⁻¹ for CT and NT.

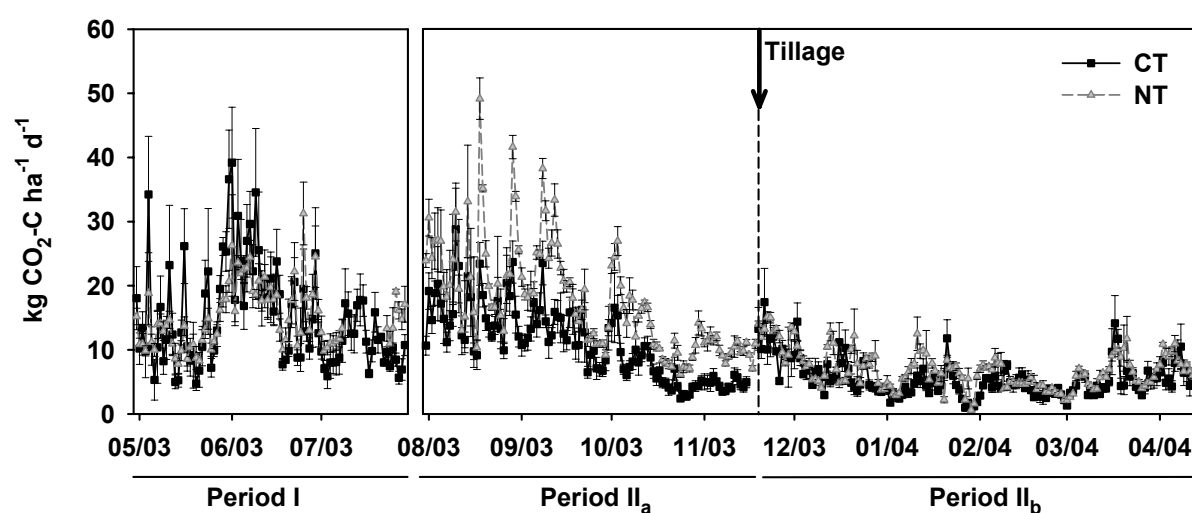


Figure 5.1: Daily CO_2 emissions in the conventionally tilled (CT) and non-tilled (NT) plots during periods I and II. Error bars indicate standard deviations.

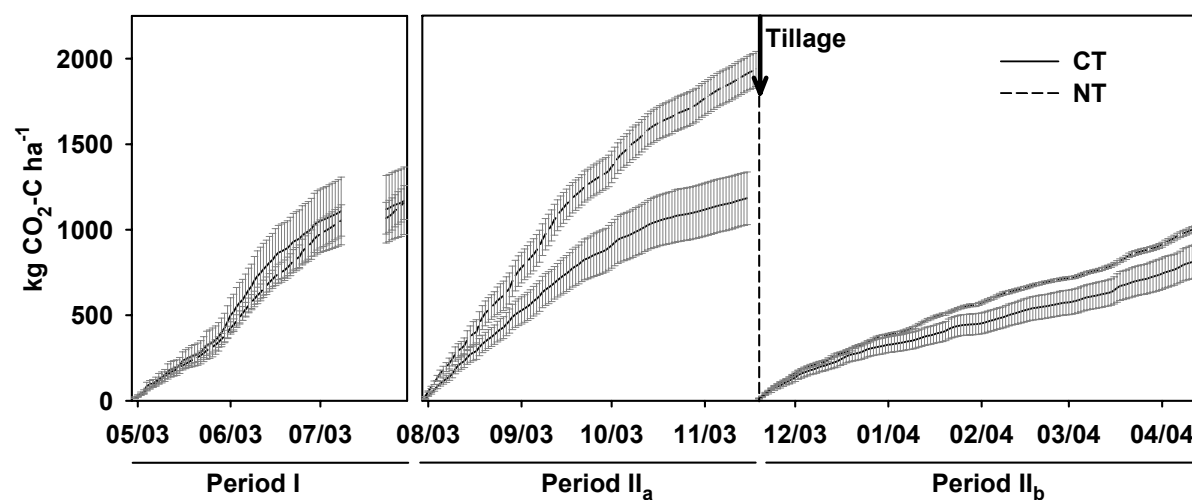


Figure 5.2: Cumulated CO_2 emissions in conventionally tilled (CT) and non-tilled (NT) plots during periods I and II. Error bars indicate standard deviations. The gap in the curves for period I is due to technical problems in the NT treatment.

5.3.2 Determining factors for CO₂ emissions

5.3.2.1 Soil CO₂ emission potential

The total soil CO₂ emission potential for the 0-25 cm soil layers inside the gas chambers during period II, did not differ significantly between CT and NT (Table 5.2). However, the distribution of this CO₂ emission potential over the 0-25 cm soil profile differed between the two systems. In NT, the CO₂ emission potential decreased with depth (59% of the total potential was situated in the 0-5 cm soil layer) whereas in CT, the distribution of CO₂ emission potential throughout the 0-25 cm soil layer was relatively homogeneous with the highest potential in the 5-20 cm soil layer. When expressed in % C present in the equivalent 0-25 cm soil profile, the specific soil CO₂ emission potentials for the equivalent 0-25 cm soil layers of NT and CT were similar. Larger soil CO₂ emission potentials were measured in the 0-20 cm layer of NT than in CT in the period I plots, unlike the period II plots.

Table 5.2: Soil CO₂ emission potentials measured at 15°C and –63 kPa during a 168-day incubation of samples from the different soil layers of the conventionally tilled (CT) and non-tilled (NT) plots for periods I and II.

	Period I		Period II	
	CT Plot 212	NT Plot 215	CT Plot 224	NT Plot 225
Soil layer	mg C kg ⁻¹ soil			
0-5 cm	$\left\{ 290 \pm 8 \right\}$	1149 ± 78 ¹	234 ± 32	1013 ± 123
5-20 cm		208 ± 20	360 ± 25	230 ± 28
20-25 cm	-	-	230 ± 31	90 ± 18
Equivalent soil layer	kg C ha ⁻¹			
0-20 cm ²	765 ± 21	1081 ± 60	867 ± 66	899 ± 61
0-25 cm ³	-	-	1042 ± 47	980 ± 71

¹ standard error

² calculated for 2640 tons dry soil ha⁻¹ corresponding to the 0-20 cm layer for CT and the 0-18.7 cm layer for NT

³ calculated for 3399 tons dry soil ha⁻¹ corresponding to the 0-25 cm layer for CT and the 0-23.7 cm layer for NT

5.3.2.2 SOM stocks

The C and N stocks in NT were generally larger (Table 5.3). However, the differences between CT and NT were only significant in the period I plots. The C and N present in both surface and incorporated residues were not taken into account because of their extreme variability over time. After 32 years of differentiation, the surplus C stock was 5.6 t C ha⁻¹ for period I and 2.4 t C ha⁻¹ for period II. This indicates a mean annual gain of 175 and 76 kg C ha⁻¹ yr⁻¹ in the NT plots followed during periods I and II, respectively.

5.3.2.3 Decomposition of crop residues

NT showed larger amounts of residue-C and -N for all sampling dates (Table 5.4). The residues in NT were a mix of recent harvested wheat residues and older weathered maize and wheat residues. The larger amounts of residue-C in NT were mainly due to larger amounts of maize-derived C. However, a larger amount of wheat-derived C was also found in the NT surface residues after the harvest in July. This larger amount of wheat-derived C in NT was probably due to the presence of older weathered wheat residues from previous years and not to larger C inputs via the recently harvested residues of the 2003 wheat. The grain yield of the wheat harvested in 2003 was 6.28 tons ha⁻¹ for CT and 6.35 tons ha⁻¹ for NT respectively (results not presented) and the grain yields for CT and NT averaged over 32 years were also comparable.

At the end of period II, 15 and 42 percent of the mass of residue-C present at harvest and 26 and 44 percent of the mass of residue-N present at harvest were recovered in CT and NT respectively. In NT, the maize residues decomposed more slowly than the wheat residues: 60 % of the initial maize-derived C was still present in April 2004 compared with only 35% of the wheat-derived C. The larger absolute decay of the crop residues (kg C or N ha⁻¹) in NT was mainly due to decomposition of the old weathered maize residues. The absolute decay of wheat residues was similar in both tillage systems.

Both the biochemical composition and the C concentration of surface residues in the two tillage systems were comparable after the July harvest (results not shown). However, the C:N ratio in the NT residues was significantly lower than in the CT residues. The C:N ratio of the CT residues decreased after incorporation into the soil, whereas the C:N ratio of the surface residues in NT remained constant throughout period II.

Table 5.3 : C and N stocks in a soil profile outside the gas chambers calculated for an equivalent mass of 3900 t dry soil in the conventionally tilled (CT) and non-tilled (NT) field plots used in period I or II.

Period	Period I				Period II			
	March 2003		October 2003		April 2004			
Treatment	CT	NT	CT	NT	CT	NT		
Plots	Plot 212	Plot 215	Plot 224	Plot 225	Plot 224	Plot 225		
Soil layer depth	cm	27.7	26.8	27.7	27.0	28.7	27.6	
Organic C	tons C ha ⁻¹	37.8 ± 2.7 ¹	43.5 ± 1.8	42.3 ± 1.1	45.1 ± 1.9	43.7 ± 1.4	45.7 ± 3.2	
Organic N	tons N ha ⁻¹	4.01 ± 0.26	4.40 ± 0.12	4.27 ± 0.11	4.55 ± 0.18	4.42 ± 0.13	4.54 ± 0.17	

¹ ± standard error

Table 5.4 : Total residue-N and -C, maize-derived C and the residue C:N ratio measured at different dates in the conventionally tilled (CT) and non-tilled (NT) field plots during period II.

Sampling date	Total N		Total C		Maize derived C ¹		C:N ratio	
	CT	NT	CT	NT	CT	NT	CT	NT
	kg N ha ⁻¹		kg C ha ⁻¹		kg C ha ⁻¹		-	
July 29 th , 2003	35.6 ± 7.1 ²	78.3 ± 12.1	3120 ± 499	5415 ± 701	0 ± 27	1557 ± 427	88 ± 7	69 ± 3
November 18 th , 2003	16.5 ± 1.8	49.2 ± 1.0	1464 ± 159	3430 ± 59	0 ± 9	1148 ± 324	89 ± 7	70 ± 2
April 13 th , 2004	9.1	34.5 ± 4.3	477	2271 ± 222	0	933 ± 37	52	66 ± 2
Decomposed residues between July 29 th and April 13 th	26.5	43.9	2644	3144	0	624	-	-

¹ Determined by the natural ¹³C abundance of the crop residues

² ± standard error

All values for July 29th and November 18th were statistically different ($P < 0.05$) between CT and NT. No statistical analyses could be done on the April 13th measurements because no standard deviation was available for the CT treatment. Crop residues were collected from the soil surface except on April 13th for the CT plot.

5.3.2.4 Soil temperature

The combination of higher soil temperatures and larger diurnal variations at 2.5 and 12.5 cm soil depth during period I, resulted in higher $f_{T,SOM}$ values for CT compared to NT in June and during a few days in July (Figure 5.3). The $f_{T,RES}$ values were generally lower than the $f_{T,SOM}$ values and there was little difference in the $f_{T,RES}$ values between CT and NT.

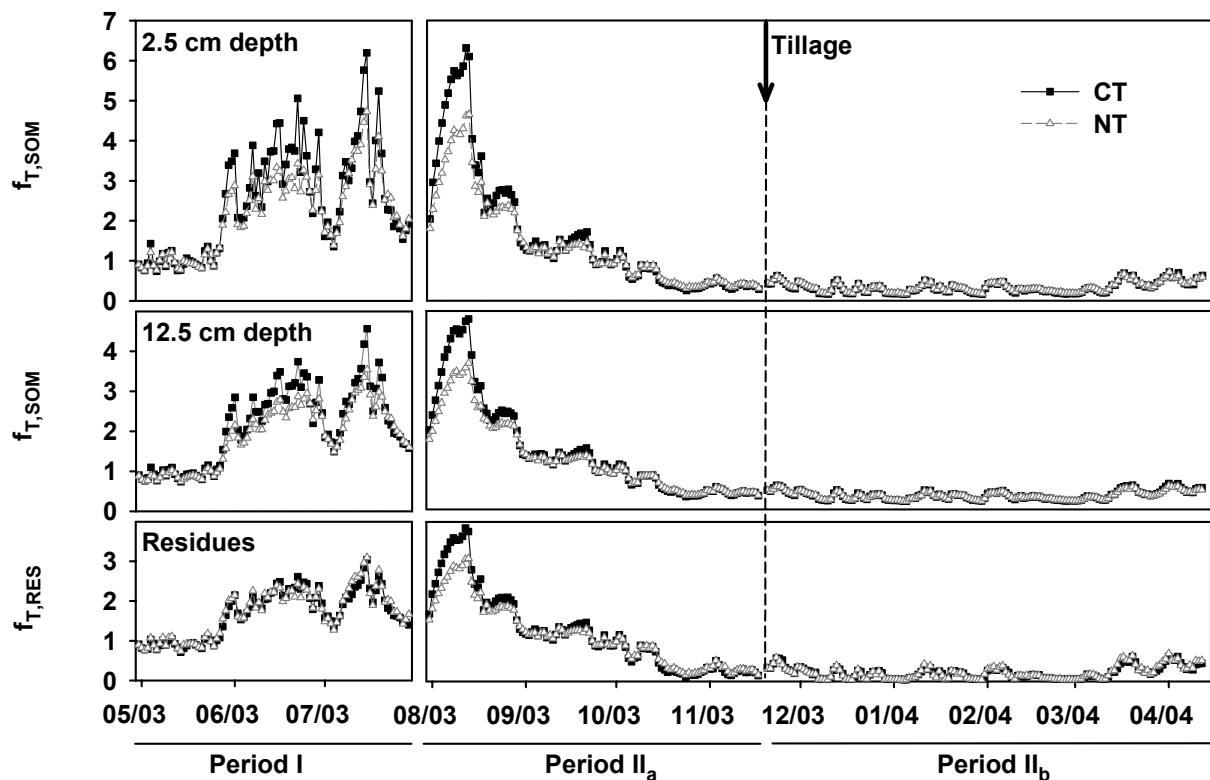


Figure 5.3: Temperature functions for SOM decomposition ($f_{T,SOM}$) at 2.5 and 12.5 cm soil depth and the location of residue decomposition ($f_{T,RES}$) in the conventionally tilled (CT) and non-tilled (NT) plots during periods I and II.

In period II_a, before the end of September, $f_{T,SOM}$ and $f_{T,RES}$ were generally higher in CT than in NT due to higher temperatures in CT. The diurnal temperature variations were similar in both tillage systems. From October onwards, the limiting effect of temperature on SOM and residue decomposition in CT and NT was similar. This temperature limitation was more pronounced for residue than for SOM decomposition in both tillage systems.

During most of the period after tillage (period II_b), the $f_{T,SOM}$ values were very low, indicating severe limitation by temperature. As in the previous period, the decomposition of residues was more limited by temperature than SOM decomposition. Similar $f_{T,SOM}$ values

were obtained for CT and NT. Residue decomposition was slightly less limited by temperature in CT than NT during cold days, whereas the opposite was true during warmer days. This was due to the temperature-buffering effect of the soil on the incorporated residues in CT.

5.3.2.5 Rainfall and soil water content

To counterbalance the temperature effect in the study of rainfall and soil moisture effects on CO₂ emissions, these latter were normalised for days with a temperature of 15°C. The daily CO₂ emissions were divided by the $f_{T,SOM}$ factor calculated for the equivalent 0-25 cm soil profile. This factor was calculated with the $f_{T,SOM}$ values at 2.5 and 12.5 cm depth taking into account the relative contribution of the CO₂ emission potentials of the 0-5 cm and 5-25 cm soil layers to the CO₂ emission potential of the entire equivalent 0-25 cm soil profile (Figure 5.4).

The normalised CO₂ emissions at 15°C, during rainfall events in the first half of period I, were smaller in CT than in NT, whereas CT presented larger normalised CO₂ emissions than NT between rainfall events. During the period after June 10th, with high PET and little rainfall, both tillage systems presented low normalised CO₂ emissions. The $f_{W,SOM}$ values for the latter period indicate that the soil water content became limiting for SOM decomposition, especially in the 0-5 cm soil layer (Figure 5.5). NT showed a generally lower limitation by soil water content than CT for both the 0-5 and the 5-20 cm soil layers.

During period II_a, rainfall induced high peaks in CO₂ emissions in both tillage systems with higher peaks in NT (Figure 5.4). Before mid-September the potential evaporation was high and the soil water content low which resulted in low $f_{W,SOM}$ values (Figure 5.5). These $f_{W,SOM}$ values were slightly lower for NT than CT. After the warm and dry summer months of 2003, the $f_{W,SOM}$ values increased to near optimal values. Additional measurements (not shown) indicated a gradual increase of the water content in all soil layers between October and December.

After tillage (period II_b), NT presented larger CO₂ emissions than CT during rainfall events. However the CO₂ emissions between rainfall events, tended to be larger in CT than in NT (Figure 5.4). The $f_{W,SOM}$ values for SOM decomposition were near optimum during period II_b except after February for the 0-5 cm soil layer of CT (Figure 5.5). This also means that the incorporated residues in CT (5-20 cm soil layer) always had optimal water conditions for residue decomposition, whereas the surface residues in NT were sometimes limited by water content between rainfall events.

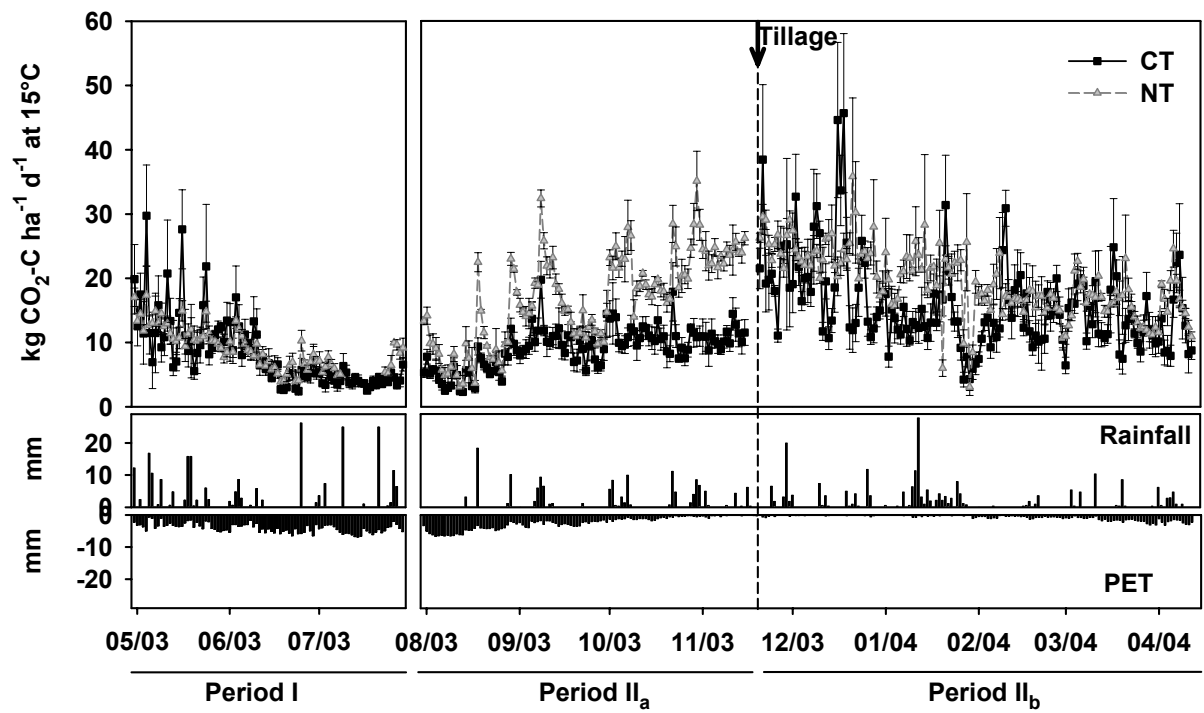


Figure 5.4: Daily CO₂ emissions for normalised days at 15°C in the conventionally tilled (CT) and non-tilled (NT) plots during periods I and II with indication of rainfall (irrigation included) events and potential evapotranspiration (PET). Error bars indicate standard deviations.

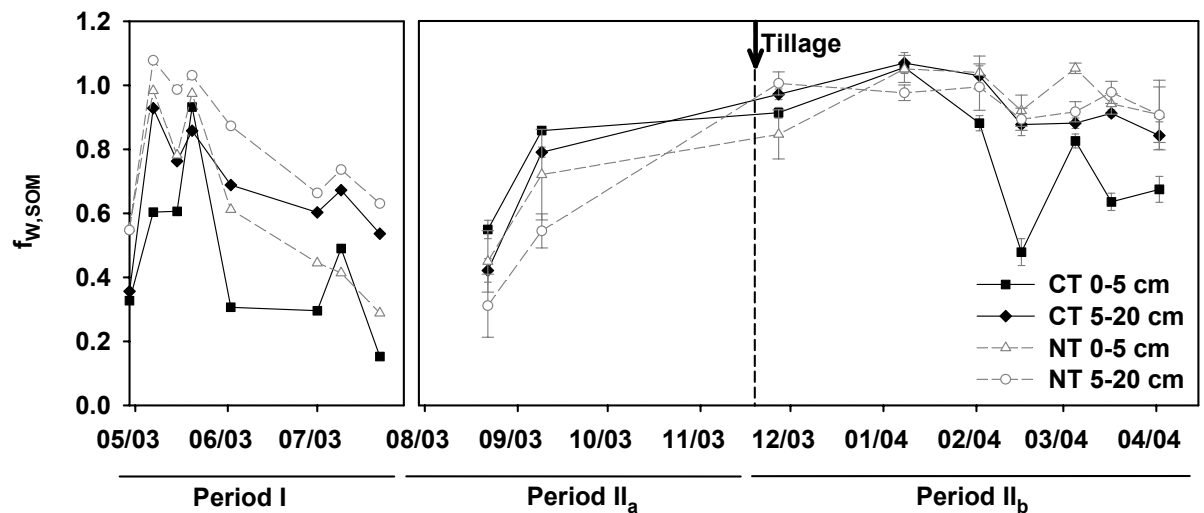


Figure 5.5: Moisture functions for SOM decomposition in the 0-5 and 5-20 cm soil layers of the conventionally tilled (CT) and non-tilled (NT) plots during periods I and II.

5.3.2.6 Soil mineral N

The soil mineral N concentration of the 0-5 cm surface layer in both CT and NT increased considerably just after the fertiliser application on May 12th 2003 and then decreased to values no lower than 50 mg N kg⁻¹ soil (results not shown). The mineral N concentration of the 5-20 cm soil layers was relatively constant and remained lower than 25 mg N kg⁻¹ soil throughout period I. The soil mineral N in the 0-5 and 5-25 cm soil layers was relatively low throughout period II (< 11 mg N kg⁻¹ except for the 0-5 cm soil layer of NT). The lowest N concentrations were found between November and February. Soil mineral N concentrations increased between August and mid-October and between March and April with a larger increase for the 0-5 cm soil layer of NT.

5.3.3 N₂O emissions

N₂O emissions presented considerable spatial variability in both CT and NT especially when emissions were large (Figure 5.6). The average N₂O emission always tended to be higher in NT than in CT. The N₂O emissions under the maize cover during period I, were significantly higher at only one of the 7 dates in NT and were usually between 1 and 23 g N ha⁻¹ day⁻¹. The largest N₂O emissions followed a fertiliser application on May 12th. During period II on bare soil, NT emitted significantly more N₂O on 8 of the 9 dates but the N₂O emissions were less than 1 g N ha⁻¹ day⁻¹ in NT except for the first 3 measurement dates.

The soil temperature, moisture content and nitrogen concentration factors affecting denitrification are presented in Figure 5.7. During period I, the $f_{T,DENIT}$ factor indicated that the temperature conditions for denitrification were slightly more favourable in CT than in NT. The factor determined by the concentration of soil mineral N ($f_{N,DENIT}$) was high after fertilisation for the 0-5 cm soil layers of both CT and NT, whereas the 5-20 cm soil layers presented low values for this factor. However, the $f_{W,DENIT}$ factor was around zero for all soil layers except for the 5-20 cm soil layer of NT. Before mid-October in period II, the $f_{T,DENIT}$ and $f_{W,DENIT}$ factors showed the same trends as in period I. From mid-October onwards, the $f_{W,DENIT}$ factor became larger in all soil layers but the $f_{T,DENIT}$ factor was extremely small. During the whole of period II, the largest $f_{N,DENIT}$ values were generally observed for the 0-5 cm layer of NT and the largest $f_{W,DENIT}$ values for the 5-20 cm soil layer of NT.

