Modifications et restauration de propriétés physiques et chimiques de deux sols forestiers soumis au passage d'un engin d'exploitation

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Résumé

Les risques de dégradation physique des sols forestiers sous l'effet de contraintes mécaniques externes liées à la mécanisation des opérations forestières, augmentent considérablement. Les mécanismes et le temps nécessaires à la restauration non assistée de la qualité des sols forestiers tassés restent encore peu étudiés, et leur identification nécessite de coupler les approches physiques, chimiques et biologiques.

L'objectif de ce travail était d'étudier l'impact de la circulation d'un porteur forestier sur les conditions de l'enracinement (aération, régime hydrique et pénétrabilité) ainsi que son évolution à court terme. Ce travail s'appuie sur l'observation de deux sites expérimentaux mis en place dans le Nord Est de la France, concernant des sols de morphologie similaire (couche limono-argileuse de 50 cm d'épaisseur reposant sur un substrat argileux) et ayant subi des contraintes identiques. Des paramètres physiques (température et humidité du sol, densité apparente et résistance à la pénétration) et chimiques (composition de l'atmosphère du sol) ont été suivis pendant trois à quatre ans, à des fréquences allant d'un pas de temps quotidien à annuel.

Le suivi du climat du sol et de la composition de son atmosphère a mis en évidence une diminution forte des conditions aérobies pendant un à un an et demi après le passage du porteur. Cet effet initial sur l'aération du sol a diminué subitement dès l'apparition de la première période de sécheresse édaphique, probablement grâce à la formation de fissures dans l'horizon de surface du sol tassé. Cependant un effet significatif du traitement sur la composition de l'atmosphère du sol pouvait toujours être observé trois à quatre ans après tassement. Pour suivre l'évolution des propriétés physiques du sol après circulation du porteur, il a été nécessaire d'opérer une normalisation par rapport à l'humidité au moment du prélèvement. Trois ou quatre ans après la circulation du porteur, une différence toujours significative existe entre les propriétés physiques des sols témoins et celles des sols tassés. Cependant une évolution de l'impact du porteur peut être mise en évidence dans la couche de surface (0–10 cm) des deux sites. Ce début de restauration se traduit, sur un des deux sites, par une différence entre traitement qui n'est plus significative quand les sols sont humides mais qui l'est encore quand les sols sont secs. Sur le deuxième site, cette différence a diminué quelle que ce soit l'humidité du sol. Ainsi, le début de régénération de la structure du sol perturbé ne s'accompagne pas d'une disparition de son comportement de prise en masse lors de son desséchement sur un des deux sites.

Ce travail a permis de mettre en évidence une évolution des conséquences du porteur en surface du sol tassé qui serait liée à des processus physiques (gonflement – retrait, gel – dégel). Cependant, l'impact sur les conditions de l'enracinement (forte résistance à la pénétration quand les sols sont secs, faible aération quand ils sont humides) reste élevé de même que sur la résilience à long terme du peuplement.

Mots clés tassement sol, sol forestier, restauration naturelle, atmosphère du sol, densité apparente, résistance à la pénétration

Changes and recovery in soil physical and chemical characteristics of two forest soils following traffic by a full-loaded forwarder

Abstract

Soil compaction belongs to the major treats to soil quality with no exceptions of forest ecosystems where the frequency and intensity of loads application increase since several decades. The mechanisms and the duration of soil quality recovery following heavy traffic in forests remain poorly documented and their study requires multidisciplinary approaches.

The aim of this work was to evaluate the impact of forwarder traffic on the potential constraints to roots growth (aeration, water content, and penetrability) growing in two forest soils sensitive to compaction, and the evolution of these consequences in the short-term. This work is based on two experimental sites, set up in the northeast of France, with soils displaying similar morphologies (50 cm thick silt loam layer laying on a clayey layer) and having being loaded with the same forwarder under similar (wet) soil conditions. Soil climate (temperature and moisture), soil air composition, and soil bulk density and resistance to penetration were investigated continuously, monthly and yearly, respectively. Soil climate and air composition monitoring showed a strong initial decrease in aerobic conditions lasting one to one and a half year. The strong initial impact on soil aeration decreased concurrently with the first soil drought experimented at both sites, probably because of soil cracks formation in the disturbed soil. Yet heavy traffic still affected significantly soil air composition 3 to 4 years after compaction at both sites. To monitor changes in soil physical parameters, we had to standardize measures with regards to soil climatic conditions at the time of sampling. Three to four years after soil compaction, the difference in soil physical properties between treatments was still significant. However, changes in the impact of the forwarder traffic on soil physical characteristics could be stated in the surface layer (0–10 cm) of both sites. This beginning of soil restoration results at one of both sites in a difference between treatments that is no longer significant when soils are wet but that is still significant when the soils are dry. At the second site, the difference is still significant whatever soil moisture conditions but it has decreased since the start of the experiment. Consequently, this beginning of soil structure recovery is not accompanied by a disappearance of the hardsetting behaviour (decrease in hydrostructural stability) of the compacted soil at one site.

In this study changes in the consequences of the forwarder traffic were stated in the surface soil layers of both sites, these changes may be due to physical processes (wetting – drying, freezing – thawing). Nevertheless, the impact remains strong on roots growth (high resistance to penetration when dry, poor gas transfer when wet) and on stand resilience to external stresses (drought, storm).

Key words soil compaction, forest soil, natural recovery dynamic, soil atmosphere, bulk density, resistance to penetration.

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SOMMAIRE

INTRODUCTION

Les gestionnaires forestiers doivent actuellement faire face à un regain d'intérêt pour le bois en tant qu'éco-matériau, quelle que soit son utilisation. Très généralement dans le monde, mais de façon plus nette dans les pays du Nord où le coût de la main d'œuvre est élevé, l'intensification de la sylviculture s'est beaucoup traduite jusqu'ici par une utilisation accrue d'engins souvent lourds et puissants pour exploiter les forêts et/ou réaliser diverses opérations sylvicoles. Les évolutions technologiques du matériel forestier ainsi que, parfois, l'urgence d'intervention suite à des catastrophes naturelles conduisent à une circulation des engins quelles que soient les conditions météorologiques et l'état des sols, en particulier leur humidité.

La récolte d'une plus grande partie de la biomasse produite, le raccourcissement des révolutions, la substitution d'essences forestières et la mécanisation conduisent à une augmentation du risque de dégradation du sol dans ses diverses composantes (physiques, chimiques et biologiques) (Ballard, 2000) et pour toutes ses fonctions (production, fonctions écologiques et environnementales). Concernant plus particulièrement la dégradation physique du sol, les contraintes verticales mesurées lors d'opérations forestières peuvent aller jusqu'à 550 kPa avec des valeurs toujours supérieures à la capacité de la grande majorité des sols forestiers humides, sans être détrempés, à résister à une contrainte mécanique extérieure (portance) (Horn *et al.*, 2007).

Ainsi, le risque de tassement des sols forestiers devient très important. De plus ses conséquences peuvent conduire à une dégradation irréversible de leur fonctionnement, notamment par une augmentation de l'hypoxie et/ou de l'engorgement (Herbauts *et al.*, 1996). La restauration des dégradations étant toujours difficile, coûteuse et rarement durable, et la valeur économique de la ressource bois limitée, la recommandation pour la forêt française est de développer la prévention, en évitant ou limitant les dégradations faute de moyens efficients de restauration. Les avancées en matière de compréhension des mécanismes consécutifs au tassement des sols peuvent permettre d'orienter les méthodes de prévention.

Si les contraintes appliquées dépassent la capacité de portance du sol, le passage d'un engin va provoquer une diminution du volume de vides par rapport au volume de sol (porosité totale) et augmenter la résistance à la pénétration du sol jusqu'à ce qu'un équilibre s'instaure entre la déformation du sol et la contrainte exercée, équilibre qui peut s'établir après un ou plusieurs passages de l'engin. La diminution de la porosité totale (ratio du volume de vide sur le volume total du sol) se fait surtout aux dépens des macropores (*e.g.* Li & Zhang, 2009 ; Richard *et al.*, 2001 ; Schäffer, 2007). Ce terme de macropores se réfère à l'ensemble des pores de grand diamètre, généralement supérieur à 30 µm, comprenant par conséquent les pores inter-agrégats, les pores formés par l'action biologique (galeries de vers de terre ou racinaires) et les fissures formées par des phénomènes physiques (humectation – dessiccation, gel – dégel) (Bronick & Lal, 2005). Le volume de macropores n'est pas seulement fortement diminué mais leur morphologie (forme, taille, séparation et connectivité) est également modifiée lors du passage d'engins à la surface du sol (Schäffer, 2007). La macroporosité étant essentielle pour les transferts des fluides (gaz et eau) dans le sol, la compaction peut modifier drastiquement la capacité de drainage du sol et les conditions d'échange gazeux (Frey *et al.*, 2009). De plus, il se crée souvent une structure lamellaire, où les pores sont allongés et orientés de manière parallèle à la surface du sol (Werner & Werner, 2001). Cette structuration propre aux sols tassés conduit à des transferts de fluides principalement de manière globalmeent horizontale et non plus verticale (Horn, 2003). La réduction des capacités d'infiltration et de drainage peut conduire à un risque accru d'érosion sur des terrains en pente ou d'engorgement sur des terrains plats lors de fortes pluies (Croke *et al.*, 2001 ; Frey *et al.*, 2009).