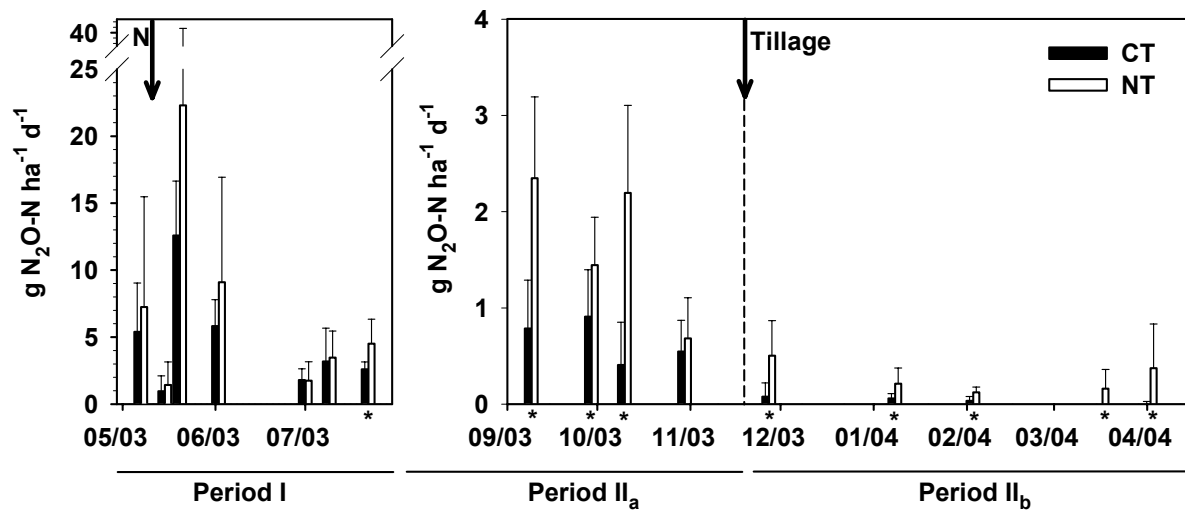


Figure 5.6: N_2O emissions in the conventionally tilled (CT) and non-tilled (NT) plots during periods I and II. * indicates significant differences at $P \leq 0.05$.

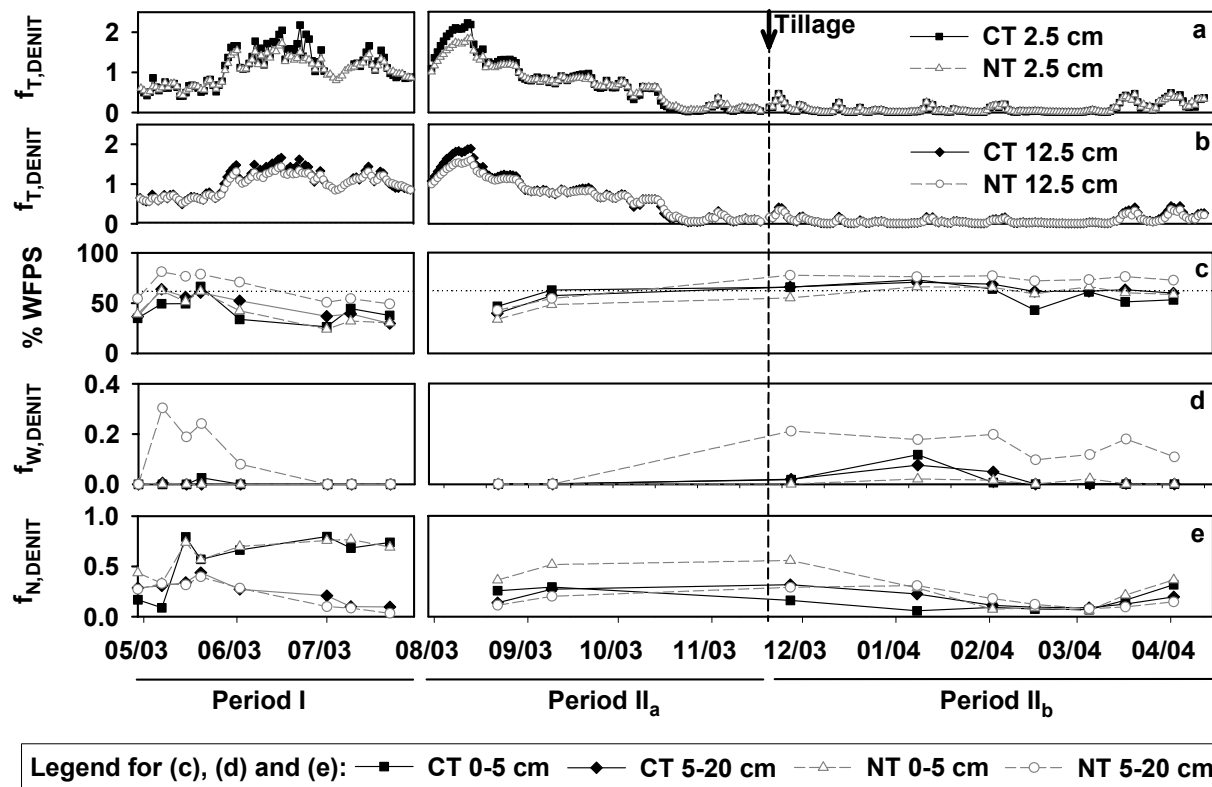


Figure 5.7: Determinants of denitrification in the conventionally tilled (CT) and non-tilled (NT) plots during periods I and II. Temperature factor for denitrification ($f_{T,DENIT}$) at (a) 2.5 and (b) 12.5 cm soil depth, (c) % water-filled pore space (WFPS), (d) moisture factor for denitrification ($f_{W,DENIT}$) and (e) nitrate concentration factor for denitrification ($f_{N,DENIT}$) in the 0-5 and 5-25 cm soil layers. Values of $f_{N,DENIT}$ for periods II_a and II_b are calculated for the 5-25 cm instead of the 5-20 cm layer.

5.4 Discussion

5.4.1 Influence of the different factors on CO₂ emissions

5.4.1.1 Soil CO₂ emission potential and SOM stocks

The 41% larger soil CO₂ emission potential measured for the NT plot compared to the CT plot in period I (Table 5.2) is consistent with the significantly larger C stocks (15%) measured in the NT plot (Table 5.3). However, the larger difference for the soil CO₂ potential than for the C stocks also means that even the specific CO₂ emission potential (expressed in %C) was 23% greater in NT than CT in the plots for period I. The absence of difference in soil CO₂ emission potentials between the CT and NT plots for period II, was consistent with the absence of differences between the SOM stocks in those plots. In accordance with Alvarez *et al.* (1995), the 0-5 cm soil layer of NT presented a larger specific soil CO₂ emission potential for both periods, whereas the contrary was observed for the deeper soil layers. This lower soil emission potential in the deeper soil layers of the NT period II plot, compensated the larger soil emission potential in the 0-5 cm soil layer. Mary and Guérif (1994) found, in accordance with the plots of period II, comparable C mineralisation coefficients between CT and NT for the SOM of the same maize/wheat rotation as in our study but during the first 12 years of tillage differentiation. Their results were obtained by fitting the total C stocks with the Hénin Dupuis model instead of by incubating SOM under controlled conditions. However, for the plots with maize monoculture at Boigneville, they found a 2 times larger mineralisation coefficient for SOM in CT compared to NT. The effect of organic matter protection by both soil structure and aggregates on SOM stocks and soil CO₂ emission potentials has already been discussed in Chapter 2 and 3 and will therefore not be repeated here.

Under field conditions, NT generally emitted larger amounts of CO₂ than tilled soil. Considering that both tillage systems had similar C inputs and that no SOM was lost by erosion due to the absence of slopes, the larger CO₂ emissions in NT seem to be in contradiction with the presence of slightly larger (period I) or similar (period II) SOM stocks in NT. Several hypotheses and observations may be quoted to explain these discrepancies. (i) First of all, the differences in SOM stocks are a result of 32 years of tillage differentiation. The differences in soil organic C accumulation between different tillage systems may change from year to year depending on the ratio of C input to C losses for each specific year that is largely controlled by climatic conditions. This was confirmed by Kessavalou *et al.* (1998b) who observed that the differences in linearly extrapolated annual C stocks for CT and NT, calculated from measured long-term changes in C stocks, were not in accordance with the

measured annual differences in C input and C loss. The temperatures and the PET from mid-May to mid-August 2003 were abnormally high during our study period, which affected both C input (rather low wheat crop yields) and C losses. (ii) Secondly, the CO₂ emissions were only measured on bare soil. NT may emit less CO₂ than CT during the maize and wheat growing season. However, the crop yields and crop C contents were similar in both CT and NT suggesting that the CO₂ emissions by the plants were also similar. (iii) Thirdly, as noted in Kessavalou *et al.* (1998b) and Ball *et al.* (1999), CH₄ uptake by the soil tends to be lower in CT than in NT but the magnitude of the CH₄ uptake differences mentioned in these papers was much smaller compared to the CO₂ emission differences in our study. (iv) Finally, after 32 years of differentiation, both tillage systems may have reached an equilibrium between input and output.

5.4.1.2 Decomposition of crop residues

Much of the 918 kg C ha⁻¹ larger CO₂ emission in NT, compared to CT, during period II can be explained by a 500 kg C ha⁻¹ larger decomposition of crop residues (Table 5.4). It was the decay of old weathered crop residues that explained the difference in residue decomposition between NT and CT. Curtin *et al.* (1998) observed only a small difference between the potential decomposition of recently harvested and weathered wheat straw. This means that weathered field residues can contribute significantly to CO₂ emissions. However, despite the larger absolute decay of residues in NT, a smaller proportion of wheat or of all residues was decomposed in NT compared to CT. This lower specific residue decomposition in NT, which was also observed by Dao (1998), explains the accumulation of residues at the soil surface in NT. This lower specific residue decomposition in NT was again in accordance with the modelling results of Mary and Guérif (1994) with the Hénin Dupuis model for the same maize/wheat rotation during the first 12 years after tillage differentiation. They namely found that, cumulated over one year, a larger fraction of the C situated in crop residues was mineralised to CO₂ in CT compared to NT. However, to further explore the contribution of crop residue decomposition to the difference in CO₂ emissions in our study, the proportions of CO₂ derived from the decomposition of residues or from SOM need to be quantified. This cannot be done without the help of modelling. Not all C decomposed from residues is emitted as CO₂. Part of the organic C is recycled in the soil microbial biomass or stabilised in SOM. In addition, earthworms may transport residue particles from the surface mulch to the soil underneath (Balabane *et al.*, 2005).

5.4.1.3 Soil temperature

The soil temperatures during summer were larger in CT than NT, which is consistent with several studies (Fortin *et al.*, 1996; Dao, 1998; Kessavalou *et al.*, 1998b; Alvarez *et al.*, 2001). During colder periods, CT and NT displayed similar temperatures consistent with Al-Kaisi and Yin (2005). In a study published by Buyanovsky *et al.* (1986), temperature was the most important abiotic factor explaining the annual CO₂ emissions in winter wheat ecosystems, but the variation in CO₂ emissions was still better explained by the combination of temperature and soil moisture. In periods without severe limitation by water content, Fortin *et al.* (1996) and Dao (1998) measured higher CO₂ emissions in CT only on those dates when the soil temperature was higher in CT. In our study, the periods with higher $f_{T,SOM}$ and $f_{T,RES}$ values in CT coincided with periods of water limitation. This suggests that part of the effect of soil temperature on CO₂ emissions could have been masked by the effect of soil moisture.

The distribution of the soil CO₂ emission potentials in the 0-25 cm soil profile showed that the soil CO₂ emission potential was highest in the 0-5 cm soil layer in NT and in the 5-20 cm soil layer in CT. When these differences in distribution of the soil emission CO₂ potential are taken into account, the differences in temperature factors between CT and NT are less pronounced for the 0-25 soil profile as a whole than as described above for the 2.5 and 12.5 cm soil depths.

5.4.1.4 Rainfall

For those periods when the crop residues were incorporated in CT (periods I and II_b), NT presented larger CO₂ emissions than CT during the occurrence of rainfall, whereas the opposite was true between the rainfall events. These field observations are in accordance with results obtained in the laboratory under controlled conditions by Coppens *et al.* (submitted). They found that rainfall suddenly increased the water content of surface residues inducing large CO₂ peaks, but that after the occurrence of rainfall, the water content of the surface residues fell very rapidly, and seriously limited their decomposition. In their incorporated residues treatment CO₂ emissions were largest between two rainfall events. Smaller CO₂ emissions under rainfall and wet conditions in CT may be partly explained by lower gas diffusivity of the CO₂ produced by the decomposition of incorporated residues in wet soils (Currie, 1984). When the profile drains, this captured CO₂ can escape from the soil resulting in large CO₂ emissions. Regarding SOM decomposition in NT, 59% of the C mineralisation potential was situated in the surface 0-5 cm layer, whereas the mineralisation of SOM in CT was relatively homogeneously distributed in the 0-25 cm layer. In other words, NT also enabled the CO₂ produced by SOM decomposition to diffuse more easily to the atmosphere due to the shallower depth of CO₂ production. Greater surface crusting due to rainfall impact

on bare soil in CT (Tebrügge and Düring, 1999) may limit CO₂ diffusion to the atmosphere even further.

Emission peaks during rainfall events in NT were smallest during periods I and II_b. This may be explained by (i) lower residue amounts during these periods compared to period II_a, (ii) low temperatures during period II_b and (iii) a slight variation in the water content of the surface residues during period II_b due to low PET and high soil water content between November and February. Vanlauwe *et al.* (1995) observed in the tropics that the number of rainfall events was better correlated with the percentage of dry matter loss from surface residues than with the total amount of precipitation. This means that in periods with high PET and normal but irregularly distributed rainfall, NT will emit less CO₂ than during periods with small but frequent rainfall events. However, rainfall distribution will have less effect on the water content of the incorporated residues in CT. So it seems likely that, depending on the pattern of rainfall, higher emissions may be measured in CT (with incorporated residues) than in NT (with surface residues), during certain periods while the opposite may be so during other periods.

CO₂ emissions peaked simultaneously in CT and NT during period II_a, due to the presence of surface residues in both tillage systems. The high peaks during rainfall events were related to the large amount of fresh residues and the high potential evaporation. The mulch dried up very rapidly due to the extensive evaporation resulting in a burst of CO₂ emission when rewetted by rain. The emission peaks were higher in NT than in CT throughout period II_a. This clearly suggests that NT emitted a larger amount of CO₂ produced by the decomposition of surface residues than CT.

5.4.1.5 Soil water content

The short-term effect of surface residues on the soil moisture content was twofold: (i) when the soil was wet and PET high, the surface residues reduced soil evaporation resulting in a wetter topsoil under NT than under CT (second half of period I); (ii) after a long dry period, it was more difficult for the soil to rewet under NT than under CT due to water retention by the surface residues (period II_a). The gravimetric water content during period I was often measured after rain or after an irrigation event, meaning that the $f_{w,SOM}$ values were probably slightly overestimated.

Several authors (Fortin *et al.*, 1996; Dao, 1998; Alvarez *et al.*, 2001; Al-Kaisi and Yin, 2005) measured larger volumetric water contents for NT than CT during certain periods, especially in the 0-5 cm layer. Fortin *et al.* (1996) attributed the higher CO₂ emissions in NT during a dry summer to the significantly lower soil water contents in the 0-15 cm soil profile in CT, the temperature in both CT and NT being similar. However, the decomposition of

SOM is determined by a $f_{W,SOM}$ factor (i.e. relative soil water content) and not directly by the gravimetric or volumetric water content (Rodrigo *et al.*, 1997). This factor takes the actual soil water content and also the water content at field capacity and wilting point into account. Consistent with Dao (1993), the water content at field capacity (or water-holding capacity) was larger in the 0-5 cm layer of long-term NT compared to CT due to a larger amount of organic matter and greater porosity. This resulted in smaller differences for the $f_{W,SOM}$ factor between CT and NT than for the gravimetric water content of the 0-5 cm soil layer, whereas the reverse (although less pronounced) was observed for the lower soil layers. In addition, for period I, when the larger proportion of soil CO₂ emission potential in the surface layer of NT compared to CT and the lower $f_{W,SOM}$ values for the 0-5 cm soil layer (compared to the 5-20 cm layer) were taken into account, the differences between the two tillage systems for the $f_{W,SOM}$ factor over the whole 0-25 cm soil profile were minimal.

5.4.1.6 Short-term effect of soil tillage

The cumulated CO₂ emission during period II_b after tillage was probably slightly underestimated since the CO₂ fluxes were not measured during the first 6 hours. Al-Kaisi and Yin (2005) measured 3 times more CO₂ emission from CT than from NT during the 6 hour period following tillage, while the CO₂ emissions measured by Reicosky and Lindstrom (1993) were 20 times more. When the emissions during this 6 hour period in our study are taken into account, the cumulated CO₂ emissions for period II_b are increased by 9-69 kg C ha⁻¹ which means that the cumulated emissions still remain significantly smaller for CT than for NT. The direct effect of tillage is probably due to a physical effect on the release of CO₂ previously trapped in small pores.

5.4.1.7 Time of soil tillage

The CT field was ploughed on November 19th 2003, nearly four months after wheat harvest, and this probably reduced the effect of residue incorporation. Firstly, between wheat harvest in July 2003 and tillage in November 2003, 53% of the residue-C had already been decomposed in CT (Table 5.4). Secondly, at the moment of tillage not much nitrate-N was present in the soil profile for the decomposition of residues. Thirdly, the temperature factor $f_{T,RES}$ for residue decomposition was extremely low and not very different between the two tillage systems during this period. Finally, the surface residues did not dry very rapidly during the period after tillage, due to a low PET and the contact with wet soil. A slower drying of surface residues reduces the differences between incorporated and surface residue decomposition. All these factors probably reduced the effect of CT on CO₂ emissions in comparison with tillage carried out directly after wheat harvest and/or under warmer

temperatures. In the maize year of the maize-wheat rotation at Boigneville, tillage is carried out at the end of October (i.e. in a comparable period).

5.4.1.8 Frequency of measurements

Most authors (Reicosky and Lindstrom, 1993; Fortin *et al.*, 1996; Kessavalou *et al.*, 1998b; Al-Kaisi and Yin, 2005) measured CO₂ emissions exclusively during the day and not during the night. This overestimates the temperature differences and thus the difference in CO₂ emissions between CT and NT. In addition, no authors measured daily CO₂ fluxes over a long period. In our study, the differences in CO₂ emissions between CT and NT were highly dependent on rainfall pattern and temperature, so weekly measurements may give erroneous estimates of the yearly CO₂ emissions. This is especially the case during warm periods when a lot of crop residues are present at the surface of the soil in NT and incorporated in CT.

5.4.2 N₂O emissions

As expected (Hénault *et al.*, 2005), the largest N₂O fluxes were measured after N fertilisation in spring. As reported by MacKenzie *et al.* (1998), Ball *et al.* (1999), Vinten *et al.* (2002) and Baggs *et al.* (2003), NT emitted a larger amount of N₂O than CT. Choudhary *et al.* (2002) measured a similar yearly N₂O flux for CT and NT but CT was rolled with a heavy roller after tillage thus reducing the porosity of the CT soil. Despite higher WFPS and nitrate contents in NT, Kessavalou *et al.* (1998b) measured similar N₂O emissions in CT and NT. However they also measured higher CO₂ emissions in CT, which suggests that the higher denitrification in NT could have been compensated by higher N₂O emissions from nitrification in CT or that the N₂:N₂O ratio produced by denitrification was higher in NT than CT. Groffman (1984) measured a higher potential denitrification activity for the 0-5 cm layer of NT compared to CT but a lower potential for the deeper soil layer resulting in a similar potential denitrification activity for the whole 0-21 cm soil layer.

In wet soils with high WFPS and low gas diffusivity in Scotland, N₂O was mainly produced by denitrification (Ball *et al.*, 1999; Vinten *et al.*, 2002). However, as in Kessavalou *et al.* (1998b) and Choudhary *et al.* (2002), the N₂O emitted in our study during spring, summer and autumn would be a product of nitrification rather than denitrification. Nitrification was namely enhanced by high soil temperatures (Figure 5.3) and the addition of nitrogen fertiliser (ammonium nitrate). However, regarding denitrification, the $f_{N,DENIT}$ factor was high for the 0-5 cm soil layers after fertilisation, but the water filled pore space (WFPS) of these soil layers was not sufficient (< 62%) for denitrification to occur (Figure 5.7). Only the WFPS of the 5-20 cm soil layer of NT was noticeably higher than 62% up to mid-June. Note that the WFPS was calculated with the average water content and bulk density of the 0-5

and 5-20 cm soil layers. However, the soil is full of micro-sites with higher than average WFPS. This means that denitrification may occur when the average WFPS is below 62%. In addition, large aggregates form hot spots of denitrification due to anaerobic conditions inside the aggregates, especially when particulate organic matter is occluded (Six *et al.*, 2002). The latter suggests that the larger stable macroaggregates in the period I field plots in NT (Chapter 2) may explain part of the larger observed N_2O fluxes in NT.

From mid-October onwards, the N_2O emissions were probably products of both nitrification and denitrification. The WFPS was above 62% (except for the 0-5 cm soil layers at certain dates) indicating that moisture conditions were favourable for denitrification (Figure 5.7). However, the WFPS did not exceed 80% which is the upper limit for nitrification (Hénault *et al.*, 2005). Low N_2O emissions were mainly the result of temperature limitation of both denitrification ($f_{T,DENIT}$ in Figure 5.7) and nitrification ($f_{T,SOM}$ in Figure 5.3). In addition, the $f_{N,DENIT}$ factor was low during period II. This limitation by nitrate concentration was smallest for the 0-5 cm soil layer of NT.

The larger N_2O emissions in NT may counterbalance the surplus C storage in NT since N_2O has a 296 times greater warming potential than CO_2 (IPPC, 2001). In order to test this, the N_2O emissions were also expressed in equivalent kg CO_2 -C by multiplying the value expressed in kg N_2O -N by 296. In addition, the measured N_2O fluxes were extrapolated to annual values of N_2O emission, namely 0.80 ± 0.15 kg N_2O -N $ha^{-1} yr^{-1}$ or 237 ± 44 kg CO_2 -C $ha^{-1} yr^{-1}$ for CT and 1.32 ± 0.52 kg N_2O -N $ha^{-1} yr^{-1}$ equivalent to 391 ± 154 kg CO_2 -C $ha^{-1} yr^{-1}$ for NT. This means that the surplus in yearly N_2O emissions in NT compared to CT amounts 0.52 kg N_2O -N $ha^{-1} yr^{-1}$ equivalent to 153 kg CO_2 -C $ha^{-1} yr^{-1}$ which is in the same order of magnitude as the earlier mentioned mean annual gain in C stocks in NT (76 to 175 kg C $ha^{-1} yr^{-1}$). Thus, this result clearly suggests that the increase in N_2O emissions may counterbalance the surplus in C storage in NT compared to CT. However, the extrapolated N_2O emissions are only rough estimates of the yearly N_2O emissions since N_2O emissions are very variable in time and space due to soil heterogeneity and their episodic nature (Ball *et al.*, 1999; Choudhary *et al.*, 2002). Reliable data for yearly N_2O emissions can only be obtained by measuring continuously and having a high number of replicates. In addition, Kessavalou *et al.* (1998b) measured larger N_2O emissions from the row than from inter-row locations of a winter wheat - summer fallow system and N_2O in this study was only measured at inter-row locations. Finally, weather conditions (rainfall and temperature) differ from year to year resulting each year in different flux magnitudes and timing of N_2O bursts in both CT and NT (MacKenzie *et al.*, 1998).

5.5 Conclusions

The greenhouse gases CO₂ and N₂O emitted from agricultural soils contribute to the global warming of our planet. Although no-tillage is often proposed as a means of decreasing CO₂ emissions, these have rarely been continuously measured in long-term differentiated tillage systems. Two tillage systems, i.e. no-tillage (NT) and conventional tillage with mouldboard ploughing (CT) were imposed in 1970 on an experimental field at Boigneville (Northern France) under a maize-wheat rotation. CO₂ emissions were continuously measured in both treatments from the end of April 2003 to mid-April 2004. N₂O emissions were periodically measured during the same period. The soil organic C and mineral N, the soil potential for CO₂ emission, the decomposition of crop residues and the soil climatic and weather data were also measured. CO₂ emissions did not differ significantly between CT and NT on 41% of the days. Emissions were significantly larger in NT on 53% and in CT on 6% of the days. The cumulative CO₂ emission over the entire 331 days measurement period was significantly larger in NT (5131 ± 169 kg CO₂-C ha⁻¹) compared to CT (4279 ± 333 kg CO₂-C ha⁻¹). This was not in agreement with our initial hypothesis that NT emits less CO₂ than CT which was based on the fact that NT tended to have 2.4-5.6 t C ha⁻¹ larger C stocks after 32 years of tillage differentiation. Differences in CO₂ emissions in the different tillage systems were largely a result of the soil climatic conditions and the amounts and location of crop residues and SOM in CT and NT. The location of residues and SOM induced differences in time and magnitude in the CO₂ emission responses to rainfall events in the two tillage systems. A large proportion of the greater cumulated CO₂ emissions in NT over the whole measurement period was due to the decomposition of old weathered residues. The effect of temperature and rainfall events on the differences in CO₂ emissions clearly demonstrated the importance of measuring CO₂ several times a day over a long period. In order to better analyse the long-term effect of no-tillage, CO₂ fluxes have to be modelled. Modelling permits isolation of the individual effect of a factor from the complex interactions between the different factors determining CO₂ emissions (temperature, rainfall, water content, amount of SOM and crop residues, etc.). A model can also calculate the proportion of CO₂ produced from the decomposition of SOM and residues.

N₂O emissions always tended to be larger in NT than in CT. However, due to considerable spatial variability (especially when emissions were high), only half of these N₂O emissions were significantly larger. Rough extrapolations of the yearly N₂O emissions indicated that the larger N₂O emissions in NT can counterbalance the mean yearly surplus in C stock in NT due to the 296 higher warming potential of N₂O than CO₂. However, in order to compute reliable annual differences in N₂O loss in both tillage systems, it is extremely

important to continuously measure N₂O fluxes over a long period with many replicates since the emission of N₂O is highly episodic and exhibits enormous spatial variability.