En grande majorité, les modèles de prévision de la déformation du sol suite au passage d'un engin, de caractéristiques données, ont été calibrés sur des sols agricoles mais n'ont été que peu testés pour des sols forestiers dont les caractéristiques sont souvent très spécifiques (*e.g.* forte acidité conduisant à la présence dominante d'aluminium sur le complexe d'échange, ou au contraire forte alcalinité, avec présence en général d'une couche de litière, teneur élevée en matière organique dans l'horizon organo-minéral, présence de racines pérennes et d'éléments grossiers, contexte où l'hydromorphie est plus ou moins fréquente). Les modèles de prévision de la densification du sol sont en général limités dès que le sol présente des hétérogénéités telles que la présence de plusieurs couches horizontales aux comportements mécaniques différents, d'un squelette minéral (cailloux) et/ou biologique (racines) (Keller & Lamandé, 2010). De plus, ils ne considèrent généralement que des forces statiques et ne sont pas capables de prendre en compte la rugosité de la surface du sol (souches, microtopographie) lors du calcul des contraintes s'appliquant au sol. Les modèles les plus complexes et précis, capables de prendre en compte les hétérogénéités du sol, requièrent pour les mettre en œuvre beaucoup de paramètres inaccessibles aux gestionnaires (Cofie *et al.*, 2000). En outre, les modèles de compaction des sols prédisent généralement la déformation en termes d'augmentation de densité apparente et de déplacement (profondeur d'ornière). Il serait intéressant de développer des modèles permettant, à partir d'une variation de densité apparente, de prédire la variation de distribution de taille de pores par type de sol et historique des contraintes appliquées au sol, comme l'ont fait Li & Zhang (2009).

Prédire l'impact d'un engin donné sur la distribution de la taille des pores permettrait plus facilement de faire le lien avec son effet sur les propriétés hydriques, sur l'aération et sur la capacité à fournir un environnement adéquat aux racines ainsi qu'à la faune et à la flore du sol (macro et micro). En modifiant la densité du sol, l'accessibilité à l'eau et à l'air, le tassement change les conditions propices à l'activité biologique. Cependant, les conséquences du passage d'un engin sur cette dernière dépendent fortement de nombreuses interactions entre les paramètres physiques, chimiques et biologiques du sol (Whalley *et al.*, 1995). Les changements de l'environnement physique des microorganismes peuvent être bénéfiques aux communautés microbiennes en augmentant le volume des pores où elles peuvent se développer tout en inhibant le développement des prédateurs qui sont souvent plus volumineux (Shestak & Busse, 2005). Ils sont néfastes quand ils diminuent la capacité de transfert des fluides du sol (eau et gaz), créant des milieux hypoxiques voire anoxiques et à tendance hydromorphe (Schnurr-Pütz *et al.*, 2006; Weisskopf *et al.*, 2010). La croissance racinaire dépend essentiellement de deux facteurs physiques également fortement affectés par le tassement des sols ; une augmentation de la résistance à la pénétration et une diminution des échanges gazeux entre le sol et l'atmosphère (Conlin & van den Driessche, 2000 ; Gaertig *et al.*, 2002 ; Whalley *et al.*, 1995). La littérature est abondante sur les effets de la compaction sur les propriétés des sols, mais l'impact d'une dégradation physique du sol sur la croissance et la productivité des peuplements forestiers est beaucoup moins documenté (Miller *et al.*, 2004). Notamment, pour caractériser et comprendre l'impact global de la circulation d'engins d'exploitation sur les écosystèmes forestiers (sol et peuplement), il est nécessaire de coupler les approches à la fois physiques, chimiques et biologiques (Kremer, 1998). C'est ainsi qu'un réseau de sites du suivi à long terme de la qualité des sols (Long Term Soil Productivity Study Sites, LTSP) a été créé aux USA dans les années 1990, afin de rassembler les savoirs et savoir faire de différentes disciplines de science du sol et forestières dans le domaine (Powers, 2006). Ces sites permettent, entre autres, d'étudier l'impact du tassement de sols forestiers dû à la circulation d'engins d'exploitation sur leurs différentes fonctions, dont la productivité des peuplements. Ils visent à déterminer des valeurs seuils à partir desquelles la qualité des sols ne permet plus de soutenir une production économiquement viable (*e.g.* Page-Dumroese *et al.*, 2006 ; Fleming *et al.*, 2006 ; Carter *et al.*, 2006). Des groupes de travail mandatés par l'union européenne ont également identifié la caractérisation de l'impact du tassement sur la productivité végétale et les fonctions environnementales des sols (*e.g.* émissions de gaz à effet de serre, épuration de l'eau) comme une des priorités de recherche sur la dégradation de la qualité des sols (Van-Camp *et al.*, 2004). Cependant, ces experts mettent surtout l'accent sur les problèmes persistants causés par la compaction du sous-sol (en-dessous de la semelle de labour) des sols agricoles. Il est certain que ces problèmes sont essentiels car la restructuration artificielle et mécanique des sols agricoles ne peut se faire correctement qu'à des profondeurs limitées (Alakukku, 1996). Cependant, dans le cas des sols forestiers le problème de la persistance des effets de la compaction se pose dès les horizons de surface, surtout pour les sols les plus acides où l'activité de la faune du sol est limitée. En effet, les pratiques d'aide mécanique ou chimique à la restauration de la qualité physique des sols ne sont pas ou peu appliquées en gestion forestière. De plus, si l'amendement calco-magnésien a déjà fait ses preuves pour améliorer l'équilibre nutritionnel de peuplement poussant sur des sols pauvres, son intérêt pour accélérer la vitesse de régénération de sols acides et compactés reste encore à évaluer (Ampoorter *et al.*, 2011). En l'absence de remédiation mécanique ou chimique, la restructuration d'un sol se réalise grâce à des processus naturels, physiques [cycles de gel – dégel et humectation – dessiccation] (Pires *et al.*, 2008) et/ ou biologiques (faune du sol et activité des racines essentiellement) (Lister *et al.*, 2004 ; Capowiez *et al.*, 2009). Cependant, la croissance et l'activité racinaires sont fortement réduites lors du passage d'engin et peuvent mettre plusieurs dizaines d'années avant de revenir à de niveaux non perturbés (Von Wilpert & Schäffer, 2006). De même, la faune du sol peut être fortement affectée par la compaction et ne pas ou peu recoloniser les zones plus denses du sol (Capowiez *et al.*, 2009). Ainsi il a été mis en évidence que les cycles d'humectation – dessiccation sont essentiels dans la restructuration de sols limono-argileux (Werner & Werner, 2001). Cependant, même l'efficacité des processus physiques à restructurer les sols peut ne pas s'exprimer si, par exemple, l'enracinement étant contraint par le tassement du sol, la consommation d'eau est alors quasiment absente dans les horizons profonds limitant l'amplitude des cycles d'humectation – dessiccation (Horn, 2004). De même l'infiltration d'eau pouvant être réduite par le tassement, la ré-humectation des horizons profonds peut en être fortement affectée. Selon les études (type de sol, climat, impact initial, paramètres étudiés), le temps nécessaire au sol pour revenir à un fonctionnement non perturbé après tassement par une machine est très différent et varie de quelques années (*e.g.* Page-Dumroese *et al.*, 2006) à plusieurs dizaine d'années (*e.g.* Von Wilpert & Schäffer, 2006). En général, la restructuration du sol commence par les horizons supérieurs où elle est plus rapide qu'en profondeur (*e.g.* Froehlich *et al.*, 1985). En effet, dans les horizons profonds, l'activité biologique et l'amplitude et la fréquence des cycles d'humectation – dessiccation et de gel – dégel sont souvent réduites. Il ne faut donc pas négliger les effets à long terme du tassement du sol en profondeur dont la restauration est beaucoup plus lente que pour les horizons de surface du sol. L'impact à court et long terme sur la qualité physique du sol après l'application de la contrainte aura même tendance, avec le temps, à s'inverser sur le profil. L'enracinement va se limiter à la surface du sol avec des conséquences multiples, entre autres, sur l'ancrage et la stabilité des peuplements au vent ainsi que la sensibilité aux stress hydriques. On voit donc que, les risques pour la production et la stabilité des peuplements peuvent être très durables. A cet égard, l'utilisation de pneus larges à faible pression appliquée par unité de surface et/ou l'amélioration de la portance, conduisent en général à augmenter la charge des engins contraignant encore plus les horizons profonds du sol et la capacité même de restauration rapide. En effet, si les contraintes exercées à la surface du sol sont essentiellement liée à la pression de gonflage des pneumatiques, celles qui s'exercent en profondeur augmentent principalement avec la charge appliquée (Schjønning *et al.*, 2009).

Pour comprendre sous quelles conditions un sol peut se restaurer après dégradation par le passage d'un engin, il est nécessaire de coupler les approches physiques, biologiques et chimiques (Kardol & Wardle, 2010 ; Milleret *et al.*, 2009).

Face au besoin de mieux comprendre comment le sol se déforme sous les roues d'engins forestiers, comment cette déformation affecte les principales fonctions du sol (production forestière, mais aussi qualité de l'eau, émission de gaz à effet de serre) et combien de temps est nécessaire pour restaurer ces fonctions (Lamandé *et al.*, 2005), deux sites du suivi du tassement de sols forestiers ont été mis en place en France par l'INRA.

Mon travail de thèse vise à répondre aux objectifs suivants :

I. Analyser la capacité d'un modèle de compaction analytique fortement utilisé en milieu agricole à prévoir la déformation observée de deux sols forestiers soumis à la circulation d'un porteur chargé. Sachant que les sols des deux sites présentent de forte hétérogénéités verticales et qu'un seul a fait l'objet de mesures mécaniques (paramètre d'entrée du modèle), le test de ce modèle et de ses limites vise surtout à identifier les développements théoriques futurs pour améliorer sa qualité prédictive. L'objectif appliqué de ce volet consiste à tester la validité d'un modèle de compaction pour quantifier les dégradations physiques potentielles dans les écosystèmes forestiers en se basant sur un nombre limité de paramètres (caractéristiques de l'engin, texture et humidité du sol), et pour évaluer des méthodes de prévention.

II. Analyser les conséquences du tassement du sol sur la modification de sa porosité et de son fonctionnement biogéochimique. La sensibilité de deux paramètres physiques largement utilisés pour caractériser le tassement des sols (densité apparente et résistance à la pénétration) et d'un paramètre chimique (composition de l'atmosphère du sol) sera analysée. Cela permettra de tester la pertinence de considérer seulement des indicateurs physiques, ou seulement chimiques ou au contraire de coupler les approches afin de caractériser l'impact du passage d'un engin d'exploitation forestière sur la capacité du sol à offrir un environnement propice à la croissance racinaire et à l'activité biologique.