NT emitted more CO₂ and N₂O than CT cumulated over the entire measurement period. Thus as regards the emission of these greenhouse gases in the Boigneville soil, NT was certainly not preferable to CT under the specific weather conditions of the measurement year.

CHAPTER 6: MODELLING CARBON AND NITROGEN DYNAMICS IN NO-TILLAGE AND CONVENTIONAL TILLAGE SYSTEMS

6.1 Introduction

Compared to conventional mouldboard ploughing tillage (CT) systems, the introduction of no-tillage (NT) systems modifies the localization of crop residues and the distribution of soil organic matter. In NT systems, the crop residues are situated in a mulch layer on top of the soil surface, whereas in CT systems these residues are incorporated into the soil. In addition, the no-tillage system induces changes in physical soil properties such as bulk density, aeration and hydraulic properties. All these modifications, in turn, interact together on the water, temperature and C and N dynamics in the soil. Understanding and quantifying these complex interactions remains a challenging issue. Modelling allows extracting the effect of each individual change. Many models exist where the decomposition of soil organic matter and incorporated residues are coupled with water and heat fluxes, but only few models simulate the impact of a mulch layer of crop residues. A mulch module exists in the Expert-N model (Berkenkamp *et al.*, 2002), the APSIM model (Thorburn *et al.*, 2001) and in the PASTIS model (Findeling *et al.*, submitted). The decomposition of surface and incorporated residues in the Expert-N model (Berkenkamp *et al.*, 2002) is mainly differentiated by N-limitation and the degree of contact with the soil. However, Coppens *et al.* (submitted) found that the difference in decomposition between surface and incorporated residues is largely determined by the water content of these residues. The mulch modules in APSIM (Probert *et al.*, 1998; Thorburn *et al.*, 2001) and PASTIS (Findeling *et al.*, submitted) take the effect of a surface mulch on the water dynamics of the whole mulch-soil system into account. Furthermore, the water and temperature dynamics and C and N transformations interact with each other in those models. Probert *et al.* (1998) simulated water and N dynamics in conventional and no-tillage systems with the APSIM-model, but they evaluated the model only with long-term differences in water and nitrate content between the two systems. Coppens (2005) used the PASTIS model to study the short-term effects of crop residue location on the water and C and N dynamics in soil under controlled laboratory conditions. Garnier *et al.* (2003) evaluated the PASTIS model for field conditions with incorporated straw in a conventional tillage system. However, the PASTIS model was not yet evaluated for field conditions in no-tillage systems.

The aim of our study is twofold: (i) to model the C and N fluxes in real farming conditions with CT and NT differentiated for 32 years and (ii) to determine, by means of simulations, the influence of the main factors explaining the differences in CO₂ emissions between the two tillage systems. These factors include soil temperature, soil water content and the amount and localization of humified and fresh organic matter (crop residues). Both the influence of the location of crop residues as the effect of long-term changes in biological and physical properties are taken into account.

6.2 Materials and methods

6.2.1 Experimental set-up

The *study site* is located at Boigneville in Northern France (48°33'N, 2°33'E). Two tillage systems, i.e. no-tillage (NT) and conventional tillage (CT), have been differentiated since 1970. Before differentiation, the two systems were yearly mouldboard ploughed to 30 cm depth. In this work we studied two plots: the CT plot (plot 224) which was yearly mouldboard ploughed to 20 cm depth and the NT plot (plot 223) which was only minimally disturbed along the sowing line by the sowing machine (< 5cm). Both have been cropped with a maize-wheat rotation and crop residues have been returned to the soil every year. In the CT plot, crop residues remained on the soil surface until incorporation by tillage whereas, in the NT plot, crop residues always remained on the soil surface. The modelling period started at the tillage event on the 19th of November 2003 and ended at maize sowing on the 13th of April 2004. During the whole period the soil remained uncropped. The wheat crop had been harvested at the end of July 2003.

The soil is a Haplic Luvisol. The characteristics of the upper 0-25 cm soil layer of the two plots are presented in Table 6.1. Chapter 5 described the experimental set-up in detail. *Volumetric soil water content* was measured using TDR probes (Time Domain Reflectory, Campbell CS616) installed at 2.5, 12.5 and 25 cm soil depths. For each soil layer, these probes were calibrated with manual measurements of gravimetric water content and bulk density in the 0-5, 5-20 and 20-25 cm soil layers. *Rainfall, potential evaporation and air temperature* were recorded by an automatic weather station at the experimental site. Soil temperature was recorded hourly with thermocouples at 0, 2.5 and 12.5 cm and 25 cm soil depths. The amounts of *surface and incorporated crop residues* were quantified on the 18th of November 2003 (day before the tillage event) and the 13th of April 2004 (end of the simulation period). The C and N concentrations of the residues were determined using an NA

1500 elemental analyser (Fisons Isochrom). Surface residues in NT were a combination of recently harvested wheat residues and older weathered maize and wheat residues. Maize and wheat plants have different natural ^{13}C abundance. Thus, ^{13}C measurements were performed using an NA 1500 elemental analyser coupled to a mass spectrometer (Fisons Isochrom) to estimate the part of the residue-C derived from maize residues. Fluxes of CO_2 were measured four times a day using four automatic chambers connected to an infra-red analyser for each tillage treatment. At the end of the experiment, soil samples were taken from the 0-5 cm, 5-20 cm and 20-25 cm layers inside the gas chambers. **Total C and N of these samples** were measured with the elemental analyser and **total microbial biomass C** was determined by the fumigation extraction technique as described by Trinsoutrot *et al.* (2000a). **Mineral N** of the 0-5 and 0-25 cm soil layers was measured monthly as explained in Chapter 4.

Table 6.1 : Soil properties of the conventional tillage (CT) and no-tillage (NT) field plots studied at Boigneville.

Soil layer (cm)	CT			NT		
	0-5	5-20	20-25	0-5	5-20	20-25
Clay ($< 2 \mu\text{m}$) ($\text{g } 100 \text{ g}^{-1}$ soil)	26.5	26.6	26.7	22.4	24.2	25.2
Silt ($2\text{-}50 \mu\text{m}$) ($\text{g } 100 \text{ g}^{-1}$ soil)	65.3	65.1	64.7	67.2	66.3	65.5
Sand ($> 50 \mu\text{m}$) ($\text{g } 100 \text{ g}^{-1}$ soil)	8.2	8.3	8.6	10.4	9.5	9.3
pH water (-)	6.6	6.6	6.7	5.1	6.3	7.0
CaCO_3 ($\text{g } 100 \text{ g}^{-1}$ soil)	0.1	< 0.1	< 0.1	0.1	< 0.1	< 0.1
CEC ($\text{cmol}_\text{c} \text{ kg}^{-1}$ soil)	13.5	13.8	13.9	13.3	12.8	13.0
Organic C ($\text{g } 100 \text{ g}^{-1}$ soil)	1.10	1.11	1.07	2.26	0.95	0.77
Organic N ($\text{g } 100 \text{ g}^{-1}$ soil)	0.112	0.113	0.109	0.193	0.101	0.085
C : N ratio (-)	9.9	9.8	9.8	11.7	9.4	9.1
Bulk density (g cm^{-3})	1.320	1.320	1.517	1.132	1.517	1.517

6.2.2 Model description

The transfer of water, heat and solute and the transformations of C and N in the soil of the CT and NT plots were modelled with the one-dimensional mechanistic model PASTIS (Prediction of Agricultural Solute Transformations In Soils) (Lafolie, 1991; Garnier *et al.*, 2001). The model includes two submodels: a transport submodel and a transformation

submodel called CANTIS (Carbon and Nitrogen Transformations In Soil). A full description of the PASTIS model is given in Appendix C. Garnier *et al.* (2003) evaluated the PASTIS model on field data collected during one year from a conventional tillage system with incorporated wheat straw. Findeling *et al.* (submitted) added a mulch module to the PASTIS model to simulate the water and C and N dynamics in the presence of a mulch of surface residues. This module takes into account the rain interception by the surface mulch and simulates changes in evaporation, thermal exchanges and C and N transformations. The extended PASTIS model was tested with data obtained under controlled laboratory conditions.

6.2.3 Description of the soil profiles

The zero depth of the soil profiles was set at the soil-mulch interface. The NT plot was modelled down to 25 cm which was below the ploughing depth in CT. In order to take into account soil profiles with similar masses of dry soil in the two tillage systems, the CT soil profile was extended with 1 cm (0-26 cm). In the text below, the soil profiles in CT and NT are considered to be 0-25 cm soil profiles to simplify the text. The modelling was limited to a 0-25 cm soil profile for two reasons: (i) no crop residues were found below that depth and (ii) total organic C and N concentrations were much lower below than above 25 cm.

The simulated NT soil profile consisted of two soil layers. The 0-5 cm soil layer had a loose structure, low bulk density and high C and N concentrations. The 5-25 cm soil layer had a high bulk density, massive structure and low C and N concentrations (Table 6.1). Crop residues in the NT plot were situated as a mulch (approximately 2 cm thick) on the soil surface.

The simulated CT profile was divided into a 0-20 cm recent plough layer and a denser 20-25 cm soil layer. The actual 0-20 cm plough layer had a very heterogeneous structure with dense and compacted clods, loose soil, voids and lumps of incorporated straw residues. Since PASTIS is a one-dimensional model, the heterogeneity in the plough layer had to be simplified. Firstly, an average bulk density was taken for the whole recent plough layer (0-20 cm). Secondly, two vertical soil strips were defined based on field estimations of the proportion of the volume occupied by groups of crop residues over the whole soil area (see soil profiles in appendix A): a soil strip containing all the incorporated crop residues (10% of the soil volume) and a soil strip without residues (90% of the soil surface area). In the strip with crop residues, residues were homogeneously spread over the 5-20 cm layer because only a negligible amount of crop residues was observed in the 0-5 cm soil layer (as also observed in a study by Staricka *et al.* (1991)). The two CT soil strips were simulated separately and then summed up taking their relative proportion of the soil volume into account.

Rainfall intensity and potential evapotranspiration were the surface boundary conditions for simulating water transport. At the bottom of the soil profile, a water pressure head was imposed. It was calculated using the measured volumetric water content and the water retention curve. The temperatures measured at depths of 0 and 25 cm were set as boundary conditions at the top and bottom, respectively. The equations of the PASTIS model were solved using a finite difference technique.

6.2.4 Transport parameters

The water flow parameters used in PASTIS are shown in Table 6.2. The water retention curve of the different soil layers was determined in each tillage plot by measuring the soil water content of undisturbed soil samples under a succession of imposed water pressure heads: 0.1, 1, 10, 50, 100 and 1585 kPa. The suction table and low and high pressure chamber methods were used for the pressure ranges 0-10, 10-300 and 300-1600 kPa respectively (Kabat and Beekma, 1994). The equation of Van Genuchten (1980) was fitted to the experimental data:

$$S_e(h) = \frac{\theta - \theta_r}{\theta_s - \theta_r} = \left[1 + (\alpha h)^n\right]^{\left(1 - \frac{1}{n}\right)}$$

where $S_e(h)$ is the effective saturation, θ_r and θ_s are respectively the residual and saturated volumetric water contents ($\text{m}^3 \text{m}^{-3}$), α is the scaling factor of water pressure head (kPa^{-1}), h is the water pressure head (kPa) and n is an empirical shape parameter (-).

Hydraulic conductivity data were obtained using the WIND method (Tamari *et al.*, 1993). The equation of Van Genuchten (1980) was selected for the hydraulic conductivity curve:

$$K(\theta) = K_s S_e^l \left[1 - (1 - S_e^{1/m})^m\right]^2$$

where K_s is the saturated soil hydraulic conductivity (m s^{-1}), m is an empirical shape parameter (-), and l is the tortuosity parameter set to 0.5. K_s and m were optimised with the measured soil water content and the hydraulic conductivity data obtained by the WIND method.

Heat flow and solute parameters were taken from Garnier *et al.* (2001) who worked on a very similar loamy soil in Mons-en-Chaussée in Northern France. They estimated the soil thermal conductivity coefficients by fitting the model to field data of temperatures at different depths. A soil dispersivity of 1 cm obtained by these authors by fitting the model to data from field leaching experiments with chloride, was used in our work.

Table 6.2 : Water flow parameters used in the PASTIS model for the water retention and hydraulic conductivity curve of the different soil layers in the conventional tillage (CT) and no-tillage (NT) plots.

Depth (cm)		CT		NT	
		0-20	20-25	0-5	5-25
Retention curve $h(\theta)$					
θ_s	Saturated volumetric water content ($\text{m}^3 \text{m}^{-3}$)	0.4200	0.4200	0.4638	0.3859
θ_r	Residual volumetric water content ($\text{m}^3 \text{m}^{-3}$)	0.000	0.000	0.000	0.000
α	Scaling factor of water pressure head (kPa^{-1})	0.634	0.634	0.236	0.123
n	Empirical shape parameter (-)	1.1441	1.1441	1.1948	1.1518
Conductivity curve $K(\theta)$					
K_s	Saturated hydraulic conductivity (m s^{-1})	1.94E-05	1.94E-05	5.41E-06	4.93E-06
m	Empirical shape parameter (-)	1.1500	1.1500	1.209	1.089

6.2.5 Biological parameters

The biological parameters of CANTIS were obtained from incubation experiments or taken from Garnier *et al.* (2003). Incubations of soil without added fresh organic matter were carried out on 12.5 mm sieved fresh soil samples taken in the field gas chambers in the 0-5, 5-20 and 20-25 cm soil layers at the end of the measurement period. During these incubations, potential C and N mineralisation of the soil samples was measured under controlled conditions (15°C and water pressure head of 63 kPa) for 168 days and using the method described in Chapter 3. The parameters used to describe the decomposition of humified organic matter, i.e. the decomposition rate of the autochthonous biomass k_A , the decomposition rate of the humified organic matter k_H and the humification coefficient h_A , were estimated by fitting the CANTIS submodel to the C and N mineralisation data. It is not possible to enter different biological parameters for different soil layers in the PASTIS model. However, for the NT treatment, the soil from the 0-5 cm layer had a higher mineralisation rate than the soil from the 5-20 and 20-25 cm soil layers. In addition, the 0-20 cm soil layer of the CT treatment also had a higher mineralisation rate than the 20-25 cm layer. To minimise this problem, the k_A , k_H and h_A parameters were optimised with the mineralisation data of the upper soil layer (Table 6.3). We then reduced the initial amounts of humified organic matter in the NT₅₋₂₀, NT₂₀₋₂₅ and CT₂₀₋₂₅ layers so that, the calculated mineralisation kinetics using the fixed biological parameters were similar to the measured mineralisation kinetics.

Parameters K_{MA} and K_{MZ} , i.e. the biomass dependent factors for the decomposition of the humified and soluble organic matter, were set to zero (i.e. no dependence of the decomposition rates on the size of the microbial biomass). The biological parameters used to describe the decomposition of fresh organic matter were taken from Garnier *et al.* (2003) because the same crop residue (wheat straw) was used.

Table 6.3 : Optimised biological parameters of the CANTIS submodel with the measured potential C and N mineralisation in the conventional tillage (CT) and no-tillage (NT) plots.

Symbol	Parameter	CT	NT
k_A	Decomposition rate of autochthonous biomass (d^{-1})	2.003E-02	5.021E-02
k_H	Decomposition rate of humified organic matter (d^{-1})	4.143E-04	4.818E-04
h_A	Humification rate of autochthonous biomass (-)	0.6361	0.3800

6.2.6 Parameters of the mulch module

The parameters specific for the mulch module were either measured, taken from literature or calibrated (Table 6.4). The covering proportion of the mulch was set to 1, based on field observations. The maximal and minimal volumetric water content of the mulch elements and the mulch density were taken from Garnier *et al.* (2004) who measured the hydraulic properties of wheat straw at Boigneville. The following parameters were calibrated: the mulch propensity to reduce soil evaporation (ξ), the mulch propensity to water recharge (α_m), the maximum depth for available N for mulch decomposition ($z_{m,zyb}$) and the proportion of the mulch dry mass in contact with the soil. The calibration of these parameters was necessary for three reasons: (i) no references were found in literature for them, (ii) they were not easy measurable and/or (iii) they change with the kind of mulch. The parameters ξ and α_m were calibrated with the soil water content of the 0-5 cm soil layer. The mulch propensity to water recharge (ξ) determines the amount of water that is intercepted by the mulch and consequently the amount of water that is directly transferred to the soil. The parameter $z_{m,zyb}$ was calibrated within a 0-5 cm range by fitting the simulated to the observed nitrate content of the 0-5 cm soil layer. A value of 1 cm gave the best results. The initial proportion of the mulch dry mass in soil contact was calibrated by fitting the simulated to the measured CO₂ emissions and resulted in a value of 100% contact of the mulch with the soil. This value is in accordance with the field observations of a thin mulch layer of surface residues with no standing stubble at the beginning of the simulation period. Those residues had remained at least 3.5 months on the soil surface which had enforced the contact with the soil.

Table 6.4 : Parameters for the mulch module for the no-tillage (NT) plot.

Symbol	Parameter	Value
$DM_{m,c}(0) / DM_m(0)$	Initial proportion of mulch dry mass in soil contact (%)	100*
τ_c	Covering proportion mulch (0-1)	1
z_m	Mulch thickness (cm)	2
ρ_{me}	Mulch element density (g cm^{-3})	0.116
$\theta_{me,max}$	Maximal water content of mulch elements ($\text{m}^3 \text{ m}^{-3}$)	0.65
$\theta_{me,res}$	Residual water content of mulch elements ($\text{m}^3 \text{ m}^{-3}$)	0.1
$R_{m,max}$	Maximal mulch water storage (cm)	0.44
$R_{m,min}$	Minimal mulch water storage (cm)	0.07
α_m	Mulch propensity to water recharge (-)	0.01*
ξ	Mulch propensity to reduce soil evaporation (-)	0.60*
$z_{m,zvb}$	Max. depth for available N for mulch decomposition (cm)	1*

* calibrated parameter

6.2.7 Initial conditions of organic matter pools

The biochemical composition of the crop residues was determined using a modification of the method proposed by Van Soest described in Trinsoutrot *et al.* (2000b). The neutral detergent fraction (NDF), hemicellulose (HEM), cellulose (CEL) and lignin (LIG) fractions were separated. The soluble fraction (SOL) of the residues was extracted by shaking in cold water at 20°C for 30 minutes. The C and N concentrations of the different fractions were determined by dry combustion using an NA 1500 elemental analyser (Fisons, Milan, Italy). The rapidly decomposable material (RDM) in PASTIS is defined as the difference between NDF and SOL. The amount of fresh organic matter (FOM) is equal to the sum of the RDM, HEM, CEL and LIG fractions. The initial amount of zymogenous biomass C was estimated to be 25% of the measured total microbial biomass C. The initial conditions for the entire soil profile are listed in Table 6.5 whereas initial conditions for the different soil layers are shown in Table 6.6.

Table 6.5 : Initial conditions of the CANTIS submodel for the conventional tillage (CT) and no-tillage (NT) plots.

Parameter	CT	NT
Amount of FOM in crop residues (kg C ha ⁻¹)	1464	3430
Proportion of rapidly decomposable material in FOM (% C)	5.2	10.2
Proportion of hemicellulose in FOM (% C)	34.0	31.6
Proportion of cellulose in FOM (% C)	51.7	45.4
Proportion of lignin in FOM (% C)	9.1	12.8
Amount of SOL in crop residues (kg C ha ⁻¹)	26	66
N:C ratio of rapidly decomposable material in FOM (g N g ⁻¹ C)	0.0511	0.0480
N:C ratio of hemicellulose in FOM (g N g ⁻¹ C)	0.0156	0.0136
N:C ratio of cellulose in FOM (g N g ⁻¹ C)	0.0000	0.0000
N:C ratio of lignin in FOM (g N g ⁻¹ C)	0.0242	0.0278
N:C ratio of the soluble organic pool (g N g ⁻¹ C)	0.0729	0.0957
N:C ratio of zymogenous microbial biomass (g N g ⁻¹ C) *	0.07	0.07
N:C ratio of autochthonous microbial biomass (g N g ⁻¹ C) *	0.125	0.125
N:C ratio of humified organic matter (g N g ⁻¹ C)	0.1027	0.0886

* These values are taken from Garnier *et al.* (2003), the other values were measured.

Table 6.6 : Initial sizes of the organic matter pools of the CANTIS submodel for the different soil layers in the conventional tillage (CT) and no-tillage (NT) plots.

Soil layer (cm)	CT			NT		
	0-5	5-20	20-25	0-5	5-20	20-25
Amount of FOM in soil (mg C kg ⁻¹ soil)	54	54	0	278	0	0
Amount of SOL in soil (mg C kg ⁻¹ soil)	7.2	7.2	9.4	27.7	6.1	1.7
Amount of ZYB in soil (mg C kg ⁻¹ soil)	61.4	61.4	54.6	75.5	46.5	42.9
Amount of AUB in soil (mg C kg ⁻¹ soil)	184	184	164	227	139	129
Amount of HUM in soil (mg C kg ⁻¹ soil)	11419	11419	8668 ¹	22580	3200 ¹	2000 ¹

¹ optimised value

6.2.8 Model evaluation

The model was evaluated statistically for soil volumetric water content and temperature dynamics, CO₂ emissions and soil nitrate content. The efficiency (*EF*) and the mean difference (*MD*) were used to assess the model performance (Smith *et al.*, 1996):

$$EF = \frac{\sum_{i=1}^n (m_i - \bar{m})^2 - \sum_{i=1}^n (s_i - m_i)^2}{\sum_{i=1}^n (m_i - \bar{m})^2}$$

$$MD = \frac{\sum_{i=1}^n (s_i - m_i)}{n}$$

where m_i and s_i are the measured and simulated results, and $\bar{m}$ is the average of the n measured results.

6.3 Results and discussion

6.3.1 Soil water content

Two periods can be distinguished regarding the soil water dynamics (Figure 6.1): an infiltration period from the 19th of November 2003 until the 28th of January 2004 and an evaporation period from the 29th of January 2004 until the 13th of April 2004.

The volumetric water content of the NT plot is well simulated by the model. However, the volumetric water content of the CT plot is badly simulated, especially for the 0-5 cm soil layer. This is probably due to the fact that the soil water content was measured during the period directly after tillage in the CT plot which resulted in an highly heterogeneous structure with a lot of voids between loose and dense soil clods. Furthermore, the bulk density of the CT plot changed considerably over the simulation period. Coutadeur-Desbourdes (2002) found that in a ploughed field the water flux progressed rapidly in zones with loose structure whereas the water flux avoided dense structures. This means that when a TDR probe is located in a void or a loose structure, the water content measured by the TDR probe can rise and decrease suddenly. On the other hand, a TDR probe located in the middle of a big clod, will not measure much variation in water content, because most of the infiltrating water does not reach the middle of the clod. This spatial and temporal variability in the CT plot resulted in a bad calibration of the TDR with the manually measured water content. This explains, in

combination with the absence of preferential flow in the PASTIS model, the bad agreement between simulations and measurements of the soil water content in CT.

The mean differences between measured and simulated values were acceptable for all soil layers except the 0-5 cm soil layer of CT (Table 6.7). Model efficiencies are rather low, but this is mainly due to the small variations in measured water contents. The large overestimation of the water content in the 0-5 cm soil layer in CT implies that the calculated values for the reduction function of moisture on SOM and crop residue decomposition were not reliable for this soil layer. However, only a small proportion of the total decomposition in CT occurred in the 0-5 cm layer since crop residues were mainly located in the 5-20 cm soil layer.

The differences in soil water content between CT and NT (Figure 6.1) are not only due to the presence of the mulch layer (e.g. effect on water evaporation as mentioned below), but also result from differences in the water retention curves after 32 years of tillage differentiation (Table 6.2). The 0-5 cm layer of NT presents both larger volumetric water contents at low water pressure heads as smaller volumetric water contents at high water pressure heads compared to the other soil layers of NT and CT. This difference in the water retention curve is both due to the larger organic matter content (Gupta *et al.*, 1977) and the lower bulk density (Reicosky *et al.*, 1981) in the 0-5 cm layer of NT. As a consequence, the 0-5 cm soil layer of NT had larger gravimetric and volumetric soil water contents than the other soil layers, although simulated water pressure heads were similar in all soil layers of the CT and NT profile. No large differences were found between the water retention curves of the CT treatment and the 5-25 cm soil layer of the NT treatment which is in accordance with the literature on this subject (Wu *et al.*, 1992; Benjamin, 1993; Hubbard *et al.*, 1994; Lal, 1999; Blanco-Canqui *et al.*, 2004; Fuentes *et al.*, 2004). The unsaturated hydraulic conductivity as a function of the water pressure head in the equivalent plough layer follows the order $NT_{0-5} > CT_{0-20} > NT_{5-20}$. There is no consensus in literature on the effect of tillage on the unsaturated hydraulic conductivity as a function of the water pressure head: some authors report larger values for NT (Dao, 1993; Fuentes *et al.*, 2004), while others report similar values for CT and NT (Datiri and Lowery, 1991; Wu *et al.*, 1992; Benjamin, 1993).

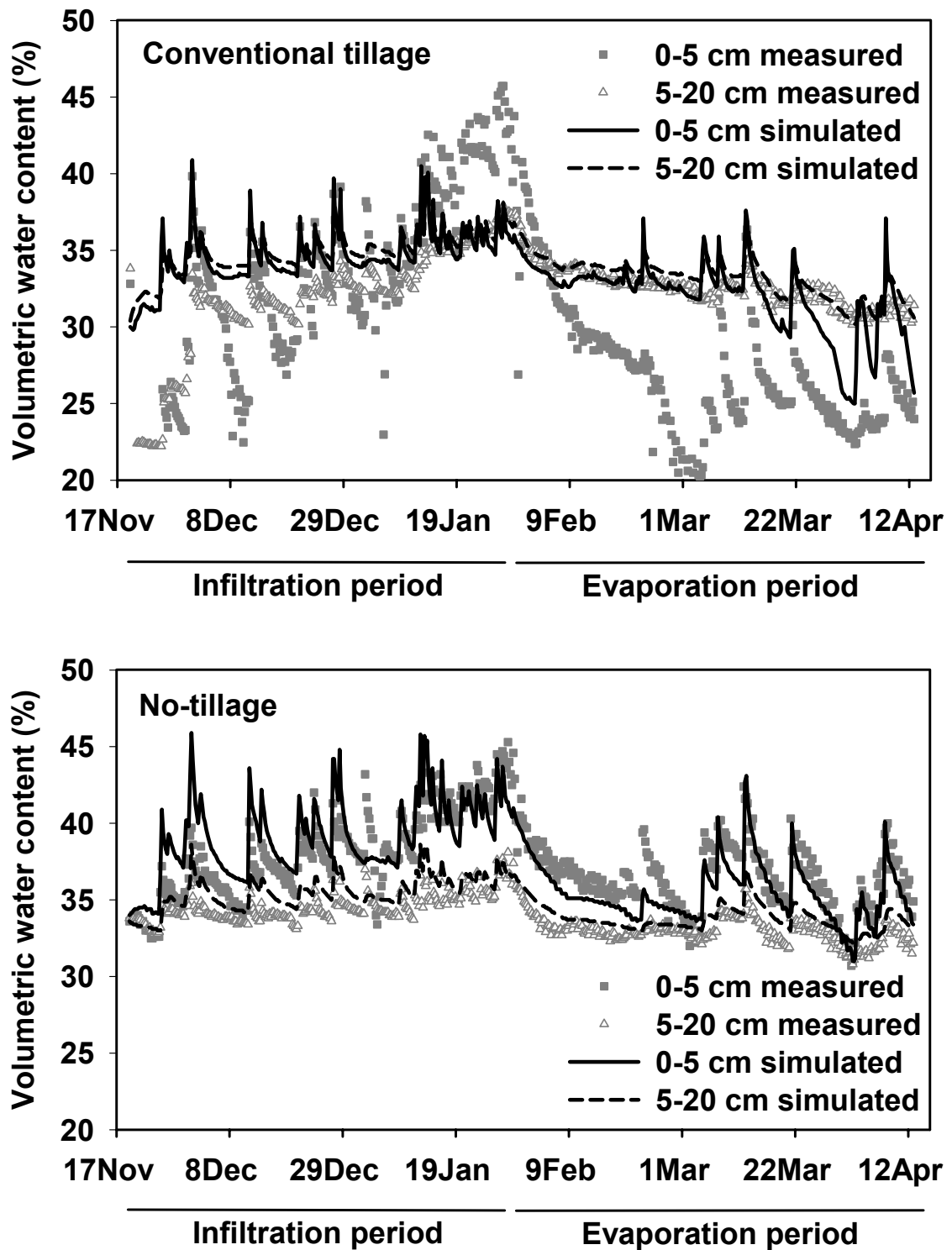


Figure 6.1 : Measured and simulated water contents of the 0-5 and 5-20 cm soil layers of the conventional tillage (CT) and no-tillage (NT) plots.

Table 6.7 : Model efficiency (EF) and mean difference (MD) for the 0-5 and 5-20 cm soil layers of the conventional tillage (CT) and no-tillage (NT) plots

Volumetric water content	CT ₀₋₅	CT ₅₋₂₀	NT ₀₋₅	NT ₅₋₂₀
<i>EF</i> (-)	0.036	-0.175	0.588	0.263
<i>MD</i> (cm ³ 100 cm ⁻³)	-3.84	-1.88	0.03	-0.87
Temperature	CT ₀₋₅	CT ₅₋₂₀	NT ₀₋₅	NT ₅₋₂₀
<i>EF</i> (-)	0.980	0.992	0.991	0.998
<i>MD</i> (°C)	0.06	0.16	0.24	0.05
Nitrate content	CT ₀₋₅	CT ₅₋₂₅	NT ₀₋₅	NT ₅₋₂₅
<i>EF</i> (-)	0.976	0.948	0.583	0.864
<i>MD</i> (kg N ha ⁻¹)	0.18	-0.48	0.44	-1.83
CO ₂ emission	CT		NT	
<i>EF</i> (-)	0.064		0.542	
<i>MD</i> (kg C ha ⁻¹ d ⁻¹)	-0.16		-0.02	

6.3.2 Water content of the mulch

The simulated volumetric water content of the mulch remained close to saturation during the infiltration period because evaporation was not large enough to evaporate mulch water between two rainfalls. In contrast, large fluctuations were simulated during the evaporation period (Figure 6.2).

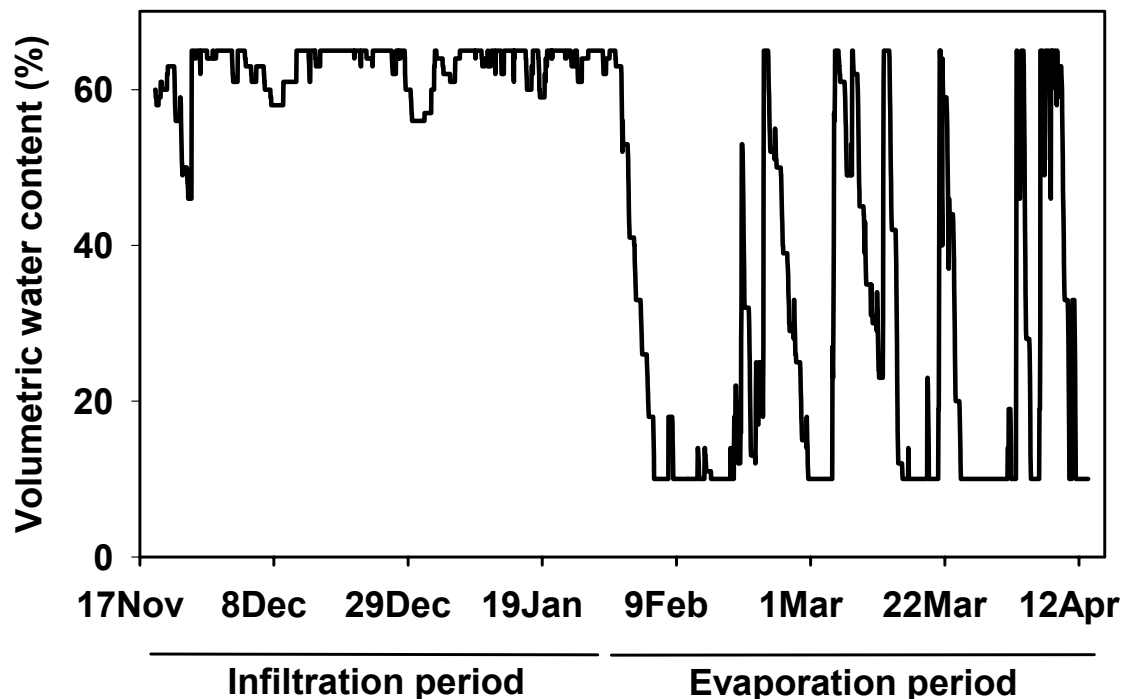


Figure 6.2 : Simulated water contents of the surface mulch of the no-tillage (NT) plot.

6.3.3 Total evaporation and water drainage

Total evaporation is defined as the sum of the evaporation from the soil and the mulch. During the infiltration phase, the cumulative total evaporation and water drainage at 25 cm depth simulated by the model were comparable between the two tillage systems (Figure 6.3). During the evaporation phase, cumulative total evaporation was lower whereas cumulative water drainage was larger in NT compared to CT. Over the entire simulation period, NT presented a 38% smaller cumulative total evaporation and a 33% larger cumulative water drainage. Total evaporation was lower due to a reduction of soil evaporation by the mulch

layer. The difference in total evaporation between the two tillage systems (38 mm) was considerably larger than the maximum water storage of the mulch (4 mm). This means that more water entered the soil in NT than in CT which generated the greater water drainage at 25 cm depth in NT. The value of 0.6 for the reduction of soil evaporation by the mulch is greater than the value found by Findeling (2001), but this is consistent with a denser mulch in our experiment. The 100 % surface cover of crop residues estimated from photographs (see Appendix B) was in accordance with the surface cover observed by Steiner *et al.* (2000) for similar masses of flat wheat residues.

6.3.4 Temperature dynamics

The temperature dynamics at 2.5 and 12.5 cm soil depth (not represented) were very well simulated with a model efficiency higher than 0.97 (Table 6.7). The hourly temperatures at 2.5 cm soil depth fluctuated between -4°C and 19°C.

6.3.5 Crop residue decomposition

Due to the presence of old weathered residues of previous years (Chapter 5), the initial amount of C in crop residues was much larger in NT (3430 kg C ha⁻¹) than CT (1464 kg C ha⁻¹) (Figure 6.4). In CT, only crop residues of the recent wheat harvest were present. Simulated and measured C decomposition corresponded well both in CT and NT. During the infiltration period, the simulated C decay in NT (809 kg C ha⁻¹) is nearly double that in CT (471 kg C ha⁻¹), whereas during the evaporation period, C decay is more comparable between NT (429 kg C ha⁻¹) and CT (319 kg C ha⁻¹). However, in both periods, the relative decay rate was smaller in NT than in CT: 24% compared to 32% during the infiltration period and 13% compared to 22% during the evaporation period, respectively. The slower decomposition is consistent with the accumulation of weathered surface residues in NT. This means that the greater amount of crop residues in NT offset the slower residue decomposition. The slower C decay in NT may partly result from the presence of old residues that decompose slower than recent added residues (Chapter 5) and/or from a (slightly) larger limitation of the decomposition by a mineral N deficiency due to reduced soil contact in NT compared to CT. However, the difference in soil contact between the two tillage systems was reduced because residues were incorporated in groups in CT instead of being homogeneously distributed over the ploughed layer (Henriksen and Breland, 2002). The effect of soil climate on the decomposition rate of crop residues will be discussed in the paragraph about the CO₂ fluxes.

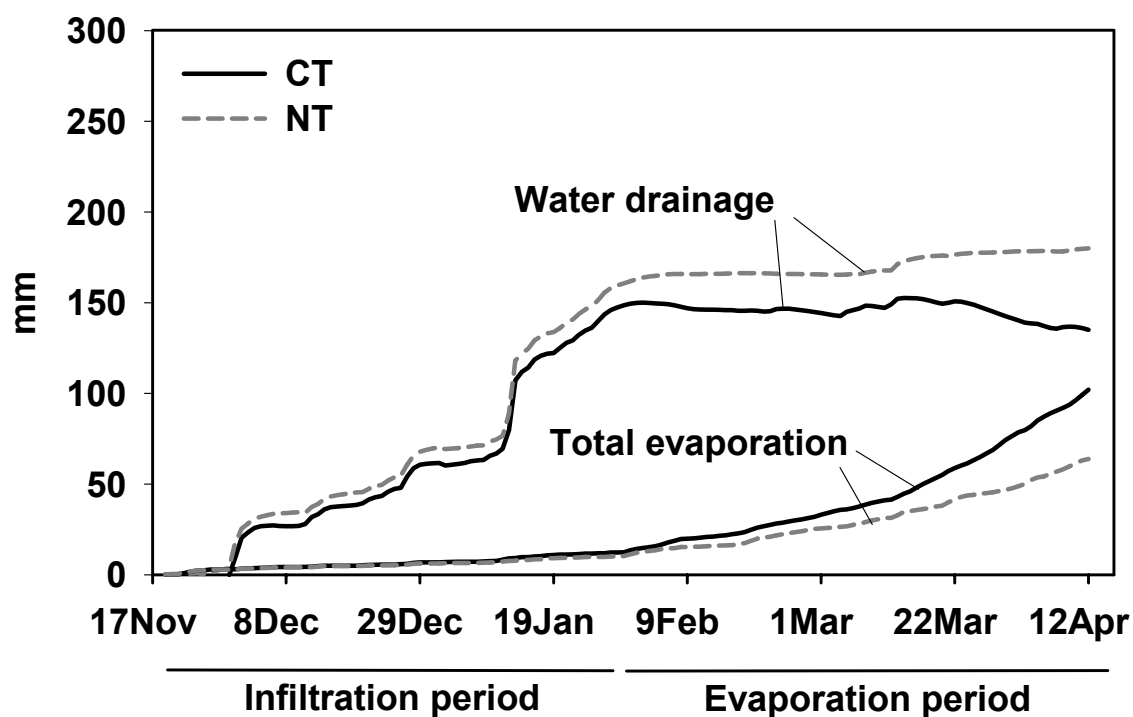


Figure 6.3 : Simulated cumulative total evaporation at the top and water drainage at the bottom of the 0-25 cm soil profile of the conventional tillage (CT) and no-tillage (NT) plots.

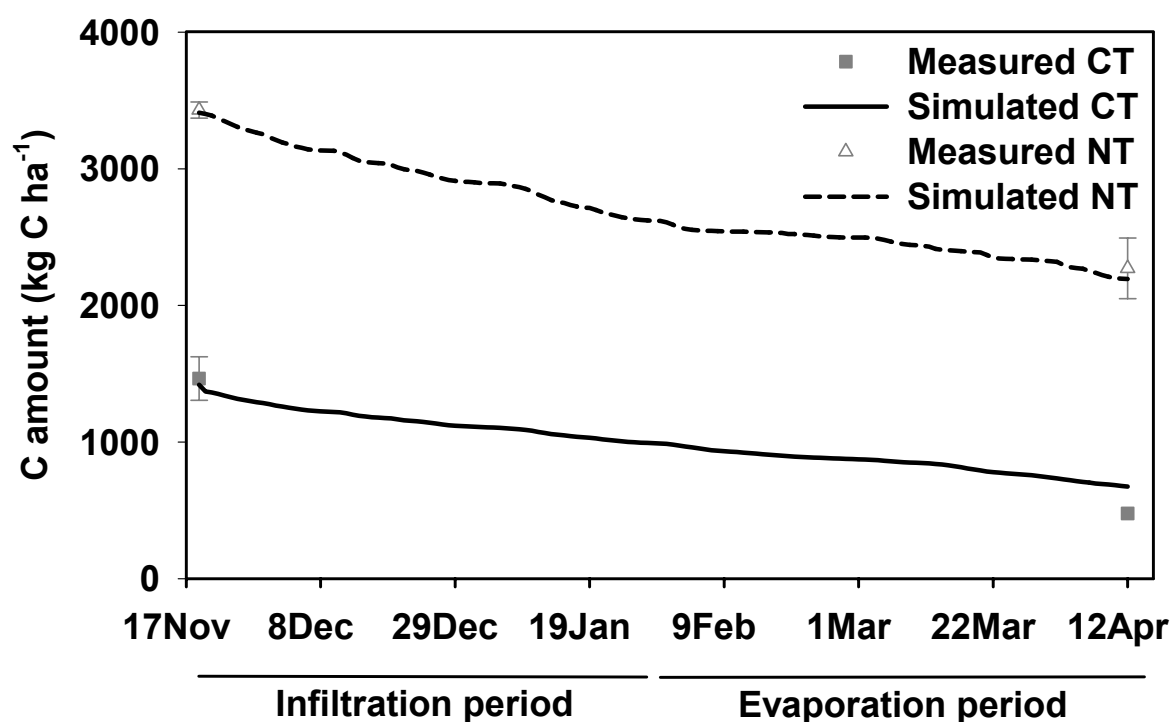


Figure 6.4 : Measured and simulated amounts of carbon in crop residues of the conventional tillage (CT) and no-tillage (NT) plots.

6.3.6 Soil inorganic N

The mineral nitrogen content in the soil profile is the result of (i) the N mineralisation and immobilisation processes during the decomposition of SOM and crop residues and (ii) the transport processes of inorganic N (N leaching and convective transport by capillary rise).

The evolution of the nitrate content in the 0-5 and 0-25 cm soil layers was first simulated using a 'standard' procedure with a 1D simulation model, i.e. assuming that the crop residues are homogeneously distributed over the horizontal plane. This procedure provided a good simulation for NT, but not for the CT plot (Figure 6.5). The model underestimated the amounts of soil mineral N in the CT plot, particularly in the 5-25 cm layer. The model was thought to overestimate the amount of immobilised N, probably because it overestimated the availability of mineral N. Indeed, the 1D model assumes that all mineral N present at a given depth is available for residue decomposition at this depth. This hypothesis is likely to be false, mainly because crop residues incorporated by mouldboard ploughing are not evenly distributed throughout the soil volume, but situated in residue clusters in zones with high porosity between two adjacent furrows as also observed by Staricka *et al.* (1991). Gaillard *et al.* (1999) determined that straw incorporation increases the microbial activity only in a zone of 0.4 cm around the crop residues. This suggests that only the soil N_{min} near the residues was available for the microbial biomass that decomposed the residues. The mineral N situated further from the residues was not available for residue decomposition. The impact of this heterogeneity in residues location was tested with our 1D model by assuming that the modelled soil volume in the CT was composed of two vertical soil strips: a first strip free of crop residues, and a second strip containing all the crop residues. We assumed that the first strip contained 90% of the soil volume and the second strip 10% of the soil volume. These figures are consistent with the observations made in the field. The simulations were run separately for each strip and the outputs were summed up according to the respective soil volume of each strip. The model thus parameterised was able to reproduce the observed soil nitrate content well (Figure 6.6). The mineral N had disappeared completely in the strip with residues due to N immobilisation, whereas mineral N accumulated in the strip without residues due to net mineralisation. The residue decomposition rate was markedly reduced due to N limitation. The model assumed no mineral N exchange between the two strips so that the nitrate present in the strip without residues was not available for residue decomposition in the strip with residues.

In NT, decomposition of the surface residues was also limited by the amount of mineral N since only the first centimetre of soil could provide mineral N. Fungi form a bridge between the mulch layer and the soil and translocate N_{min} from the surface soil to the mulch, while C is transferred from the mulch to the surface soil (Frey *et al.*, 2003). The model responded to the

mineral N limitation by the simulation of both a reduction in decomposition rate and an increase in the C:N ratio of the decomposing (zymogeneous) biomass. These changes resulted in a reduced decomposition of residues and lower CO₂ emissions compared to non limiting conditions. The model predicted a cumulative gross N mineralisation of 171 kg N ha⁻¹ for CT and 164 kg N ha⁻¹ for NT; a cumulative gross N immobilisation of 147 kg N ha⁻¹ for CT and 130 kg N ha⁻¹ for NT; and a cumulative net mineralisation of 24 kg N ha⁻¹ for CT and 34 kg N ha⁻¹ for NT.

The calculated amount of cumulative nitrate leached below 25 cm soil depth was 22 kg N ha⁻¹ for CT and 45 kg N ha⁻¹ for NT over the entire study period. The larger N leaching in NT was mainly the result of the larger initial nitrate content and the larger amount of drainage. For both tillage systems, the N leaching during the evaporation period was negligible. There is no consensus about the effect of tillage on N leaching in literature (Elliott and Coleman, 1988; Germon *et al.*, 1994; Shipitalo *et al.*, 2000).

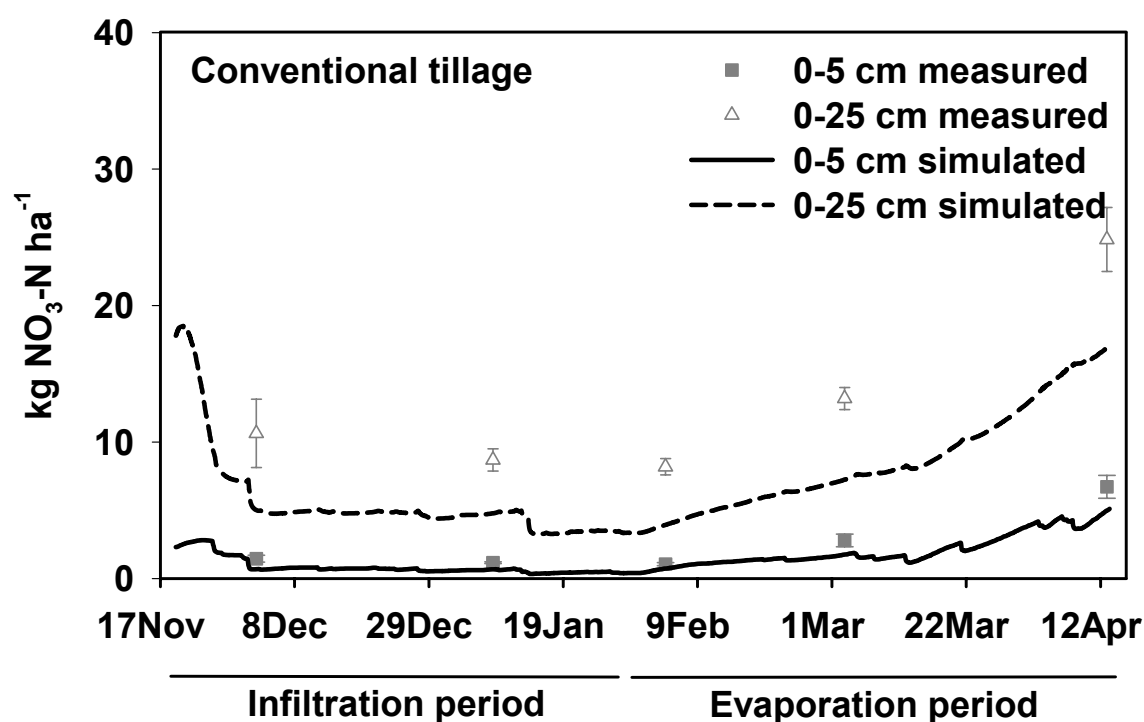


Figure 6.5 : Measured and simulated nitrate contents in the 0-5 and 0-25 cm soil layers in the conventional tillage (CT) plots under the assumption that crop residues in the PASTIS are homogeneously distributed model over the entire soil volume.

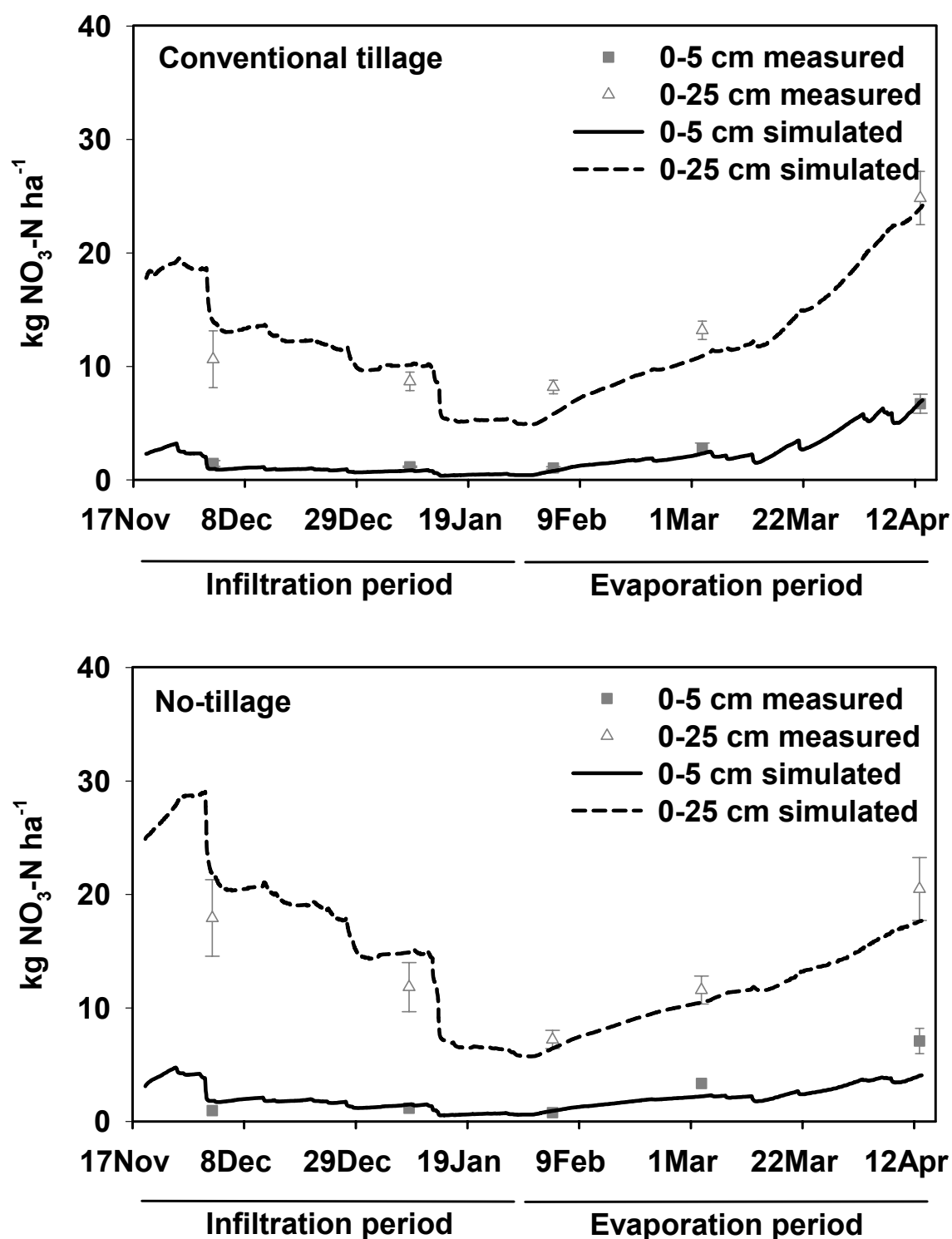


Figure 6.6 : Measured and simulated nitrate contents in the 0-5 and 0-25 cm soil layers in the conventional tillage (CT) and no-tillage (NT) plots, under the assumption that crop residues in CT are located in 10% of the entire soil volume.