III. Etudier la vitesse de restauration de chacun des paramètres et identifier l'indicateur le plus pertinent pour le suivi de la qualité physique de sols forestiers. Il est certain que la vitesse de restructuration du sol va conditionner les décisions prises quant à la gestion des forêts et à la fréquence des passages d'engins ou au développement d'alternatives. Si la durée de restauration est plus longue que la fréquence de passage des engins, la contrainte sera appliquée sur des sols encore non, ou peu, restaurés, et les conséquences sur leur qualité seront d'autant plus fortes. La restauration est identifiée par différents indicateurs qui n'ont pas les mêmes sensibilités : ce n'est pas parce qu'un paramètre semble indiquer une certaine restauration que tous les autres vont suivre la même évolution, et que les conditions propices à la production végétale (enracinement, nutrition, alimentation en eau et air) sont toutes revenues à leur état initial. Il est donc important de connaitre les indicateurs les plus susceptibles de caractériser un réel retour à des conditions initiales.

Au vu des résultats, des recommandations seront établies pour aider le gestionnaire à décider des mesures préventives à prendre en fonction des conséquences physico-chimiques attendues suite au passage d'engin en forêt mais aussi des paramètres à caractériser pour établir un diagnostic de tassement (cas de dépérissement ou de problèmes de régénération).

Mon projet de thèse porte sur deux sols sensibles à la dégradation présentant des morphologies similaires (limon plus ou moins argileux reposant sur une couche très argileuse) mais dont les propriétés physico-chimiques et chimiques sont notablement différentes (*e.g.* minéralogie des argiles, pH, taux de saturation).

Le suivi « continu », pour de nombreux paramètres, des sols de sites expérimentaux pendant plusieurs années après l'application de la contrainte, permet de tester la sensibilité de chaque indicateur et sa capacité à discriminer le comportement de chacun des deux sols face aux contraintes et après.

Les différents articles présentés en annexe (I, II, III, IV et V) présentent les études individuelles de facteurs. Je propose une synthèse de toutes ces informations dans un schéma conceptuel mettant en évidence l'état du sol trois ou quatre années après l'application de la contrainte mécanique, en identifiant la dynamique de la restauration du sol et les indicateurs simples ou complexes pour caractériser celle-ci. À la suite de cette synthèse scientifique, je proposerai des recommandations pour la gestion durable de ces sols sensibles.

MATERIEL ET METHODES

SITES D'ETUDE

Les sites d'études ont été décrits dans les articles (I, II, III, IV et V) et sont présentés dans une plaquette rééditée annuellement (Ranger *et al.*, 2010). Je ne présenterai ici que les caractéristiques les plus importantes de ces dispositifs expérimentaux pour la compréhension de la synthèse.

Un projet de recherche (« dégradation physique des sols agricoles et forestiers liée au tassement », DST) cofinancé par l'agence nationale pour la recherche (ANR) et le ministère de l'écologie, de l'énergie, du développement durable et de l'aménagement du territoire (MEEDDAT) a permis l'installation, avec le soutien financier de l'office national des forêts (ONF), de deux dispositifs de suivi à court et long terme de la dégradation physique de sols forestiers suite au passage d'un débardeur (Ranger *et al.*, 2010). Les deux sites sont situés en forêt des « Hauts Bois », commune d'Azerailles (48° 29′ 19″ N, 6° 41′ 43″ E), Meurthe et Moselle (54), et en forêt de « Grand Pays », commune de Clermont en Argonne (49° 06′ 23″ N, 5° 04′ 18″ E), Meuse (55). Dans ce qui suit, je ferai référence aux sites d'Azerailles et de Clermont en Argonne par les abréviations respectives AZ et CA. Ces deux sites ont été choisis car ils présentaient des sols de morphologie similaire et sensibles au tassement (annexe 1). Cette sensibilité à des contraintes mécaniques provient à la fois de leur texture très limoneuse (plus de 50% de limons entre la surface et 50 cm de profondeur), et de la présence d'un engorgement temporaire du à la présence d'un substrat beaucoup plus argileux (issu de l'altération d'une marne du Keuper à AZ et de l'altération d'une gaize du Cénomanien à CA) et imperméable à partir de 50 cm de profondeur environ. Outre leur sensibilité au tassement et leur morphologie similaire, les deux sols diffèrent légèrement, notamment en terme de pH (*e.g.* à 0-10 cm le pH eau est de 4.4 à CA et 4.8 à AZ), de teneur en argile (10% de plus à AZ qu'à CA en surface), de minéralogie des argiles (présence d'argile gonflante à CA seulement), de rapport C/N (supérieur à 18 à CA, inférieur à 16 à AZ en surface) et de taux de saturation en base (29 et 63% à 0-10 cm de profondeur à CA et AZ respectivement) (I, II, III, IV et V, annexe 2). Ces différences vont permettre d'étudier des dynamiques spécifiques de comportement mécanique vis-à-vis des contraintes externes et de restauration de la porosité du sol.

Les deux sites ont fait l'objet d'un traitement expérimental identique (annexe 3). Le peuplement existant (futaie régulière de hêtre et chêne sessile avec un sous étage de charme, de bouleau et de tremble, plus abondant à CA qu'à AZ) a tout d'abord été coupé à blanc début 2007 pour les deux sites et débardé par câble-mât afin d'occasionner le moins de dégâts possible au sol. Les rémanents ont été ensuite ramassés manuellement ou en utilisant un cheval de fer, afin de limiter l'incorporation au sol de résidus d'exploitation lors du passage du débardeur. Chaque site a été divisé en trois blocs en accord avec la cartographie pédologique à grande échelle réalisée auparavant (annexe 2). En mai 2007 (AZ) et mars 2008 (CA) un porteur forestier a compacté le sol sur une bande de 30 × 50 m dans chaque traitement identifié au sein de chaque bloc. Les caractéristiques de la machine et les conditions du sol au moment du tassement sont détaillées dans l'article I. Les deux sites comportent 4 (AZ) et 5 (CA) placeaux de 50 × 50 m (deux bandes de 10 × 50 m sont laissées non perturbées par l'engin de part et d'autre de la bande tassée) par bloc avec les modalités suivantes :

-pour les deux sites, témoin (C, control), tassé (T, trafficked), tassé et décompacté par cover crop (D, non traité par la suite) ;

-puis à AZ, tassé et décompacté par potet travaillé au profit du plant (P, non traité par la suite) et à CA, témoin amendé (A, amended) et tassé amendé (TA, trafficked amended).

Le site CA a fait l'objet d'un chaulage en septembre 2008 (III) pour tester l'hypothèse d'une accélération de la restauration physique en améliorant les conditions physico-chimiques du sol (pH, garniture cationique de la CEC) et en stimulant l'activité biologique, dont les communautés lombriciennes dans ce milieu acide.

En novembre 2007 (AZ) et novembre 2008 (CA), une plantation a été réalisée à raison de 1600 plants ha^{-1} de chênes sessiles (*Quercus petraea* L.) ; les plants ont fait l'objet de dégagement les trois premières années après plantation, et de regarni, surtout à AZ (forte mortalité dans le traitement tassé).

INSTRUMENTATION ET SUIVI

Un suivi allant du pas horaire au pas annuel de différents paramètres a été mis en place sur les deux sites (Ranger *et al.*, 2010). Il inclut le suivi du mésoclimat (une station météorologique simplifiée par site mesurant en continu la pluie, la température et l'humidité relative de l'air), de paramètres biologiques (flore spontanée, mortalité et croissance des chênes sessiles à une fréquence annuelle), de paramètres physiques (profondeur et durée d'apparition de la nappe, température et humidité du sol) ; densité apparente, résistance à la pénétration, structure du sol [lames minces, méthode de Wind, porosimétrie mercure] à une fréquence annuelle) et de paramètres chimiques (composition de l'atmosphère du sol et des solutions de sol libre ou faiblement liée à la phase solide à une fréquence mensuelle). Des mesures ponctuelles interviennent également sur les sites, comme la détermination de la réserve utile du sol en avril 2009 (III), de la teneur en carbone et azote du sol en avril 2009, mai 2010 (AZ) et juin 2011 (mesure spatialisée, concernant uniquement la profondeur 0-10 cm) (I), de la résistance électrique du sol (thèse de Samir Seladji, 2010, Université de Pierre et Marie Curie UMR Sysiphe et INRA Orléans UR Sols), de la dynamique saisonnière de la stabilité structurale et de la conductivité hydraulique (post-doctorat de N. Bottinelli) ou encore de la courbe de retrait des sols (II).

Mon travail de thèse a porté sur la caractérisation de l'état du sol après tassement et sur la dynamique de son fonctionnement les premières années après l'application de la contrainte. Pour ce faire, j'ai étudié les relations entre les paramètres physiques du sol (masse volumique), le pédoclimat et l'atmosphère du sol à différentes profondeurs et dans les traitements C et T (IV et V). J'ai caractérisé leur réserve utile (III), leur densité apparente (I et II), leur résistance (III), leur comportement en rétractométrie (II) et la dynamique de la chimie du sol (I et III).

J'ai, pour chaque étude, favorisé l'analyse statistique par modèles linéaires mixtes qui permettent de considérer un effet aléatoire (bloc, capteur ou profil de sol selon le cas étudié) ainsi qu'une structure d'auto corrélation dans la distribution des résidus (cas fréquent lors d'étude de profil, les différentes profondeurs du même profil n'étant pas indépendantes les unes des autres ou lors d'études sur capteur permanent). Toutes les analyses statistiques ont été réalisées grâce au logiciel R.

SYNTHESE DES ACQUIS

PREVISION DE LA DEFORMATION

Un modèle analytique de compaction des sols (*SoilFlex*) développé par Keller *et al.* (2007) a été testé sur les deux sites en utilisant les caractéristiques du porteur, les mesures de densité apparente réalisées avant tassement, d'humidité des sols au moment de la circulation du porteur et de propriétés mécaniques réalisées à AZ (Saffih-Hdadi *et al.*, 2009). Pour CA, les paramètres mécaniques requis par *SoilFlex* (paramètres de la relation contrainte-déformation) ont été calculés à partir de des relations calibrées avec les données d'AZ entre paramètres mécaniques d'une part et densité apparente et humidité du sol d'autre part (fonction de pédotransfert). Pour tester la validité des simulations du modèle, nous avons comparé les valeurs de densité apparente après tassement prédite par le modèle à celles observées immédiatement après le passage du porteur (I).