6.3.7 CO₂ emissions

The simulations of CO₂ production were also initiated with the 'standard' procedure assuming an even distribution of crop residues (one soil volume). They lead to the same conclusions than for mineral N: the simulations were quite satisfactory for NT (Figure 6.7), but they overestimated the cumulative CO₂ production in the CT plot (not shown). The simulations conducted with two soil volumes (two vertical strips) again resulted in very good simulations in the CT plot (Figure 6.7). It is noteworthy that the cumulative CO₂ production was greater in NT than in CT plot.

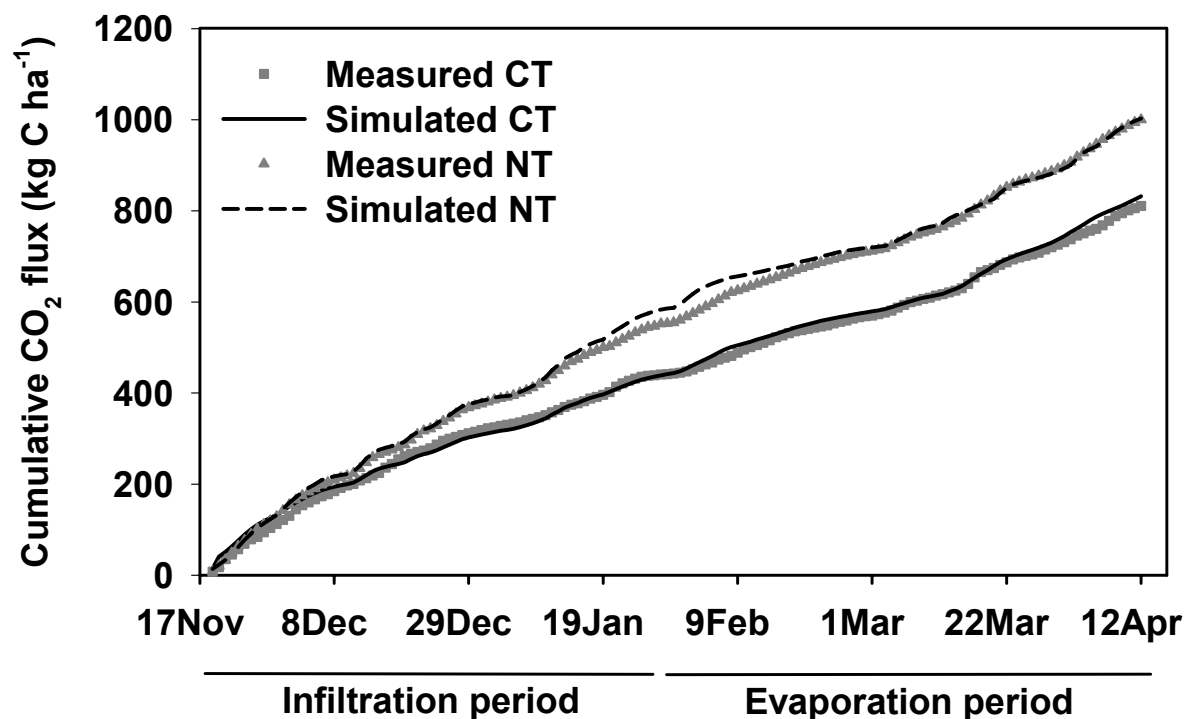


Figure 6.7 : Measured and simulated cumulated CO₂ emission in the conventional tillage (CT) and no-tillage (NT) plots, under the assumption that crop residues in CT are located in 10% of the entire soil volume.

Considering the daily CO₂ fluxes in the NT plot, both the magnitude and the fluctuations of the daily CO₂ emissions were very well simulated (Figure 6.8). However, in CT, the timing and the magnitude of the peaks or dips of the daily CO₂ emissions were not always well reproduced. The measured CO₂ emissions (normalised for a temperature of 15°C) decreased in the CT plot during and shortly after rainfall events (Chapter 5). This may be explained by

reduced gas diffusivity in wet soils (Currie, 1984), i.e. CO₂ gases were captured in the soil profile. This captured CO₂ gas escaped from the soil profile between rainfall events (Chapter 5). The discrepancies between the simulated and observed daily CO₂ emissions in CT can be explained by the fact that the PASTIS model does not account for reduced gas diffusivity in wet soils. Although the model may simulate well CO₂ production by the soil, it may overestimates CO₂ fluxes soon after rainfall and underestimate CO₂ fluxes later between rainfall events. This probably explains why the cumulative CO₂ fluxes were well simulated in CT. In NT, the reduction in gas diffusivity had less impact because most of the CO₂ was produced in the mulch layer and near the soil surface (0-5 cm soil layer).

The model was used to analyse the origins of the CO₂ fluxes. It calculated that about half of the CO₂ was produced during the decomposition of stabilised organic matter (SOM) and the other half during the decomposition of crop residues (Figure 6.9). The CT and NT treatments presented the same CO₂ emissions as a product from SOM decomposition (Figure 6.9), even if we consider the daily CO₂ emissions (Figure 6.10). The potentials of CO₂ emission from SOM measured under controlled laboratory conditions for the 0-25 cm soil profile were also similar between CT (1042 ± 47 kg CO₂-C ha⁻¹) and NT (980 ± 71 kg CO₂-C ha⁻¹) (Chapter 5). This suggests that during the simulation period, soil climatic conditions had a similar effect on SOM decomposition in CT and NT. Indeed, the influence of soil temperature on the differences between CT and NT was negligible during the simulated period (Chapter 5). Furthermore, most of the time, the reduction function on SOM decomposition due to water content calculated by the model varied between 0.9 and 1, indicating that water conditions were close to optimum for decomposition in both systems. However, differences in water content and soil temperature between CT and NT may play a significant role on SOM decomposition in other periods of the year (Chapter 5). Despite similar CO₂ emissions from SOM for the entire 0-25 cm soil profile, differences between CT and NT were calculated in the location of the production of CO₂. This CO₂ production was more pronounced near the soil surface in NT, whereas it was more homogeneously distributed over the whole 0-25 cm layer in CT (in accordance with the observed distribution of the SOM emissions potentials found in the laboratory incubations reported in Chapter 5).

The simulated cumulative CO₂ fluxes produced during crop residue decomposition were 51% larger in NT than CT (Figure 6.9). In spite of the slower residue decomposition in NT, the greater amount of crop residues (residues from previous years) resulted in a larger C decay. The CO₂ produced during crop residue decomposition over the entire infiltration period account for 81% of the difference observed over the entire simulation period. During this infiltration period, the daily emissions followed the same dynamics in CT and NT, but their magnitude was larger in NT (Figure 6.10). During the evaporation period, the simulated

peaks of daily CO₂ emissions did not occur at the same moment in time in CT and NT. During this period, daily emissions from residues in NT followed the same dynamics as the water content of the mulch layer (Figure 6.2). Soil temperature had a similar effect on the surface and incorporated residues during the simulation period (Chapter 5). However, the water content of the crop residues largely controlled the magnitude of the difference in CO₂ emissions between CT and NT. When the water content of the mulch remained optimal for decomposition (i.e. during the infiltration period), CO₂ emissions in the NT system were much larger than in CT. During the evaporation period, however, the low water content of the mulch limited the residue decomposition in NT, thereby decreasing the difference in CO₂ emissions between CT and NT. This resulted in bursts of CO₂ emissions in the NT plot, whereas emissions were more constant in CT. Note that, despite the larger C amounts in NT, severe limitation of the mulch decomposition by water sometimes resulted in larger CO₂ emissions in CT (where abiotic factors remained favourable) than in NT, e.g. in dry periods with high evaporation. The effect of the water content of the residues is confirmed in literature. Abiven and Recous (submitted) measured no difference in cumulative CO₂ emissions between surface and incorporated wheat straw under optimal moisture, temperature and nitrogen conditions in the laboratory. Under variable moisture conditions, Coppens *et al.* (submitted) observed lower cumulative CO₂ emissions for surface compared to incorporated residues in the laboratory, mainly due to a difference in water content of the residues.

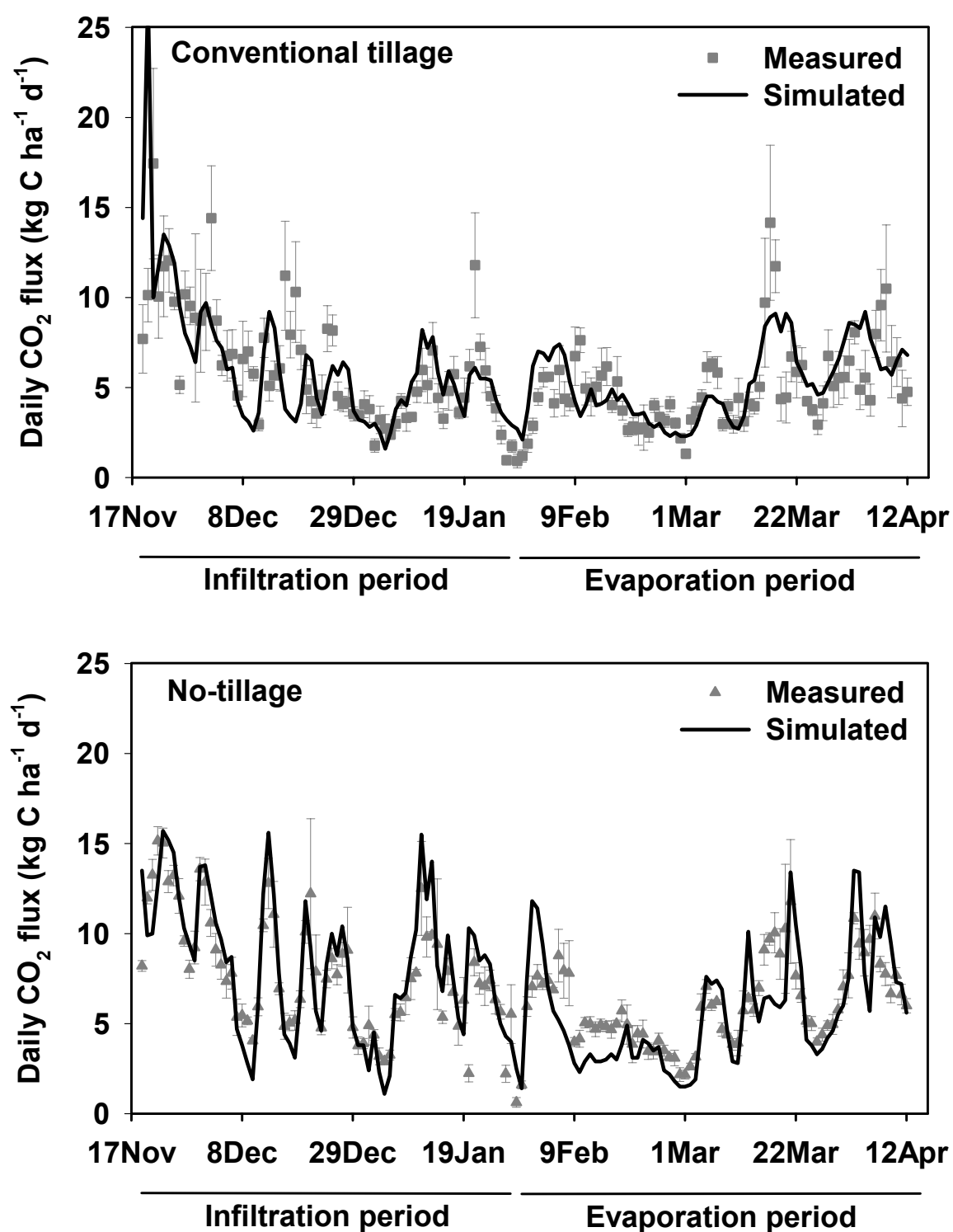


Figure 6.8 : Measured and simulated daily CO₂ emissions in the conventional tillage (CT) and no-tillage (NT) plots, under the assumption that crop residues in CT are located in 10% of the entire soil volume. Bars indicate standard errors on the measured values.

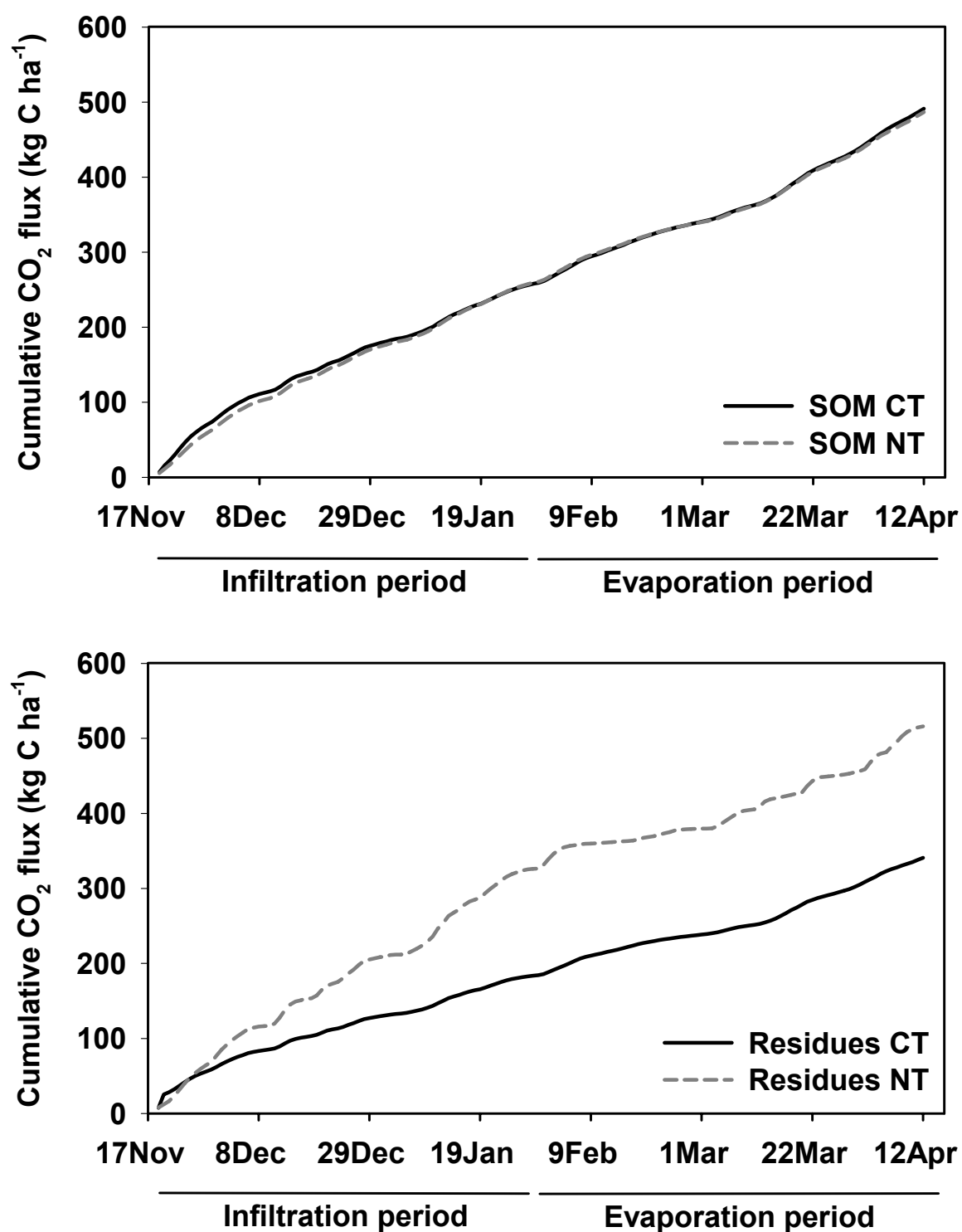


Figure 6.9 : Simulated cumulated CO₂ emissions as a product from the decomposition of soil organic matter (SOM) and crop residues in the conventional tillage (CT) and no-tillage (NT) plots, under the assumption that crop residues in CT are located in 10% of the entire soil volume.

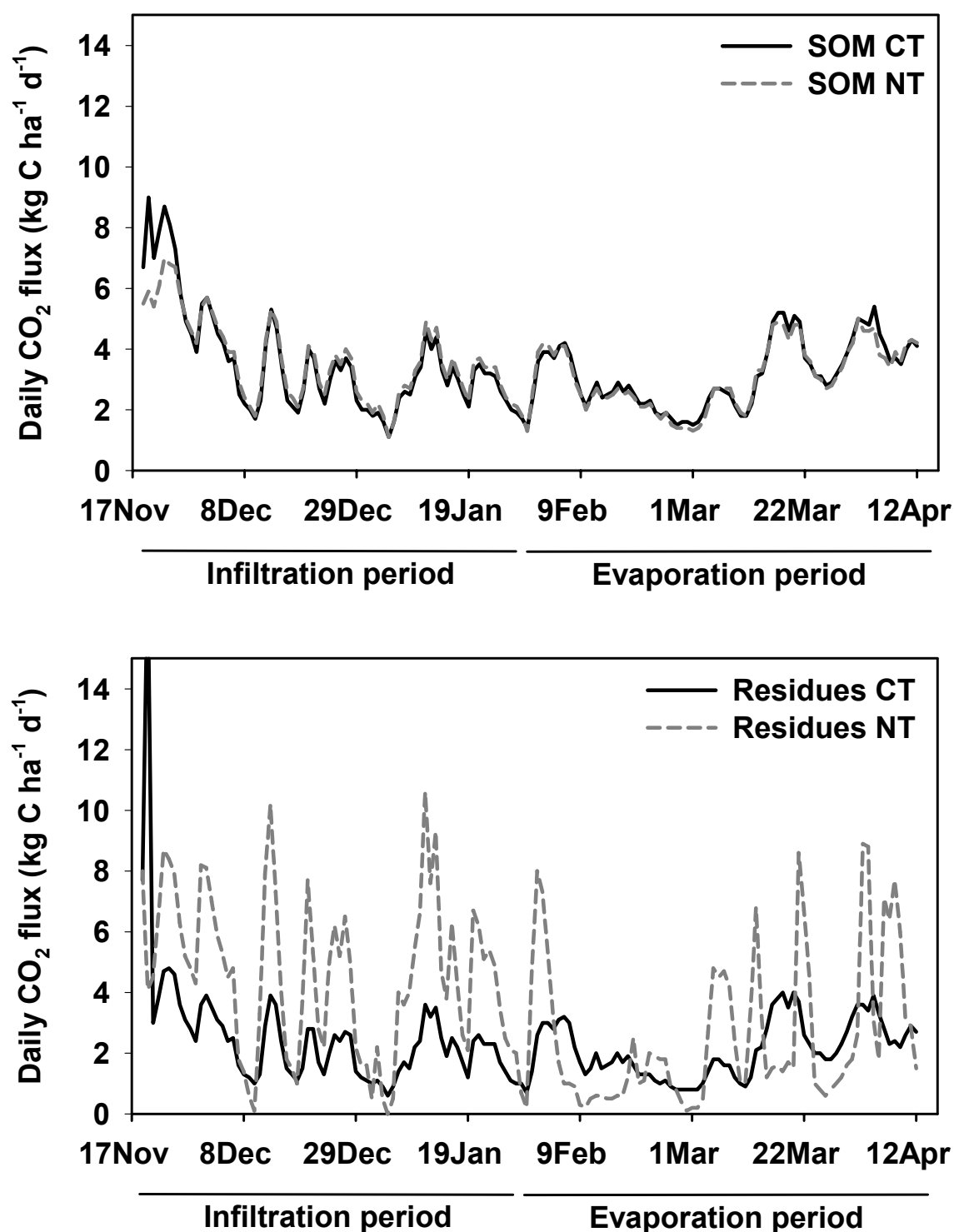


Figure 6.10 : Simulated daily CO₂ emissions as a product from the decomposition of soil organic matter (SOM) and crop residues in the conventional tillage (CT) and no-tillage (NT) plots, under the assumption that crop residues in CT are located in 10% of the entire soil volume.

6.4 CONCLUSIONS

The effect of the suppression of ploughing on soil C and N fluxes is the result of complex interactions of modified soil climate and biological and physical soil properties. Modelling allows isolating the sole effect of one modification. In this study, the feasibility of the PASTIS model was tested to model C and N fluxes in real farming conditions with conventional mouldboard ploughing (CT) and no-tillage (NT) systems differentiated for 32 years. Afterwards, by means of simulations, the influences on the C and N fluxes by the soil temperature, soil water content and both the quantity and the localization of soil organic matter and crop residues were determined. The model was able to provide good simulations of the evolution of soil water content, soil temperature, CO₂ emissions, crop residue decomposition and soil mineral nitrogen over a 5 months period in the two tillage systems. The measurement and simulation of the soil water content of CT turned out to be difficult, due to the large spatial and temporal variations of both soil structures and properties directly after mouldboard ploughing. The simulation of the NT treatment was easier to carry out because of the more homogeneous soil structure.

The mulch layer exerted a large influence on the water dynamics: evaporation was reduced while the water drainage increased. Cumulative and daily CO₂ emissions produced during soil organic matter (SOM) decomposition were similar in both tillage systems which was a result of similar CO₂ emission potentials and similar regimes of soil temperature and water pressure head. The observed difference in CO₂ emissions between the two tillage systems (larger in NT) was due to crop residue decomposition. Despite a slower decomposition of crop residues in NT, the larger amount of crop residues gave rise to a larger amount of C decay in NT. This larger amount of crop residues in NT was due to the presence of weathered residues from previous crops resulting from the slower residue decomposition in NT. No large temperature differences were observed between the surface residues in NT and the incorporated residues in CT. The water content of the mulch had the largest influence on the difference in decomposition rate of crop residues between CT and NT. Between rainfall events in periods with larger amounts of evaporation, the rate of residue decomposition in NT was reduced compared to CT due to the low water content of the surface residues.

For long-time differentiated CT and NT fields, the simulation results showed that the large amount of accumulated organic matter on the soil surface in NT can counterbalance the slower decomposition rate and result in larger CO₂ emissions in NT than CT under specific climatic conditions. Finally, as the climatic variables had a large influence on the simulated CO₂ fluxes between CT and NT, it would be interesting to run the model with data from fields under other climatic conditions.

CHAPTER 7: GENERAL CONCLUSIONS AND RECOMMENDATIONS

7.1 General conclusions

The overall objective of this work was to quantify and understand the differences in C and N pools and fluxes between different long-term (32 year) tillage systems in cereal cropping systems in Northern France. The work focused on two extreme tillage systems, i.e. mouldboard ploughing to 20 cm depth (CT) and no-tillage (NT). For some investigations, an intermediate tillage system was also included, namely non-inversion superficial tillage (ST) to 10 cm depth. These tillage systems were studied on two different sets of plots of the same experimental site but at different periods of time, indicated by period I and period II. Finally, this work focused mainly on those parameters having important agronomical or environmental impacts: soil organic C and N, soil mineral N dynamics and CO₂ and N₂O emissions.

7.1.1 Quantity and quality of SOM stocks (research question I)

Soil organic C and N stocks were calculated for an equivalent soil mass of dry soil corresponding to the plough layer before tillage differentiation. After 32 years of tillage differentiation, NT presented 5-15% larger C stocks and 3-10% larger N stocks compared to CT. However, these differences were only significant for plots studied during period I, not for plots examined during period II (Table 5.3). In NT, soil organic C and N concentrations decreased with increasing depth whereas, in CT, these concentrations were rather homogeneously distributed over the plough layer.

The stock differences for the period I plots were further investigated by examining the changes at different levels of structural complexity, i.e. organic matter fractions, aggregation and pore size distribution. Mineral-associated N and particulate organic matter each accounted for about 50% of the total difference in N stock. However, 66% of the total difference in C stock was due to differences in the particulate organic matter (58%) and free residues (8%) fractions. In NT, additional C and N was detected in macroaggregates, whereas CT contained more C and N that was not occluded in aggregates. In CT, no differences in

mineral-associated OM, POM, aggregate size distribution and the composition of aggregates were found within the 0-20 cm plough layer, whereas large differences were found between the 0-5 and 5-20 cm layers of NT. Our results suggest that the larger C and N stocks in NT can be attributed to i) enhanced macroaggregate formation in the 0-5 cm layer due to higher microbial activity and SOM content and ii) a better protection of soil organic matter in the 5-20 cm layer due to a larger proportion of small pores and lack of soil disruption by tillage or climate.