La déformation est prédite de manière très satisfaisante surtout à AZ où les paramètres mécaniques ont été mesurés, ceci à l'exception de la couche de surface (0–10 cm) pour les deux sites. En augmentant la valeur du paramètre de « rebound » (indice de gonflement), les prédictions de densité apparente après compaction pour la couche de surface sont largement améliorées.

Nous avons émis l'hypothèse que cette augmentation du gain en volume dès que la contrainte est enlevée (déchargement) est intimement liée à la forte teneur en matière organique de l'horizon de surface. Cette hypothèse est renforcée par l'analyse de covariance réalisée sur la densité apparente de la couche 0–10 cm. En effet, dans le traitement C (control, sol non perturbé), plus le sol est riche en carbone, plus il est humide et moins il est dense, ce qui devrait augmenter sa sensibilité à la compaction (Saffih-Hdadi *et al.*, 2009). Cependant, dans le traitement T (trafficked, sol compacté), la relation entre densité apparente et humidité du sol d'une part avec la teneur en carbone organique d'autre part est inchangée et est parallèle à celle du traitement C. Ainsi plus le sol est riche en carbone plus sa sensibilité au tassement augmente si on considère les propriétés du sol qui déterminent usuellement cette sensibilité. Pourtant l'impact de l'engin sur sa densité apparente (différence entre les résultats des traitements T et C) est identique à celui d'un échantillon moins riche en carbone. Il semblerait qu'un paramètre mécanique supplémentaire soit à prendre en compte pour des sols riches en carbone, qui pourrait dépendre plus spécifiquement de la qualité de la matière organique (Soane, 1990).

Prendre en compte les valeurs extrêmes des conditions initiales (minima et maxima des valeurs de densité apparente et humidité avant circulation du porteur) n'a pas augmenté la qualité des prédictions de *SoilFlex* (I). Même si ce constat est à élargir et à valider dans d'autres cas de tassement de sols forestiers, il présente un intérêt dans le cas de ces sols fortement hétérogènes en simplifiant la procédure de calcul de déformation. Cependant, les résultats des simulations *SoiFlex* sont à considérer avec précaution dès que les sols sont très humides lors du passage de l'engin (cas qui peuvent se rencontrer même s'ils doivent être normalement évités, note de service ONF 09-T-297), probablement car l'estimation des propriétés mécaniques se fait en dehors des limites de validité des fonctions de pédotransfert (I).

L'utilisation de modèle analytique de compaction semble adaptée à la caractérisation du tassement de sols forestiers et devrait permettre d'identifier le choix de l'engin le moins susceptible de dégrader le sol dans des conditions données. Cependant, cela requiert encore de développer des fonctions permettant de calculer les propriétés mécaniques des sols en fonction de leur état d'humidité, notamment pour la couche organique de surface des sols forestiers.

Densité apparente

L'indicateur le plus souvent utilisé dans les études de qualité physique des sols est la densité apparente (masse de sol sec rapportée à son volume brut ou apparent). En effet, contrairement à d'autres paramètres physiques, la densité apparente est sensible aux actions anthropiques et, pour un sol donné, n'est normalement affectée que par ces actions (Schoenholtz *et al.*, 2000). De plus, la densification du sol est directement connectée à une perte de porosité et elle a pu être mise en relation avec une diminution de la prospection racinaire associée à une perte de productivité végétale dans certains cas (Miller *et al.*, 2004). Cependant, il a été démontré que ce n'est pas toujours l'indicateur de dégradation physique le plus sensible dans le cas de sols forestiers (Ampoorter *et al.*, 2010). Un des indicateurs inclus dans le Processus de Montréal (4.1.e) porte sur la quantification de la « proportion des surfaces forestières affectée par un changement significatif de densité apparente d'au moins un des horizons de surface du sol (0–30 cm) ». Pennington & Laffan (2004) ont testé la pertinence de cet indicateur en le mettant en pratique dans le suivi de l'impact d'exploitations forestières. Ils ont trouvé que les changements de densité apparente suite aux exploitations forestières pouvaient avoir lieu dans les deux sens (augmentation, ou diminution suite à l'incorporation des résidus organiques dans le sol). De plus la variabilité associée à la mesure de densité apparente est tellement forte qu'ils n'ont pas trouvé cet indicateur pertinent pour caractériser de manière facile, peu coûteuse et sensible, la dégradation du sol suite à des opérations sylvicoles. Notamment, la forte teneur en matière organique des sols forestiers (et la diversité probable de qualité de la matière organique entre les sites à comparer) joue un rôle non négligeable à la fois sur la densité apparente du sol (cf. Table 3 de l'article I, De Vos *et al.*, 2005) et sur son comportement mécanique (I). Ainsi la forte variabilité des propriétés physico-chimiques des sols forestiers due à l'absence d'homogénéisation par les pratiques culturales (De Vos *et* *al.*, 2005) peut conduire à une mauvaise estimation de l'impact d'une opération sylvicole mécanisée sur la densité apparente du sol (Ampoorter *et al.*, 2010). De plus, le comportement de gonflement – retrait des sols est fortement affecté par le tassement (Schäffer, 2007). Ainsi, l'amplitude de la différence de densité apparente entre un sol compact et un sol non perturbé peut varier fortement avec la teneur en eau du sol au moment de la mesure. Boivin *et al.* (2006) ont mis en évidence un effet significatif sur la courbe de retrait de sols dû à des passages répétés de voitures (parking temporaire) ; l'impact qui y était évalué par des mesures classiques (densité apparente, résistance à la pénétration, courbe de rétention en eau) n'était pas facile à interpréter, à cause de la forte teneur en éléments grossiers et des variations de teneur en eau.