7.1.2 Soil mineral N supply and N leaching (research question II)

Water and mineral N contents of several soil layers were measured at several dates in the CT, ST and NT plots during the bare period between wheat and maize crops (period II). Using these data, N mineralisation and N leaching were calculated by the LIXIM model. Afterwards, 'in situ' potential N mineralisation, expressed as a function of normalised days at 15°C and soil moisture content at field capacity, was calculated. The tillage systems did not induce large differences in water and nitrate content. In addition, LIXIM calculated comparable N mineralisation and potential N mineralisation rates between the three tillage systems. These calculations were in accordance with the comparable potential N mineralisation obtained in the laboratory between the CT and NT plots of period II. These results offer clear evidence that soil N supply is comparable between the three tillage systems for the specific combination of the soil, climate and cropping system. This contradicts our initial hypothesis of a smaller net mineralisation in reduced tillage systems. The amount of N leaching tended to be larger in the NT and ST compared to the CT plots due to larger water drainage and probably more leaching of the mineralised N by preferential flow. However, in the presence of plants, plant uptake of mineral N would likely reduce this difference in N leaching between the tillage systems.

7.1.3 CO₂ and N₂O emissions (research question III)

Although no-tillage is often proposed as a means of decreasing CO₂ emissions, these have rarely been continuously measured in long-term differentiated tillage systems. Regarding N₂O emissions, it is often stated that conventional tillage systems are preferable to no-tillage systems. In our experiments, CO₂ emissions were continuously measured in CT and NT from the end of April 2003 to mid-April 2004. N₂O emissions were periodically measured during the same period. As expected, NT always tended to emit more N₂O than CT throughout the entire measurement period, although only 50 percent of the differences were significant due to considerable variability. N₂O emissions were generally less than 5 g N ha⁻¹ day⁻¹, except for

three days when emissions increased to $21 \text{ g N ha}^{-1} \text{ day}^{-1}$. Differences in CO_2 emissions between NT and CT treatments were significant on 59% of the days. Depending on weather conditions (rainfall and temperature) and the amount and location of residues, either CT or NT could emit the larger amount of CO_2 . Inconsistent with our initial hypothesis, the cumulated CO_2 emissions for the specific climatic conditions of the measurement year were significantly larger for NT than for CT. As a consequence, regarding the emission of the greenhouse gases CO_2 and N_2O in the long-term tillage systems, NT was certainly not preferable to CT under the specific climatic conditions of the measurement year.

7.1.4 Are the differences in C and N stocks and fluxes in CT and NT the result from the potential decomposability of the SOM and/or physical protection of SOM? (research question IV)

Potential decomposability of SOM was measured during laboratory incubations under controlled temperature and moisture conditions. For the different tillage systems, potential C and N mineralisation was measured on fresh soil samples after minimal disruption of soil structure (sieving through 12.5 mm) in order to minimise any influence on existing SOM protection. In the period I plots, the 'in situ' heterogeneity of soil structure was taken into account during sampling. Two megastructure zones induced by tillage and compaction were identified in the ploughed layer of CT: zones with loose structure (CT_{Loose}) and clods with dense structure (CT_{Dense}). The soil samples in NT were taken from the 0-5 cm and 5-20 cm soil layers that differed both in structure and organic matter content (NT_{0-5} and NT_{5-20}).

Undisrupted soil from NT_{0-5} showed greater potential C and N mineralisation than CT_{Loose} and CT_{Dense} , which, in turn, had a slightly greater potential C and N mineralisation than NT_{5-20} . Throughout the entire equivalent plough layer depth, total potential C and N mineralisation was significantly larger for NT compared to CT, although the differences were small. As the specific C and N mineralisation values were also slightly higher for NT, the larger amounts of OM in NT could only partially explain the higher mineralisation. For plots studied during period II, both total and specific C and N mineralisation did not differ significantly between CT and NT. This may be due to the smaller differences observed in C and N stocks for these plots compared to the period I plots. These findings contradict our initial hypothesis and clearly showed that potential C and N mineralisation was not smaller in NT compared to CT after 32 years of tillage differentiation for the specific combination of soil, climate and crop rotation.

It is often stated that physical organic matter protection against C and N mineralisation is responsible for the larger C and N stocks in NT. Therefore, for the four structural zones

(CT_{Loose}, CT_{Dense}, NT₀₋₅ and NT₅₋₂₀) of the CT and NT plots followed in period I, the physical protection of soil organic matter was evaluated at different scales of soil structure. This was done by incubating soil samples in which the structure had been progressively destroyed by sieving through decreasing mesh sizes (12.5 and 2 mm, 250 and 50 μm). Limited soil structure destruction (sieving through 2 mm) had no effect on C and N mineralisation. Increased disturbance (sieving down to 250 μm) only induced a significant increase of both C and N mineralisation in the 5-20 cm layer of NT. Still further disruption of soil structures (sieving through 50 μm) resulted in greater C and N mineralisation for all treatments except C mineralisation in the upper layer of NT. The effect of physical protection of SOM in the four structural zones in CT and NT was, in general, greatest in the deeper layer of NT and smallest or absent in the upper layer of NT. The extra mineralisation that occurred after soil disruption in the four structural zones was in good agreement with the relative amount of micro-porosity in each zone. In CT, soil from both zones of different megastructure showed similar levels of protection and similar C and N mineralisation. These results indicate that — in contradiction to our initial hypothesis — the structural zone with the largest C and N stocks and the largest amount of water stable aggregates (NT₀₋₅) showed after soil structure disruption the smallest increase in N mineralisation and no increase in C mineralisation. Thus, it is likely that the large C and N concentrations in the 0-5 cm soil layer are determined by other mechanisms than physical OM protection. The surplus C and N in this structural zone after 32 years of differentiation is probably (i) situated in recalcitrant OM, which does not induce an increase in C and N mineralisation when liberated, or (ii) mainly located in a rather labile form which is not highly protected because they both do not induce an increase in C mineralisation when the structural soil elements are broken.

7.1.5 Do the differences in C and N stocks and fluxes in CT and NT result from ‘in situ’ changes in the localization and amounts of SOM and crop residues, modifications of the soil properties, or the changes of soil climatic conditions? (research question V)

Regarding **SOM decomposition**, NT presented larger potential C and N mineralisation near the soil surface, whereas in CT, the potential C and N mineralisation was rather homogeneously distributed over the 0-25 cm soil profile. However, as mentioned above, differences between CT and NT throughout the entire 0-25 cm soil profile were small. Because soil temperatures in the CT plot were higher than those in NT from June to the end of September, soil temperature was slightly more favourable to ‘in situ’ SOM decomposition in

CT than in NT. During colder periods, soil temperature was comparable between CT and NT. The difference in soil water content also had an effect on the 'in situ' SOM decomposition in the 0-25 cm soil profile. In NT, the soil remained wet for a longer period of time due to reduced soil water evaporation in the presence of the surface residues. On the other hand, NT soil rewetted more slowly after a dry period. Due to the different distribution of the SOM potential C and N mineralisation within the equivalent plough layer, it was not feasible to compute the overall effect of water content and temperature on 'in situ' C and N mineralisation of SOM throughout the entire 0-25 cm soil profile without the help of a complex model. However, these findings suggest that — considering the entire soil 0-25 cm profile — 'in situ' SOM decomposition may be different between CT and NT, but the differences are often small and not consistently larger over time in one tillage system.

Despite the slower decomposition rate of the crop residues in NT, its larger amount induced a larger total **residue decomposition** compared to that in CT. This larger amount of crop residues in NT was due to the presence of residues from previous years, which, in turn, was explained by the slower residue decomposition rate in NT.

The distribution and amount of rainfall and water evaporation had a large influence on the dynamics of the CO₂ fluxes. The results indicate that, depending on these factors, higher emissions may be measured in CT (with incorporated residues) compared to NT (with surface residues) during certain periods, while the opposite may be true during other periods. In NT, rainfall induced large residue decomposition and, consequently, bursts of CO₂ emissions because the water content of the surface residues was suddenly increased. However, after the occurrence of rainfall, the water content of the surface residues fell very rapidly and, again, seriously limited their decomposition. These fluctuations in water content of the surface residues were more pronounced in periods with high evaporation. In CT, residues are incorporated and have less fluctuations of water content. Moreover, under rainfall and wet conditions, CO₂ fluxes in CT are captured in the soil profile due to reduced gas diffusion. When the profile drains after rainfall, this captured CO₂ escapes from the soil profile resulting in large CO₂ emissions in CT.

In order to better analyse the long-term effects of tillage systems, C and N fluxes were simulated using the PASTIS model. **Modelling** permits (i) to take the complex interactions between the different determining factors into account and (ii) to isolate the individual effect of factors. For example, the PASTIS model was able to calculate the proportion of CO₂ produced from the decomposition of SOM and crop residues. Modelling was done for the period after soil tillage in period II. The model was able to provide good simulations for the soil water content, the temperature dynamics, the CO₂ emissions, the residue decomposition and the mineral nitrogen evolution. So, by means of simulations with the PASTIS model, the

individual influences on the C and N fluxes by the soil temperature, soil water content and both the quantity and the localization of soil organic matter and crop residues could be determined.

The modelling showed that the larger cumulative total CO₂ fluxes in NT were the result of larger CO₂ emissions as a product of crop residue decomposition and not as product of SOM decomposition. For these long-term differentiated tillage systems, the larger amount of accumulated residues of previous crops in NT more than compensated the slower residue decomposition rate of surface compared to incorporated residues. It was the water content of the surface crop residues that largely controlled the magnitude of the difference in decomposition rate of the crop residues between CT and NT. The influence of soil temperature and soil water content was comparable between CT and NT throughout the simulation period.

The model also showed that the decomposition of both surface and incorporated residues was limited by the availability of mineral N. The decomposing microbial biomass of surface residues was limited to the mineral N present in the first centimetre of the soil profile. Since the residues in CT were incorporated in groups between adjacent furrows and not homogeneously distributed within the plough layer, the microbial biomass was limited by the amounts of mineral N present in the hot spots concerned by the decomposition of residues (estimated to be 10% of the plough layer volume). Effectively, the microbial biomass can only use the soil mineral N at a few millimetres from the residues. This finding is of significant importance since it suggests that experiments with homogeneously incorporated crop residues will overestimate the decomposition of incorporated residues in real mouldboard ploughing systems.

7.2 Recommendations

The conclusions of this work were drawn for one specific year in a specific maize-wheat rotation in one specific soil under one specific climate. The differences in C and N fluxes and the organic C and N accumulation change from year to year depending on the ratio of C input to C losses. During our period, the temperatures and the potential evapotranspiration from mid-May to mid-August 2003 were exceptionally high, which affected both C input (rather low wheat crop yields) and C losses. It is recommended to repeat the measurements for another year (or several if possible) to test if our conclusions are valid over several years.

Within our study period, the measured differences in CO₂ fluxes between the tillage systems showed large temporal variations which were — to a great extent — determined by the balance between rainfall and evaporation. Therefore it can be recommended to compare

our results with continuously measured CO₂ fluxes in other climatic conditions (data in literature are lacking). In addition, N₂O emissions in this work could only be determined periodically. However, N₂O emissions show an even higher temporal variation than CO₂ emissions. It is thus highly recommended to measure these fluxes continuously in order to be able to compute reliable differences in yearly N₂O fluxes between the different tillage systems.

Concerning the modelling, it would be worthwhile to simulate also the C and N dynamics of period I and the period before tillage of period II and to test the model under different climatic conditions.

The lack of an increase in C and N mineralisation after the disruption of the upper 0-5 cm NT soil structure led us to believe that part of the organic matter could be resistant soil C, which is chemically protected against C and N mineralisation. Again, future research could test this hypothesis.

Some fluxes of the C and N cycle in bare soil were not dealt with in this work. First of all, C and N losses via N₂, NH₃ and CH₄ gas fluxes were not determined. However, during the measurement period I, NH₃ emissions might not have been negligible after the addition of ammonium-nitrate fertiliser. The N₂, NH₃ and CH₄ gases could be considered in the future to determine somewhat more complete C and N balances. Secondly, the fluxes of soluble C were not considered. For example, this soluble C can be leached from surface residues during rain. Thirdly, only net N mineralisation fluxes were measured in this work. These fluxes are the balance of gross N mineralisation and gross N immobilisation fluxes. In order to check the gross N fluxes calculated by the PASTIS model and to test the several stated hypotheses about gross N fluxes, it could be revealing to measure these gross N fluxes using ¹⁵N in the field. Moreover, in addition to the measured mineral N supply by the soil, it is also important to know the impact of a tillage system on the fate and efficiency of fertiliser N which can also be measured using ¹⁵N labelling techniques.

Our study focused on bare soil between wheat harvest and maize sowing in order to be able to quantify C and N fluxes induced by soil organic matter and crop residue decomposition without the influence of plants (e.g. plant respiration, rhizodeposition). A further step is to quantify C and N fluxes in the whole plant-soil system. In addition to these C and N fluxes in the whole plant-soil system, the global impact of the tillage systems on crop growth has to be considered. Crop growth of certain crops can be limited by soil structure (like sugarbeets and potatoes) or problems with parasitism and damaging slugs in no-till systems. The mouldboard ploughing system is preferable in soils and rotations that are prone to soil compaction. It would be a challenge to further develop the modeling system as to allow the model to simulate the effect of tillage on the C and N dynamics in the whole plant system.

Finally, the measured agronomical and environmental impacts of the different tillage systems have to be combined with socio-economic indicators for both farmers and the surrounding community. There will be no situation where a specific tillage system stands out for all aspects considered. There will be advantages and disadvantages. All depends on the priorities of the specific farmer or government of the specific region. In addition, the effect of the adoption of no-tillage largely depends on the specific soil, climatic condition and cropping system. Although our measurements were done for only one specific combination of crop rotation, soil and climate, these data are important to complete the dataset of the impact of no-tillage systems. Finally, this database of scientific knowledge has to be synthesised by defining easy-to-apply decision tools. These tools have to be diffused to the farmers and the governments in order to help them analyse their own situation. Regarding the environmental and agronomical impacts of soil tillage, this will be done in an expertise conducted by ARVALIS-Institut du végétal and INRA financed by ADEME (Agence de l'Environnement et de la Maîtrise et de l'Energie).

REFERENCES

- Abiven, S., Recous, S., submitted. Mineralization of crop residues placed at the soil surface or incorporated. *Biology and Fertility of Soils*.
- Alavoine, G., Nicolardot, B., 2002. Comparison of potentiometric titration, IR spectrophotometry and segmented micro-flow analysis to determine inorganic C in alkaline solutions. *Anal Bioanal Chem* 374, 354-358.
- Al-Kaisi, M.M., Yin, X., 2005. Tillage and crop residue effects on soil carbon and carbon dioxide emission in corn-soybean rotations. *J. Environ. Qual.* 34, 437-445.
- Alvarez, R., Alvarez, C.R., Lorenzo, G., 2001. Carbon dioxide fluxes following tillage from a mollisol in the Argentine Rolling Pampa. *Eur. J. Soil Biol.* 37, 161-166.
- Alvarez, R., Diaz, R.A., Barbero, N., Santanatoglia, O.J., Blotta, L., 1995. Soil organic carbon, microbial biomass and CO₂-C production from three tillage systems. *Soil & Tillage Research* 33, 17-28.
- Angers, D.A., N'dayegamiye, A., Côté, D., 1993. Tillage-induced differences in organic matter of particle-size fractions and microbial biomass. *Soil Science Society of America Journal* 57, 512-516.
- Angers, D.A., Recous, S., 1997. Decomposition of wheat straw and rye residues as affected by particle size. *Plant and Soil* 189, 197-203.
- Angers, D.A., Recous, S., Aita, C., 1997. Fate of carbon and nitrogen in water-stable aggregates during decomposition of ¹³C¹⁵N-labelled straw *in situ*. *European journal of Soil Science* 48, 295-300.
- Anken, T., Weisskopf, P., Zihlmann, U., Forrer, H., Jansa, J., Perhacova, K., 2004. Long-term tillage system effects under moist cool conditions in Switzerland. *Soil & Tillage Research* 78, 171-183.
- Aoyama, M., Angers, D.A., N'Dayegamiye, A., Bissonnette, N., 1999. Protected organic matter in water-stable aggregates as affected by mineral fertilizer and manure applications. *Canadian Journal of Soil Science* 79, 419-425.
- Ashman, M.R., Hallett, P.D., Brookes, P.C., 2003. Are the links between soil aggregate size class, soil organic matter and respiration rate artefacts of the fractionation procedure? *Soil Biology & Biochemistry* 35, 435-444.
- Aslam, T., Choudhary, M.A., Saggar, S., 2000. Influence of land-use management on CO₂ emissions from a silt loam soil in New Zealand. *Agriculture, Ecosystems and Environment* 77, 257-262.
- Baggs, E.M., Stevenson, M., Pihlatie, M., Regar, A., Cook, H., Cadish, G., 2003. Nitrous oxide emissions following application of residues and fertiliser under zero and conventional tillage. *Plant and Soil* 254, 361-370.

- Balabane, M., Chenu, C., Akpa, M., Arrouays, D., Barray, S., Bertrand, M., Bodet, J., Brygoo, Y., Bureau, F., Cluzeau, D., Decaens, T., Guichard, L., Hedde, M., Houot, S., Labreuche, J., Laval, K., Le Bissonnais, Y., Pawlak, B., Picard, D., Puget, P., Saulas, P., 2005. Restauration de fonctions et propriétés des sols de grande culture intensive. Effets de systèmes de culture alternatifs sur les matières organiques et la structure des sols limoneux, et approche du rôle fonctionnel de la diversité biologique des sols. Rapport final du projet Dmostra. Subvention n° 01105, Programme GESSOL n° A01494, Ministère de l'Ecologie et du Développement durable, Paris.
- Balesdent, J., 1996. The significance of organic separates to carbon dynamics and its modelling in some cultivated soils. *European Journal of soil science* 47, 485-493.
- Balesdent, J., Chenu, C., Balabane, M., 2000. Relationship of soil organic matter dynamics to physical protection and tillage. *Soil & Tillage Research* 53, 215-230.
- Balesdent, J., Mariotti, A., Boissongtier, D., 1990. Effect of tillage on soil organic carbon mineralization estimated from ^{13}C abundance in maize fields. *Journal of Soil Science* 41, 587-596.
- Ball, B.C., Scott, A., Parker, J.P., 1999. Field N_2O , CO_2 and CH_4 fluxes in relation to tillage, compaction and soil quality in Scotland. *Soil & Tillage Research* 53, 29-39.
- Beare, M.H., Bruce, R.R., 1993. A comparison of methods for measuring water-stable aggregates: implications for determining environmental effects on soil structure. *Geoderma* 56, 87-104.
- Beare, M.H., Cabrera, M.L., Hendrix, P.F., Coleman, D.C., 1994. Aggregate-protected and unprotected organic matter pools in conventional- and no-tillage soils. *Soil Science Society of America Journal* 58, 787-795.
- Benjamin, J.G., 1993. Tillage effects on near-surface soil hydraulic properties. *Soil & Tillage Research* 26, 277-288.
- Berkenkamp, A., Priesack, E., Munch, J.C., 2002. Modelling the mineralisation of plant residues on the soil surface. *Agronomie* 22, 711-722.
- Blanco-Canqui, H., Gantzer, C.J., Anderson, S.H., Alberts, E.E., 2004. Tillage and crop influences on physical properties for an Epiaqualf. *Soil Science Society of America Journal* 68, 567-576.
- Blevins, R.L., Frye, W.W., 1993. Conservation tillage: an ecological approach to soil management. *Advances in Agronomy* 51, 33-78.
- Blevins, R.L., Thomas, G.W., Smith, M.S., Frye, W.W., Cornelius, P.L., 1983. Changes in soil properties after 10 years continuous non-tilled and conventionally tilled corn. *Soil & Tillage Research* 3, 135-146.
- Boiffin, J., Monnier, G., 1991. Simplification du travail du sol et érosion hydrique. *Perspectives Agricoles* 162, 24-30.
- Boissongtier, D., 1982. Matière organique, simplification du travail du sol et irrigation. *Perspectives Agricoles* 64, 16-43.
- Bossuyt, H., Denef, K., Six, J., Frey, S.D., Merckx, R., Paustian, K., 2001. Influence of microbial populations and residue quality on aggregate stability. *Applied Soil Ecology* 16, 195-208.
- Bossuyt, H., Six, J., Hendrix, P.F., 2002. Aggregate-protected carbon in no-tillage and conventional tillage agroecosystems using carbon-14 labeled plant residue. *Soil Science Society of America Journal* 66, 1965-1973.

- Bremer, E., van Houtum, W., van Kessel, C., 1991. Carbon dioxide evolution from wheat and lentil residues as affected by grinding, added nitrogen, and the absence of soil. *Biology and Fertility of Soils* 11, 221-227.
- Brye, K.R., Norman, J.M., Gower, S.T., Bundy, L.G., 2003. Effects of management practices on annual net N-mineralization in a restored prairie and maize agroecosystems. *Biogeochemistry* 63, 135-160.
- Burgess, M.S., Mehuys, G.R., Madramootoo, C.A., 2002. Nitrogen dynamics of decomposition corn residue components under three tillage systems. *Soil Science Society of American Journal* 66, 1350-1358.
- Burns, I.G., 1974. A model for predicting the redistribution of salts applied to fallow soils after excess rainfall or evaporation. *J. Soil Sci.* 25, 165-178.
- Buyanovsky, G.A., Wagner, G.H., Gantzer, C.J., 1986. Soil respiration in a winter wheat ecosystem. *SSSAJ* 50, 338-344.
- Cabrera, M.L., Kissel, D.E., 1988. Potentially mineralizable nitrogen in disturbed and undisturbed soil samples. *Soil Science Society of America Journal* 52, 1010-1015.
- Calderon, F.J., Jackson, L.E., Scow, K.M., Rolston, D.E., 2000. Microbial responses to simulated tillage in cultivated and uncultivated soils. *Soil Biology and Biochemistry* 32, 1547-1559.
- Caron, J., Espinolda, C.R., Angers, D.A., 1996. Soil structural stability during rapid wetting: influence of land use on some aggregate properties. *Soil Science Society of America Journal* 60, 901-908.
- Carter, M.R., Rennie, D.A., 1982. Changes in soil quality under zero tillage farming systems: Distribution of microbial biomass and mineralizable C and N potentials. *Canadian Journal of Soil Science* 62, 587-597.
- Chan, K.Y., Heenan, D.P., Ashley, R., 1994. Seasonal changes in surface aggregate stability under different tillage ad crops. *Soil & Tillage Research* 28, 301-314.
- Choudhary, M.A., Akramkhanov, A., Saggar, S., 2002. Nitrous oxide emissions from a New Zealand cropped soil: tillage effects, spatial and seasonal variability. *Agriculture, Ecosystems and Environment* 93, 33-43.
- Christensen, B.T., 2001. Physical fractionation of soil and structural and functional complexity in organic matter turnover. *European journal of Soil Science* 52, 345-353.
- Collins, H.P., Elliott, E.T., Paustian, K., Bundy, L.G., Dick, W.A., Huggins, D.R., Smucker, A.J.M., Paul, E.A., 2000. Soil carbon pools and fluxes in long-term corn belt agroecosystems. *Soil Biology & Biochemistry* 32, 157-168.
- Coppens, F., 2005. Water, carbon and nitrogen dynamics in soil. Influence of crop residue location and quality. PhD thesis. Insitut National Agronomique Paris-Grignon and Katholieke Univeristeit Leuven, Paris (France) and Leuven (Belgium), 203p.
- Coppens, F., Garnier, P., Merckx, R., Recous, S., submitted. Soil moisture carbon and nitrogen dynamics following incorporation versus surface application of labelled residues in soil columns. *European journal of Soil Science*.
- Coutadeur-Desbourdes, C., 2002. Etude du transport de l'eau dans un sol labouré. Modélisation 2-D de l'infiltration et de la redistribution dans un sol à structure hétérogène. PhD thesis. Institut National Agronomique Paris-Grignon, Paris, 182p.
- Craswell, E.T., Waring, S.A., 1972. Effect of grinding on the decomposition of soil organic matter- I. The mineralization of organic nitrogen in relation to soil type. *Soil Biology & Biochemistry* 4, 427-433.

- Currie, J.A., 1984. Gas diffusion through soil crumbs: the effects of compaction and wetting. *Journal of Soil Science* 35, 1-10.
- Curtin, D., Selles, F., Wang, H., Campbell, C.A., Biederbeck, V.O., 1998. Carbon dioxide emissions and transformation of soil carbon and nitrogen during wheat straw decomposition. *Soil Science Society of America Journal* 62, 1035-1041.
- Dao, T.H., 1993. Tillage and winter wheat residue management effects on soil water infiltration and storage. *Soil Science Society of America Journal* 57, 1586-1595.
- Dao, T.H., 1998. Tillage and crop residue effects on carbon dioxide evolution and carbon storage in a Paleustoll. *Soil Science Society of America Journal* 62, 250-256.
- Datiri, B.C., Lowery, B., 1991. Effects of conservation tillage on hydraulic properties of a Griswold silt loam soil. *Soil & Tillage Research* 21, 257-271.
- De Gryze, S., Six, J., Paustian, K., Morris, S.J., Paul, E.A., Merckx, R., 2004. Soil organic carbon pool changes following land-use conversions. *Global Change Biology* 10, 1120-1132.
- De Vleeschauwer, D., Lal, R., Malafa, R., 1980. Effects of amounts of surface mulch on physical and chemical properties of an Alfisol from Nigeria. *Journal of the Science of Food and Agriculture* 31, 730-738.
- Denef, K., Six, J., Merckx, R., Paustian, K., 2004. Carbon sequestration in microaggregates of no-tillage soils with different clay mineralogy. *Soil Science Society of America Journal* 68, 1935-1944.
- Dharmakeerthi, R.S., Kay, B.D., Beauchamp, E.G., 2004. Effect of soil disturbance on N availability across a variable landscape in southern Ontario. *Soil & Tillage Research* 79, 101-112.
- Doran, J.W., 1987. Microbial biomass and mineralizable nitrogen distributions in no-tillage and plowed soils. *Biology and Fertility of Soils* 5, 68-75.
- Elliott, E.T., 1986. Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. *Soil Science Society of America Journal* 50, 627-633.
- Elliott, E.T., Coleman, D.C., 1988. Let the soil work for us. *Ecological bulletins* 39, 23-32.
- FAO, ISRIC, ISSS, 1998. World reference base for soil resources. Rome, Italy, FAO.
- Findeling, A., 2001. Etude et modélisation de certains effets du semis direct avec paillis de résidus sur les bilans hydrique, thermique et azoté d'une culture de maïs pluvial au Mexique. PhD thesis. ENGREF, Montpellier, p.
- Findeling, A., Chanzy, A., de Louvigny, N., 2003a. Modeling heat and water flows through a mulch allowing for radiative and long distance convective exchanges in the mulch. *Water Resources Research* 39, 1244.
- Findeling, A., Garnier, P., Coppens, F., Lafolie, F., Recous, S., submitted. Modeling water, carbon and nitrogen dynamics in soil covered with decomposing mulch. *European Journal of Soil Science*.
- Fortin, M.-C., Rochette, P., Pattey, E., 1996. Soil Carbon Dioxide Fluxes from Conventional and No-Tillage Small-Grain Cropping Systems. *Soil Science Society of America Journal* 60, 1541-1547.
- Foy, N., 2003. Effet des systèmes de culture sur l'évolution de la structure d'un sol limoneux. PhD thesis. Ecole Nationale d'Ingénieur des Travaux Agricoles de Bordeaux, 62p.
- Franzluebbers, A.J., 1999. Potential C and N mineralization and microbial biomass from intact and increasingly disturbed soils of varying texture. *Soil Biology & Biochemistry* 31, 1083-1090.

- Franzluebbers, A.J., Arshad, M.A., 1997a. Particulate organic carbon content and potential mineralization as affected by tillage and texture. *Soil Science Society of America Journal* 61, 1382-1386.
- Franzluebbers, A.J., Arshad, M.A., 1997b. Soil microbial biomass and mineralizable carbon of water-stable aggregates. *Soil Science Society of America Journal* 61, 1090-1097.
- Franzluebbers, A.J., Hons, F.M., Zuberer, D.A., 1995. Tillage and crop effects on seasonal soil carbon and nitrogen dynamics. *Soil Science Society of America Journal* 59, 1618-1624.
- Frey, S.D., Elliott, E.T., Paustian, K., 1999. Bacterial and fungal abundance and biomass in conventional and no-tillage agroecosystems along two climatic gradients. *Soil Biology and Biochemistry* 31, 573-585.
- Frey, S.D., Six, J., Elliott, E.T., 2003. Reciprocal transfer of carbon and nitrogen by decomposer fungi at the soil-litter interface. *Soil Biol. Biochem.* 35, 1001-1004.
- Fuentes, J.P., Flury, M., Bezdicek, D.F., 2004. Hydraulic properties in a silt loam soil under natural prairie, conventional till, and no-till. *Soil Science Society of America Journal* 68, 1679-1688.
- Fuentes, J.P., Flury, M., Huggins, D.R., Bezdicek, D.F., 2003. Soil water and nitrogen dynamics in dryland cropping systems of Washington State, USA. *Soil & Tillage Research* 71, 33-47.
- Gaillard, V., Chenu, C., Recous, S., Richard, G., 1999. Carbon, nitrogen and microbial gradients induced by plant residues decomposing in soil. *European Journal of Soil Science* 50, 567-578.
- Garnier, P., Ezzine, N., De Gryze, S., Richard, G., 2004. Hydraulic properties of soil-straw mixtures. *Vadose Zone Journal* 3, 714-721.
- Garnier, P., Néel, C., Aita, C., Recous, S., Lafolie, F., Mary, B., 2003. Modelling carbon and nitrogen dynamics in a bare soil with and without straw incorporation. *European Journal of Soil Science* 54, 555-568.
- Garnier, P., Néel, C., Mary, B., Lafolie, F., 2001. Evaluation of a nitrogen transport and transformation model in a bare soil. *European Journal of Soil Science* 52, 253-268.
- Germon, J.C., Taureau, J.C., Thomas, J.M., 1994. Effets des méthodes simplifiées de travail du sol sur les transformations de l'azote et leurs conséquences sur le lessivage des nitrates. In: INRA (Ed.), *Simplification du travail du sol*. Paris (France), pp. 125-154.
- Gregorich, E.G., Kachanoski, R.G., Voroney, R.P., 1989. Carbon mineralization in soil size fractions after various amounts of aggregate disruption. *Journal of Soil Science* 40, 649-659.
- Groffman, P.M., 1984. Nitrification and denitrification in conventional and no-tillage soils. *Soil Science Society of America Journal* 49, 329-334.
- Guérif, J., Richard, G., Dürr, C., Machet, J.M., Recous, S., Roger-Estrade, J., 2001. A review of tillage effects on crop residue management, seedbed conditions and seedling establishment. *Soil & Tillage Research* 61, 13-32.
- Gupta, S.C., Dowdy, R.H., Larson, W.E., 1977. Hydraulic and thermal properties of a sandy soil as influenced by incorporation of sewage sludge. *Soil Science Society of America Journal* 41, 601-605.
- Gupta, V.V.S.R., Germida, J.J., 1988. Distribution of microbial biomass and its activity in different soil aggregate size classes as affected by cultivation. *Soil Biology and Biochemistry* 20, 777-786.

- Hassink, J., 1992. Effects of soil texture and structure on carbon and nitrogen mineralization in grassland soils. *Biology and Fertility of Soils* 14, 126-134.
- Haynes, R.J., 1993. Effect of sample pretreatment on aggregate stability measured by wet sieving or turbidimetry on soils of different cropping history. *Journal of Soil Science* 44, 261-270.
- Hénault, C., Bizouard, F., Laville, P., Babrielle, B., Nicoullaud, B., Germon, J.C., Cellier, P., 2005. Predicting in situ soil N₂O emission using NOE algorithm and soil database. *Global Change Biology* 11, 115-127.
- Hendrix, P.F., Han, C.-R., Groffman, P.M., 1988. Soil respiration in conventional and no-tillage agroecosystems under different winter cover crop rotations. *Soil & Tillage Research* 12, 135-148.
- Henriksen, T.M., Breland, T.A., 2002. Carbon mineralization, fungal and bacterial growth, and enzyme activities as affected by contact between crop residues and soil. *Biology and Fertility of Soils* 35, 41-48.
- Holland, E.A., Coleman, D.C., 1987. Litter placement effects on microbial and organic matter dynamics in an agroecosystem. *Ecology* 68, 425-433.
- Houot, S., Mordelet, P., Tardieu, F., Molina, J., 1990. Effets du tassement et de la teneur en eau du sol sur la biomasse microbienne et la libération d'azote minéral. In: Calvet, R. (Ed.), *Nitrates, agriculture, eau*. Institut National de la Recherche Agronomique, Paris, pp. 201-207.
- Hubbard, R.K., Hargrove, W.L., Lowrance, R.R., Williams, R.G., Mullinix, B.G., 1994. Physical properties of a clayey coastal plain soil as affected by tillage. *Journal of Soil and Water Conservation* 49, 276-283.
- IPPC, 2001. *Climate Changes: The Scientific Basis Contribution*. Cambridge, Cambridge University Press.
- Jastrow, J.D., 1996. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. *Soil Biology & Biochemistry* 28, 665-676.
- Kabat, P., Beekma, J., 1994. Water in the unsaturated zone. In: Ritzema, H.P. (Ed.), *Drainage Principles and Applications*. International Institute for Land Reclamation and Improvement (ILRI), Wageningen, the Netherlands, pp. 383-434.
- Kamphake, L.J., Hannah, S.A., Coehn, J.M., 1967. Automated analysis for nitrate by hydrazine reduction. *Water Research* 1, 205-216.
- Kandeler, E., Tschierko, D., Spiegel, H., 1999. Long-term monitoring of microbial biomass, N mineralisation and enzyme activities of a Chernozem under different tillage management. *Biology and Fertility of Soils* 28, 343-351.
- Kessavalou, A., Doran, J.W., Mosier, A.R., R.A., D., 1998a. Greenhouse Gas Fluxes following Tillage and Wetting in a Wheat-Fallow Cropping System. *J. Environ. Qual.* 27, 1105-1116.
- Kessavalou, A., Mosier, A.R., Doran, J.W., Drijber, R.A., Lyon, D.W., Heinemeyer, O., 1998b. Fluxes of Carbon Dioxide, Nitrous Oxide, and Methane in Grass Sod and Winter Wheat-Fallow Tillage Management. *J. Environ. Qual.* 27, 1094-1104.
- Kieft, T.L., Soroker, E., Firestone, M.K., 1987. Microbial biomass response to a rapid increase in water potential when dry soil is wetted. *Soil Biology and Biochemistry* 19, 119-126.
- Kitur, B.K., Olson, K.R., Siemens, J.C., Philips, S.R., 1993. Tillage effects on selected physical properties of Grantsburg silt loam. *Communications in Soil Science and Plant Analysis* 24, 1509-1527.

- Kristensen, H.L., McCarty, G.W., Meisinger, J.J., 2000. Effects of soil structure disturbance on mineralization of organic soil nitrogen. *Soil Science Society of America Journal* 64, 371-378.
- Krom, M.D., 1980. Spectrophotometric determination of ammonia: a study of a modified Berthelot reaction using salicylate and dichlorocyanurate. *The analyst* 105, 305-316.
- Kushwaha, C.P., Singh, K.P., 2005. Crop productivity and soil fertility in a tropical dryland agro-ecosystem: impact of residue and tillage management. *Experimental Agriculture* 41, 39-50.
- Labreuche, J., Couture, D., Bodet, J.-M., 2003. Essai travail du sol de longue durée de Boigneville. *Perspectives Agricoles* 286, 56-58.
- Ladd, J.N., Foster, R.C., Skjemstad, J.O., 1993. Soil structure: carbon and nitrogen metabolism. *Geoderma* 56, 401-434.
- Lafolie, F., 1991. Modelling water flow, nitrogen transport and root uptake including physical non-equilibrium and optimization of the root water potential. *Fertilizer Research* 27, 215-232.
- Lal, R., 1999. Soil compaction and tillage effects on soil physical properties of a Mollic Ochraqualf in northwest Ohio. *Journal of Sustainable Agriculture* 14, 53-65.
- Langlet, B., Rémy, J.C., 1976. Incidence de la simplification du travail du sol sur la dynamique de l'azote. In: ITCF (Ed.), *Simplification du travail du sol en production céréalière*. Paris, pp. 189-204.
- Li, D., Velde, B., Zhang, T., 2004. Observations of pores and aggregates during aggregation in some clay-rich agricultural soils as seen in 2D image analysis. *Geoderma* 118, 191-207.
- Linères, M., Djakovitch, J.L., 1993. Caractérisation de la stabilité biologique des apports organiques par l'analyse biochimique. In: Decroux, J. and Ignazi, J.C. (Ed.), *Matières organiques et agricultures. Quatrième journées de l'analyse de terre et cinquième forum de la fertilisation raisonnée*. GEMAS-COMIFER, Paris, pp. 159-168.
- MacKenzie, A.F., Fan, M.X., Cadrin, F., 1998. Nitrous oxide Emission in Three Years as Affected by Tillage, Corn-Soybean-Alfalfa Rotations, and Nitrogen Fertilization. *J. Environ. Qual.* 27, 698-703.
- Manichon, H., 1982. Influence des systèmes de culture sur le profil cultural: élaboration d'une méthode de diagnostic basée sur l'observation morphologique. PhD thesis. Institut National Agronomique Paris-Grignon, Paris, p.
- Martens, D.A., 2001. Nitrogen cycling under different soil management systems. *Advances in Agronomy* 70, 143-192.
- Mary, B., Beaudoin, N., Justes, E., Machet, J.-M., 1999. Calculation of nitrogen mineralization and leaching in fallow soil using a simple dynamic model. *European Journal of Soil Science* 50, 549-566.
- Mary, B., Guérif, J., 1994. Intérêts et limites des modèles de prévision de l'évolution des matières organiques et de l'azote dans le sol. *Cahiers Agricultures* 3, 247-257.
- Mary, B., Recous, S., Darwis, D., Robin, D., 1996. Interactions between decomposition of plant residues and nitrogen cycling in soil. *Plant and Soil* 181, 71-82.
- Meek, B.D., Carter, D.L., Westermann, D.T., Peckenpaugh, R.E., 1994. Root-Zone Mineral Nitrogen Changes as Affected by Crop Sequence and Tillage. *Soil Science Society of America Journal* 58, 1464-1469.
- Mikha, M.M., Rice, C.W., 2004. Tillage and manure effects on soil and aggregate-associated carbon and nitrogen. *Soil Science Society of America Journal* 68, 809-816.

- Mrabet, R., Saber, N., El-Brahli, A., Lahlou, S., Bessam, F., 2001. Total, particulate organic matter and structural stability of a Calcixeroll soil under different wheat rotations and tillage systems in a semiarid area of Morocco. *Soil & Tillage Research* 57, 225-235.
- Papini, R., Gamba, C., Piovaneli, C., Brandi, G., 2002. Impact of tillage practices on seasonal nitrate dynamics, leaching and microbial activity in a crop rotation. In: Violante, A., Huang, P.M., Bollag, J.M. and Gianfreda, L. (Ed.), *Developments in soil science*. Elsevier Science, pp. 275-287.
- Pekrun, C., Kaul, H.P., Claupein, W., 2003. Soil tillage for sustainable nutrient management. In: El Titi, A. (Ed.), *Soil tillage in agrosystems*. CRC Press, Boca Raton, pp. 83-113.
- Powlson, D.S., 1980. The effects of grinding on microbial and non-microbial organic matter in soil. *Journal of Soil Science* 31, 77-85.
- Probert, M.E., Dimes, J.P., Keating, B.A., Dalal, R.C., Strong, W.M., 1998. APSIM's water and nitrogen modules and simulation of the dynamics of water and nitrogen in fallow systems. *Agricultural Systems* 56, 1-28.
- Puget, P., Chenu, C., Balesdent, J., 1995. Total and young organic matter distributions in aggregates of silty cultivated soils. *European journal of Soil Science* 46, 449-459.
- Pulleman, M.M., Marinissen, J.C.Y., 2004. Physical protection of mineralizable C in aggregates from long-term pasture and arable soil. *Geoderma* 120, 273-282.
- Recous, S., 1995. Réponse des matières organiques des sols aux changements atmosphériques globaux. II - Effet de la température sur la minéralisation d'un résidu végétal (maïs) et de la matière organique des sols. *Ecosystèmes et Changements Globaux: Les Dossiers de l'Environnement de l'INRA* 8, 81-85.
- Reicosky, D.C., Lindstrom, M.J., 1993. Fall tillage method: effect on short-term carbon dioxide flux from soil. *Agron. J.* 85, 1237-1243.
- Reicosky, D.C., Voorhees, W.B., Radke, J.K., 1981. Unsaturated water flow through a simulated wheel track. *Soil Science Society of America Journal* 45, 3-8.
- Rice, C.W., Smith, M.S., Blevins, R.L., 1986. Soil nitrogen availability after long-term continuous no-tillage and conventional tillage corn production. *Soil Science Society of America Journal* 50, 1206-1210.
- Robert, M., Capillon, A., Raunet, M., 2004. Les techniques culturales sans labour: historique et enjeux. *Proceedings of the Techniques culturales sans labour. Impacts économiques et environnementaux congress*, Paris, Comité d'Orientation pour des Pratiques agricoles respectueuses de l'Environnement.
- Rodrigo, A., Recous, S., Neel, C., Mary, B., 1997. Modelling temperature and moisture effects on C-N transformations in soils: comparison of nine models. *Ecological Modelling* 102, 325-339.
- Roger-Estrade, J., Richard, G., Caneill, J., Boizard, H., Coquet, Y., Defossez, P., Manichon, H., 2004. Morphological characterisation of soil structure in tilled fields: from a diagnosis method to the modelling of structural changes over time. *Soil & Tillage Research* 79, 33-49.
- Roger-Estrade, J., Richard, G., Manichon, H., 2000. A compartmental model to simulate temporal changes in soil structure under two cropping systems with annual mouldboard ploughing in a silt loam. *Soil & Tillage Research* 54, 41-53.
- Roth, C.H., Meyer, B., Frede, H.-G., Derpsch, R., 1988. Effect of mulch rates and tillage systems on infiltrability and other soil physical properties of an Oxisol in Parana, Brazil. *Soil & Tillage Research* 11, 81-91.

- Sainju, U.M., Singh, B.P., Yaffa, S., 2002. Soil organic matter and tomato yield following tillage, covercropping, and nitrogen fertilization. *Agronomy Journal* 94, 594-602.
- Salazar-Sosa, E., Leos-Rodriguez, J.A., Fortis-Hernandez, M., Vasquez-Vasquez, C., 2002. Nitrogen recovery and uptake by wheat and sorghum in stubble mulch and no tillage systems. *Agrociencia (Monticello)* 36, 433-440.
- SAS Institute, 2001. SAS for windows Version 8.02. SAS Institute Inc., Cary, NC, USA.
- Sebillotte, M., Meynard, J.-M., 2004. Enjeux agronomiques, économiques et environnementaux des techniques culturales sans labour. Proceedings of the Techniques culturales sans labour. Impacts économiques et environnementaux congress, Paris, Comité d'Orientation pour des Pratiques agricoles respectueuses de l'ENvironnement.
- Shipitalo, M.J., Dick, W.A., Edwards, W.M., 2000. Conservation tillage and macropore factors that affect water movement and the fate of chemicals. *Soil & Tillage Research* 53, 167-183.
- Shukla, M.K., 2003. Tillage effects on physical and hydrological properties of a Typic Argiaquoll in Central Ohio. *Soil Science* 168, 802-811.
- Six, J., Bossuyt, H., Degryze, S., Denef, K., 2004. A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. *Soil & Tillage Research* 79, 7-31.
- Six, J., Elliott, E.T., Paustian, K., 1999a. Aggregate and soil organic matter dynamics under conventional and no-tillage systems. *Soil Science Society of America Journal* 63, 1350-1358.
- Six, J., Elliott, E.T., Paustian, K., 2000a. Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. *Soil Biology & Biochemistry* 32, 2099-2103.
- Six, J., Elliott, E.T., Paustian, K., 2000b. Soil structure and soil organic matter: II. A normalized stability index and the effect of mineralogy. *Soil Science Society of America Journal* 64, 1042-1049.
- Six, J., Elliott, E.T., Paustian, K., Doran, J.W., 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Science Society of America Journal* 62, 1367-1377.
- Six, J., Feller, C., Denef, K., Ogle, S.M., Sa, J.C.M., Albrecht, A., 2002. Soil organic matter, biota and aggregation in temperate and tropical soils - Effects of no-tillage. *Agronomie* 22, 755-775.
- Six, J., Paustian, K., Elliott, E.T., Combrink, C., 2000c. Soil structure and organic matter: I. Distribution of aggregate-size classes and aggregate-associated carbon. *Soil Science Society of America Journal* 64, 681-689.
- Six, J., Schultz, P.A., Jastrow, J.D., Merckx, R., 1999b. Recycling of sodium polytungstate used in soil organic matter studies. *Soil Biology & Biochemistry* 31, 1193-1196.
- Smith, J.U., Smith, P., Addiscott, T.M., 1996. Quantitative methods to evaluate and compare organic matter models. In: Poulson, D.S., Smith, P. and Smith, J. (Ed.), Evaluation of soil organic matter models. Springer-Verlag, Berlin, pp. 182-199.
- Soon, Y.K., Clayton, G.W., Rice, W.A., 2001. Tillage and previous crop effects on dynamics of nitrogen in a wheat-soil system. *Agronomy Journal* 93, 842-849.
- Sorensen, L.H., 1983. The influence of stress treatments on the microbial biomass and the rate of decomposition of humified matter in soils containing different amounts of clay. *Plant and Soil* 75, 107-119.

- Staricka, J.A., Allmaras, R.R., Nelson, W.V., 1991. Spatial variation of crop residue incorporated by tillage. *Soil Science Society of America Journal* 55, 1668-1674.
- Steinbach, H.S., Alvarez, R., Valente, C.R., 2004. Balance between mineralization and immobilization of nitrogen as affected by soil mineral nitrogen level. *Agrochimica* 68, 204-212.
- Steiner, J.L., Schomberg, H.H., Unger, P.W., Cresap, J., 2000. Biomass and residue cover relationships of fresh and decomposing small grain residue. *Soil Science Society of America Journal* 64, 2109-2114.
- Stockfisch, N., Forstreuter, T., Ehlers, W., 1999. Ploughing effects on soil organic matter after twenty years of conservation tillage in Lower Saxony, Germany. *Soil & Tillage Research* 52, 91-101.
- Strong, D.T., De Wever, H., Merckx, R., Recous, S., 2004. Spatial location of carbon decomposition in the soil pore system. *European Journal of Soil Science* 55, 739-750.
- Tamari, S., Bruckler, L., Halbertsma, J., Chadoeuf, J., 1993. A simple method for determining soil hydraulic properties in the laboratory. *Soil Science Society of America Journal* 57, 642-651.
- Tebrügge, F., Düring, R.-A., 1999. Reducing tillage intensity - a review of results from a long-term study in Germany. *Soil & Tillage Research* 53, 15-28.
- Thorburn, P.J., Probert, M.E., Robertson, F.A., 2001. Modelling decomposition of sugar cane surface residues with APSIM-Residue. *Field Crops Research* 70, 223-232.
- Trinsoutrot, I., Nicolardot, B., Justes, E., Recous, S., 2000a. Decomposition in the field of residues of oilseed rape grown at two levels of nitrogen fertilisation. Effects on the dynamics of soil mineral nitrogen between successive crops. *Nutrient Cycling in Agroecosystems* 56, 125-137.
- Trinsoutrot, I., Recous, S., Bentz, B., Linères, M., Chèneby, D., Nicolardot, B., 2000b. Biochemical quality of crop residues and carbon and nitrogen mineralization kinetics under nonlimiting nitrogen conditions. *Soil Science Society of America Journal* 64, 918-926.
- Trocherie, F., Rabaud, V., 2004. Le développement des techniques sans labour (enquête 'pratiques culturales'). *Proceedings of the Techniques culturales sans labour. Impacts économiques et environnementaux congress, Paris, Comité d'Orientation pour des Pratiques agricoles respectueuses de l'Environnement*.
- Van Genuchten, M.T., 1980. A closed-form equation for predicting the hydraulic conductivity of unsaturated soils. *Soil Science Society of America Journal* 44, 892-898.
- Van Gestel, M., Merckx, R., Vlassak, K., 1993. Microbial biomass responses to soil drying and rewetting: the fate of fast- and slow-growing microorganisms in soils from different climates. *Soil Biology & Biochemistry* 25, 109-123.
- Vanlauwe, B., Vanlangenhove, G., Merckx, R., Vlassak, K., 1995. Impact of rainfall regime on the decomposition of leaf litter with contrasting quality under subhumid tropical conditions. *Biology and Fertility of Soils* 20, 8-16.
- Vestergaard, P., Ronn, R., Christensen, S., 2001. Reduced particle size of plant material does not stimulate decomposition, but affects the microbivorous microfauna. *Soil Biology & Biochemistry* 33, 1805-1810.
- Vinten, A.J.A., Ball, B.C., O'Sullivan, M.F., Henshall, J.K., 2002. The effects of cultivation method, fertilizer input and previous sward type on organic C and N storage and gaseous losses under spring and winter barley following long-term leys. *Journal of Agricultural Science* 139, 231-243.

- Wander, M.M., Bidart, M.G., 2000. Tillage practice influences on the physical protection, bioavailability and composition of particulate organic matter. *Biology and Fertility of Soils* 32, 360-367.
- Wander, M.M., Bidart, M.G., Aref, S., 1998. Tillage impacts on depth distribution of total and particulate organic matter in three Illinois soils. *Soil Science Society of America Journal* 62, 1704-1711.
- Wander, M.M., Yang, X., 2000. Influence of tillage on the dynamics of loose- and occluded-particulate and humified organic matter fractions. *Soil Biology & Biochemistry* 32, 1151-1160.
- Wu, L., Swan, J.B., Paulson, W.H., Randall, G.W., 1992. Tillage effects on measured soil hydraulic properties. *Soil & Tillage Research* 25, 17-33.
- Yang, X.M., Kay, B.D., 2001. Impacts of tillage practices on total, loose- and occluded-particulate, and humified organic carbon fractions in soils within a field in southern Ontario. *Canadian Journal of Soil Science* 81, 149-156.
- Yang, X.M., Wander, M.M., 1999. Tillage effects on soil organic carbon distribution and storage in a silt loam soil in Illinois. *Soil & Tillage Research* 52, 1-9.
- Zihlmann, U., Weisskopf, P., Bohren, C., Dubois, D., 2002. Soil nitrogen dynamics in different maize cropping systems. *Agrarforschung* 9, 392-397.
- Zihlmann, U., Weisskopf, P., Müller, M., Schafflützel, R., Chervet, A., Sturny, W.G., 2001. Effect of no-tillage on nutrient content and organic matter of the soil. *Agrarforschung* 8, 18-22.

LIST OF PUBLICATIONS

International journals with peer review:

- Oorts, K., Bossuyt, H., Labreuche, J., Merckx, R., Nicolardot, B. Carbon and nitrogen stocks in relation to organic matter fractions, aggregation and pore size distribution in no-tillage and conventional tillage in Northern France. Submitted to European Journal of Soil Science.
- Oorts, K., Nicolardot, B., Merckx, R., Richard, G., Boizard, H. C and N mineralisation of undisturbed and disrupted soil from different structural zones of conventional tillage and no-tillage systems in Northern France. Submitted to Soil Biology & Biochemistry.
- Oorts, K., Nicolardot, B., Thiébeau, P., Merckx, R., Laurent, F. Long-term effect of tillage systems on soil organic nitrogen mineralisation under field conditions. In preparation.
- Oorts, K., Merckx, R., Gréhan, E., Labreuche, J., Nicolardot, B. Determinants of CO₂ and N₂O fluxes in long-term no-tillage and conventional tillage systems in Northern France. Soil & Tillage Research. In preparation.
- Oorts, K., Garnier, P., Findeling, A., Mary, B., Richard, G., Nicolardot, B. Modelling carbon and nitrogen dynamics in no-tillage and conventional tillage systems in Northern France. In preparation.

Conference proceedings

- Laurent, F., Oorts, K., Nicolardot, B., Thiébeau, P., 2004. Effects of tillage systems on soil organic nitrogen mineralization under field conditions. Abstracts of the Eurosoil congress, 4-12 September, Albert-Ludwigs-Universität Freiburg (Germany), pp. 65.
- Nicolardot, B., Oorts, K., Richard, G., Boizard, H., Merckx, R., Gréhan, E., Labreuche, J., 2004. In situ greenhouse gas emissions (CO₂ and N₂O) from long-term no-till and conventional systems. Abstracts of the Eurosoil congress, 4-12 September, Albert-Ludwigs-Universität Freiburg (Germany), pp. 346.
- Oorts, K., Nicolardot, B., Merckx, R., Richard, G., Boizard, H., Labreuche, J., 2004. Effects of soil structure and organic matter protection induced by tillage systems on C and N mineralization measured during soil incubations. Abstracts of the Eurosoil congress, 4-12 September, Albert-Ludwigs-Universität Freiburg (Germany), pp. 417.

APPENDIX

Appendix A: Structural soil profiles

For the different tillage plots studied in this thesis, soil profiles were characterised by the method formalised by Manichon (1982) and described by Roger-Estrade *et al.* (2004) in order to study the effect of the tillage system on the soil structure and the localization of crop residues. The three long-term tillage systems considered were mouldboard ploughing to 20 cm depth (CT), superficial tillage to 10 cm depth (ST) and no-tillage (NT). Table A.1 summarises the timing, the tillage treatments and the plot numbers of the characterised soil profiles.

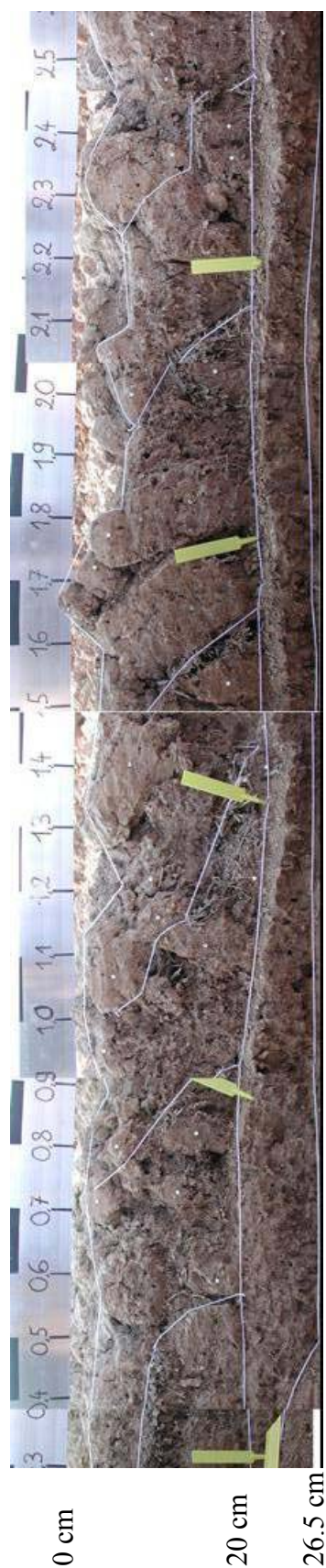
Table A1: Timing, plot numbers and tillage treatments of the characterised soil profiles.

Date	Field plots	Study period	Months after wheat harvest	Months after tillage for CT and ST
March 2003	212 (CT), 214 (ST) and 215 (NT)	I	7	4
October 2003	221 (ST), 223 (NT) and 224 (CT)	II _a	2	12
April 2004	223 (NT) and 224 (CT)	II _b	8	5

Soil profiles were situated perpendicular to the direction of field operations and had a similar width as the management tools. Soil zones characterized by different soil structures were separated. In addition, zones with crop residues and biological activity were identified.

Soil structure was either massive or fragmentary. A fragmentary structure is characterised by the presence of distinguishable soil clods and/or fine soil with eventually voids between the soil clods. The fine soil has soil structures smaller than 2 cm. The soil clods > 2 cm are classified in three categories based on their internal structural porosity. (1) Δ Clods do not have eye-visible structural porosity and present smooth rupture faces. These clods are the result of a severe compaction of anthropologic origin. (2) Γ Clods have clearly eye-visible structural porosity. (3) Φ Clods originate from Δ clods but have cracks due to the action of the climate (a typical effect of freeze/thaw and wet/dry cycles). Image analysis allowed estimating the percentage of Δ and Φ clods of the total area of the plough layer in the CT plots.

Structural profile of the conventional tillage plot in March 2003 (plot 212)



Soil layer	Depth	Description
Recent plough layer	0-20 cm	Fragmentary structure with Δ clods (28%), Φ clods (18%) and Γ clods, fine soil and a large amount of voids. Presence of biological activity. Crop residues are located in the furrows (with on average a distance of 40 cm between the furrows).
Ancient plough layer	20-26.5 cm	Very dense massive Δ structure.

Structural profile of the no-tillage plot in March 2003 (plot 215)



0 cm

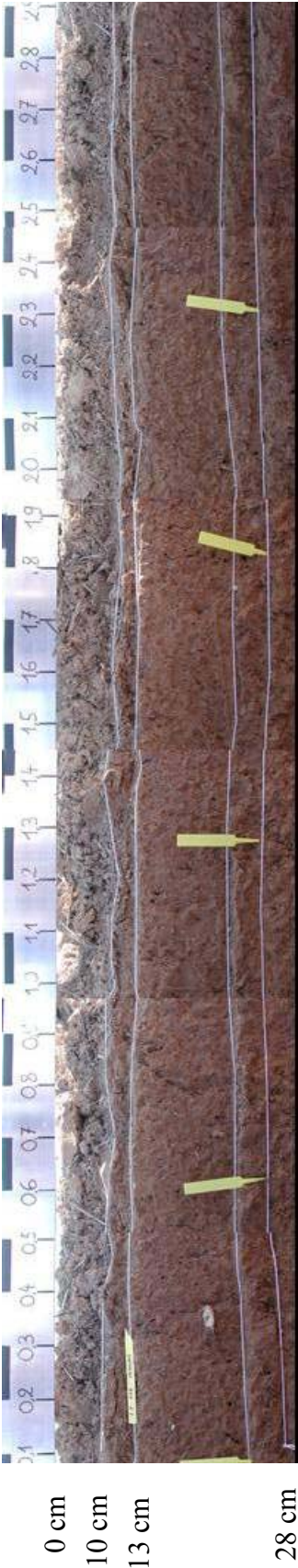
5 cm

15 cm

29 cm

Soil layer	Depth	Description
Mulch layer	Above 0 cm	Surface layer of crop residues.
Seedbed	0-5 cm	Fragmentary structure with fissures and very low cohesion between the soil clods and an important presence of crop residues, soil organic matter, roots and biological activity.
Cracked ancient plough layer	5-15 cm	Massive Φ structure. A large presence of lamellar fissures. Presence of biological activity.
Ancient plough layer	15-29 cm	Massive Δ structure with few Φ . The little structural porosity is localised and related to biological activity (earthworms). The rare presence of fissures is localised.

Structural profile of the superficial tillage plot in March 2003 (plot 214)



Soil layer	Depth	Description
Recent plough layer	0-10 cm	Fragmentary structure. Presence of small clods and a small amount of fine soil.
Cracked ancient plough layer	10-13 cm	Massive Φ structure. A large presence of lamellar fissures. Very low structural porosity.
Ancient plough layer	13-28 cm	Massive Δ structure. The small amount of structural porosity is localised and related to biological activity (earthworms).

Structural profile of the conventional tillage plot in October 2003 (plot 224)



Soil layer	Depth	Description
Mulch layer	Above 0 cm	Surface layer of crop residues.
Recent plough layer	0-3 cm	Fine fragmentary structure with dominance of Γ clods
Recent plough layer	3-11 cm	Fragmentary structure with dominance of Γ clods and fine soil but with some Δ and Φ clods.
Recent plough layer	11-22 cm	Coarse fragmentary structure with Γ , Δ and Φ clods. Presence of fissures and earthworms (especially in Δ clods).
Ancient plough layer	22-27 cm	Massive Δ structure (a small number of zones with Γ clods). Presence of earthworms.

The whole recent plough layer contains 14% Δ and 9% Φ clods. Visible incorporated crop residues were rare in the recent plough layer after 12 months of incorporation of surface residues by tillage.

Structural profile of the no-tillage plot in October 2003 (plot 223)



Soil layer	Depth	Description
Mulch layer	Above 0 cm	Surface layer of crop residues.
Seedbed	0-6 cm	Fragmentary structure with fissures and very low cohesion between the mainly Γ clods and very high porosity. There is an important presence of crop residues, soil organic matter, roots and biological activity.
Cracked ancient plough layer	6-10 cm	Massive Φ structure. A large presence of lamellar fissures.
Ancient plough layer	10-25 cm	Massive Φ structure.

Structural profile of the superficial tillage plot in October 2003 (plot 221)



Soil layer	Depth	Description
Mulch layer	Above 0 cm	Surface layer of crop residues.
Recent plough layer	0-3 cm	Highly fragmentary structure.
Recent plough layer	3-6 cm	Fragmentary structure with Γ clods. Presence of lamellar zones.
Cracked ancient plough layer	6-10 cm	Massive Φ structure. A large presence of lamellar fissures.
Ancient plough layer	10-21 cm	Massive Δ structure with 30% Φ . Large presence of earthworms. Some zones with higher porosity.

Structural profile of the conventional tillage plot in April 2004 (plot 224)

Soil layer	Depth	Description
Recent plough layer	0-23 cm	Fragmentary structure with Δ clods (3%), Φ clods (18/%) and Γ clods, fine soil and voids.
Ancient plough layer	24-31 cm	Massive Δ and Φ structure. Presence of a plough pan.

Structural profile of the no-tillage plot in April 2004 (plot 223)



Soil layer	Depth	Description
Mulch layer	Above 0 cm	Surface layer of crop residues.
Seedbed	0-7 cm	Massive Δ and Φ structure. A large presence of lamellar fissures.
Ancient plough layer	8-25 cm	Massive Φ structure. Large presence of earthworms.

Appendix B: Illustrations

Illustration A: Gas chambers for the measurement of CO₂



Illustration B: Gas chambers for the measurement of N₂O



Illustration C: Photographs of the soil covered by crop residues in the three studied tillage systems during period II.

Conventional tillage



No-tillage



Superficial tillage



July 29th
Start period II

November 19th
Day before
tillage

April 13th
End period II

Appendix C: Description of the PASTIS model¹

The one-dimensional mechanistic model PASTIS (Prediction of Agricultural Solute Transfer In Soils), described by Lafolie (1991), Garnier *et al.* (2001) and Garnier *et al.* (2003) simulates the transport of water, solutes and heat using the Richard's equation for water flow (1), the convection-dispersion for solute transport (2) and the convection-diffusion equation for heat flow (3):

$$C_h \frac{\partial h}{\partial t} = \frac{\partial}{\partial z} \left(K \left(\frac{\partial h}{\partial z} - 1 \right) \right) \quad (1)$$

$$\frac{\partial(\theta C)}{\partial t} = \frac{\partial}{\partial z} \left(\theta D \frac{\partial C}{\partial z} - qC \right) \quad (2)$$

$$C_T \frac{\partial T}{\partial t} = \frac{\partial}{\partial z} \left(\lambda_T \frac{\partial T}{\partial z} - qC_w T \right) \quad (3)$$

where $C_h = \left(\frac{\partial \theta}{\partial h} \right)_T$ is the soil capillary capacity (m^{-1}), h is the pressure head of soil water

(m), t is the time (s), z is the vertical coordinate (m), K is the soil hydraulic conductivity (m s^{-1}), C is the solute concentration (kg m^{-3}), θ is the soil volumetric water content ($\text{m}^3 \text{m}^{-3}$), D is the dispersion coefficient ($\text{m}^2 \text{s}^{-1}$), q is the Darcy water flux (m s^{-1}), C_T is the soil volumetric thermal capacity ($\text{J m}^{-3} \text{K}^{-1}$), T is the soil temperature (K), λ_T is the soil thermal conductivity ($\text{W m}^{-1} \text{K}^{-1}$) and C_w is the water volumetric thermal capacity ($\text{J m}^{-3} \text{K}^{-1}$). The model does not consider vapour phase transport, hysteresis of hydraulic properties or preferential flow. Boundary conditions can be applied at the top and bottom of the soil for water, solutes and temperature.

¹ Adapted from: Findeling, A., Garnier, P., Coppens, F., Lafolie, F., Recous, S., submitted. Modeling water, carbon and nitrogen dynamics in soil covered with decomposing mulch. European Journal of Soil Science.

CANTIS submodel

The CANTIS submodel (Carbon And Nitrogen Transformations In Soil) simulates the transformations of carbon and nitrogen (Figure A.1 and Table A.2). It considers the decomposition of organic matter, mineralisation, immobilisation, nitrification and humification. The organic matter is divided into three non-living organic pools: fresh, soluble and humified organic matter (FOM, SOL, HOM, respectively) and two living pools. The microbial population is split into an autochthonous biomass (AUB) that decomposes humified organic matter and a zymogenous biomass (ZYB) that decomposes fresh and soluble organic matter. Crop residues are added to the fresh and soluble organic matter pool. The FOM pool consists of rapidly decomposable material (RDM), hemicellulose (HCE), cellulose (CEL) and lignin (LIG). Decomposition of fresh or soluble organic matter is assumed to follow first-order kinetics as:

$$\frac{dC_i}{dt} = -k_i C_i f_T f_W f_N f_B \quad (4)$$

where C_i is the carbon content of the organic matter pool i , k_i is the decomposition rate constant of that pool, f_T is the temperature function, f_W is the moisture function, f_N is the nitrogen limitation function, and f_B is the biomass-dependent function. These functions have been described previously by Garnier *et al.* (2001). A sink/source term is introduced in Eq. (2) to account for mineralisation or immobilisation of N by micro-organisms.

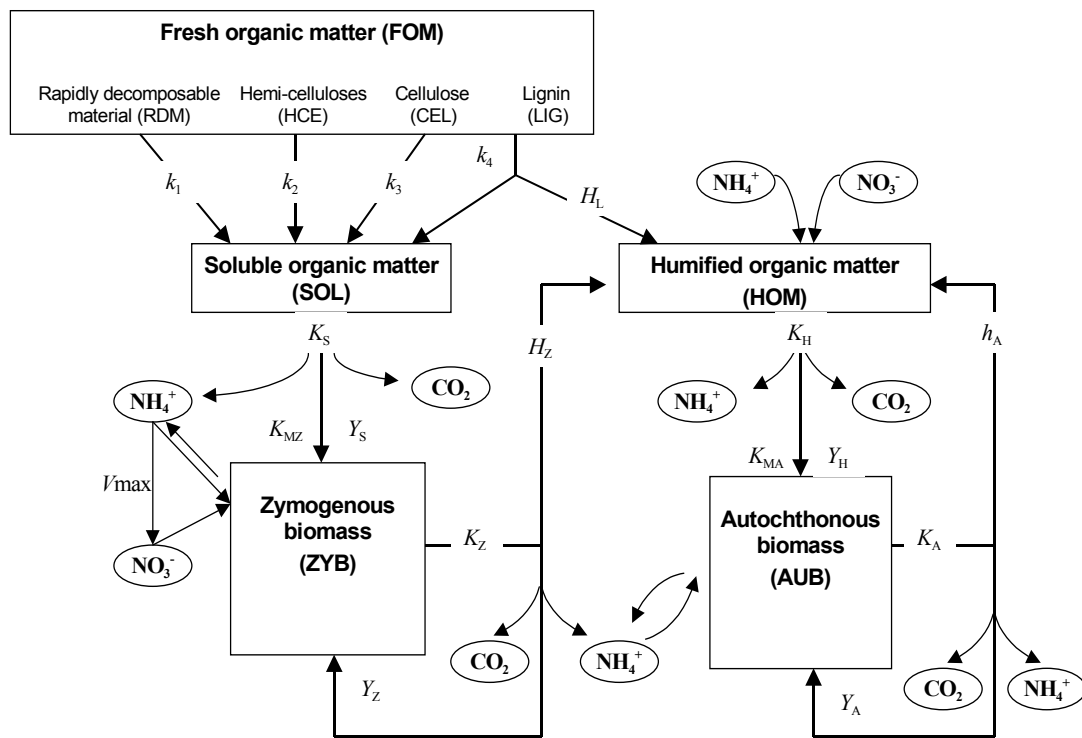


Figure A.1: Flow diagram of the CANTIS (Carbon And Nitrogen Transformations In Soil) submodel from Garnier et al. (2003). The meanings of the parameters are listed in Table A.1.

Table A.2: Biological parameters used in the CANTIS (Carbon And Nitrogen Transformations In Soil) submodel (Garnier et al., 2003).

Symbol	Parameter	Unit
k_1	Decomposition rate of RDM	d^{-1}
k_2	Decomposition rate of HCE	d^{-1}
k_3	Decomposition rate of CEL	d^{-1}
k_4	Decomposition rate of LIG	d^{-1}
k_S	Decomposition rate of SOL	d^{-1}
k_Z	Decomposition rate of zymogenous biomass (ZYB)	d^{-1}
k_A	Decomposition rate of autochthonous biomass (AUB)	d^{-1}
k_H	Decomposition rate of humified organic matter (HOM)	d^{-1}
Y_S	C assimilation yield of SOL by ZYB	-
Y_Z	C assimilation yield by ZYB	-
Y_A	C assimilation yield by AUB	-
Y_M	C assimilation yield of the humified pool by AUB	-
h_L	Humification coefficient by LIG	-
h_Z	Humification coefficient by ZYB	-
h_A	Humification coefficient by AUB	-
$V_{\max}$	Maximal nitrification rate	$mg\ N\ kg^{-1}\ d^{-1}$
K_{MZ}	Michaëlis-Menten constant for decomposition of SOL	$mg\ kg^{-1}$
K_{MA}	Michaëlis-Menten constant for decomposition of HOM	$mg\ kg^{-1}$

Mulch module

PASTIS was originally designed for incorporated residues. The model was adapted to the case of a surface residue mulch to take into account the physical effects of the mulch on rain interception and evaporation (Findeling, 2001), and the specific dynamics of the decomposition of the latter.

For rain interception, the mulch as a porous medium can intercept rainfall up to a maximum value $R_{m,max}$ (m) defined as:

$$R_{m,max}(t) = \frac{\theta_{me,max}}{\rho_{me}} DM_m(t) \quad (5)$$

where $\theta_{me,max}$ is the mulch element maximum volumetric water content ($\text{m}^3 \text{m}^{-3}$), ρ_{me} is the density of mulch components (kg m^{-3}) and DM_m is the mulch dry mass (kg m^{-2}). Mulch water storage, R_m (m), is calculated as:

$$\Delta R_m = \begin{cases} \Delta R \exp\left(-\alpha_m \frac{R_{m,max} - R_{m,res}}{R_{m,max} - R_m}\right) & \text{if } R_m < R_{m,max} \\ 0 & \text{if } R_m = R_{m,max} \end{cases} \quad (6)$$

where R is the cumulative rainfall (m), α_m is the mulch propensity to water recharge (-), and $R_{m,res}$ is the residual mulch water content (m) calculated from the residual mulch volumetric water content $\theta_{m,res}$ ($\text{m}^3 \text{m}^{-3}$) as in Eq. (5). The amount of rain which is not intercepted by the mulch is transmitted to the soil. When the flux of water reaching the soil exceeds the Darcy's infiltration flux, ponding water is stored and infiltrated later on in the soil.

For evaporation, the mulch acts as a physical barrier to convective and diffusive vapour fluxes between the soil and the atmosphere (Findeling *et al.*, 2003a). As a consequence, the total potential evaporation rate, E^{pot} ($\text{kg m}^2 \text{s}^{-1}$), is split into the potential evaporation rate of the mulch, E_m^{pot} ($\text{kg m}^2 \text{s}^{-1}$), and the soil, E_s^{pot} ($\text{kg m}^2 \text{s}^{-1}$):

$$\begin{aligned} E_m^{pot} &= \xi E^{pot} \\ E_s^{pot} &= (1 - \xi) E^{pot} \end{aligned} \quad (7)$$

where ξ is the propensity of the mulch to reduce soil evaporation demand (Findeling, 2001). E_s^{pot} is applied to the soil. E_m^{pot} is applied to the mulch to evaporate its water storage:

$$\Delta R_m = -\min(E_m^{pot} \Delta t; R_m) \quad (8)$$

For decomposition, residue decomposition is highly dependent on the contact with micro-organisms and the available mineral nitrogen resource (Henriksen and Breland, 2002; Garnier *et al.*, 2003). In the case of a mulch, the residue elements which are closely in contact with the soil will decompose more easily because micro-organisms from the soil can colonise them and fungi can use their hyphae to absorb mineral nitrogen in the soil to decompose efficiently the residue. On the contrary, the upper elements which are not in contact with the soil will hardly decompose because of water limitation which hinders micro-organisms activity. Based on an approach similar to that of Thorburn *et al.* (2001) in the APSIM-Residue model, two compartments for the mulch were defined (Figure A.2):

$$DM_m(t) = DM_{m,c}(t) + DM_{m,nc}(t) \quad (9)$$

Where $DM_{m,c}$ is the mulch dry mass in contact with the soil (kg m^{-2}) and $DM_{m,nc}$ is the mulch dry mass not in contact with the soil (kg m^{-2}). The former can decompose whereas the latter is assumed unavailable for decomposition. $DM_{m,nc}$ is set to an initial value and then decreases exponentially with time to account for the progressive rearrangement of the mulch due to climatic action (rain and wind). This decrease corresponds to an equivalent increase in $DM_{m,c}$:

$$\Delta DM_{m,c}(t) = \alpha_{feed} DM_{m,nc}(t) \Delta t \quad (10)$$

Where α_{feed} is the feeding rate (d^{-1}) of the decomposable mulch compartment by the non decomposable mulch compartment. Decomposition of $DM_{m,c}$ is based on the CANTIS submodel. $DM_{m,c}$ is split into fresh and soluble organic matter like incorporated residues in PASTIS. However, mulch decomposition is controlled by specific abiotic functions (Eq. (4)). The mulch temperature, T_{me} ($^{\circ}\text{C}$), and the mulch element volumetric water content, θ_{me} ($\text{m}^3 \text{m}^{-3}$), are used to calculate respectively f_T and f_W for the mulch decomposition. Only a limited topsoil layer can be involved in mulch decomposition since the influence of the zymogenous biomass, which decomposes the mulch, is limited by the maximum length of the fungi hyphae (Figure A.2). This limit is imposed by setting a maximum extension depth of the zymogenous biomass, $z_{m,zyb}$ (m), and considering that N immobilisation for mulch decomposition is restricted to the topsoil layer of thickness $z_{m,zyb}$. f_N is calculated from the available amount of mineral N in this layer. Finally, mineralised N from the mulch is assumed to be produced in the same topsoil layer.

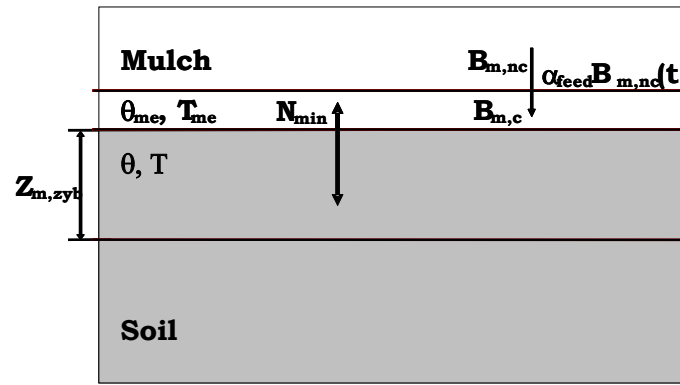


Figure A.2: Conceptual diagram of mulch decomposition process based on a to mulch compartments approach. The meanings of the parameters are mentioned in the text. ($B_{m,nc}$ and $B_{m,c}$ refer to $DM_{m,nc}$ and $DM_{m,c}$, respectively, in the text).