Sur les deux sites de suivi du tassement des sols forestiers après le passage d'un porteur, l'évaluation de l'impact sur la densité apparente n'était également pas facile à interpréter sur quelques campagnes de prélèvement. Cette difficulté pouvait être attribuée à la fois à la variabilité spatiale des constituants du sol (*e.g.* I), à des phénomènes de gonflement – retrait (II) et à des problèmes méthodologiques de prélèvement volumique sur un sol engorgé (II) et/ou sur un sol sujet à des phénomènes d'éclatement – prise en masse lors des variations d'humidité (sol du traitement T à CA, article III ; de Lima *et al.*, 2005). Il faut noter que la prise en compte de l'effet de l'humidité au moment de la mesure de densité apparente a nettement amélioré la qualité des modèles prédisant la densité apparente en fonction du traitement et de la profondeur d'échantillonnage (II). Cependant cette approche n'a pas permis de distinguer la part de variabilité du paramètre « densité apparente » liée à la variabilité spatiale des constituants du sol de celle liée au phénomène de gonflement – retrait. De plus, les prélèvements sur fosses ayant eu lieu le plus souvent dans des conditions proches de la saturation, l'humidité au moment du prélèvement était également fortement liée à la structure du sol. Par conséquent, en standardisant par rapport à l'humidité, une part non estimable de l'impact du tassement sur la structure n'était plus caractérisée.

Par conséquent, si le paramètre « densité apparente» reste indispensable pour le suivi de la qualité des sols (Schoenholtz *et al.*, 2000), une standardisation des méthodes de mesure s'impose. Par exemple, soit la mesure de la densité apparente d'un échantillon de sol seulement une fois rééquilibré à un potentiel matriciel donné (Heuscher *et al.*, 2005) soit l'établissement de la courbe de retrait de chaque échantillon (Boivin *et al.*, 2006 ; Schäffer *et al.*, 2008) permettraient de limiter l'interférence avec le phénomène de gonflement – retrait dépendant de l'humidité, rendant ainsi comparables les résultats des différentes études. L'intérêt supplémentaire de mesurer le volume, la masse humide et la masse sèche d'un échantillon de sol équilibré à un potentiel donné est de pouvoir s'affranchir de la variabilité spatiale des constituants du sol, car l'humidité du sol à une succion donnée est fortement dépendante de ses constituants (Heuscher *et al.*, 2005). On pourrait ainsi standardiser la différence de densité apparente entre traitements vis-à-vis des différences en constituants des échantillons (essentiellement teneur en argile et carbone organique ; Schäffer *et al.*, 2008) sans avoir à mesurer les constituants du sol sur chaque échantillon. Cependant, selon le potentiel matriciel choisi, l'effet d'un constituant donné sur l'humidité du sol ne sera pas le même. De plus, dans les potentiels proches de la saturation en eau, l'humidité est également fortement liée à la structure du sol. Ainsi l'analyse de covariance expliquant la densité à ce potentiel (choisi proche de la saturation) en fonction de l'humidité et du traitement fausserait, dans une certaine mesure, la quantification l'impact du tassement sur la structure du sol. Il serait, par conséquent, plus intéressant, dans le cas d'études et de suivi de la compaction des sols, et comme déjà évoqué, soit de déterminer la courbe de retrait de chaque échantillon soit de déterminer la densité apparente d'échantillons rééquilibrés à un potentiel supérieur à la capacité au champ. En effet la déstructuration des sols sous les roues d'un engin a en général un effet faible sur les valeurs d'humidité à des succions élevées (Richard *et al.*, 2001) qui reflètent alors principalement l'effet des constituants du sol sur la structuration du sol.

De plus, il est indispensable, dans certain cas (*e.g.* sols très limoneux, présentant peu d'agents structurants comme les oxydes de fer et d'aluminium), de déterminer le comportement du sol dans les fortes et les faibles humidités pour vérifier qu'il ne présente pas les traits caractéristiques de la prise en masse lors du dessèchement et d'éclatement lors de l'humectation. En effet, si tel était le cas, il serait nécessaire de prélever les cylindres de sol dans une gamme d'humidité réduite afin de ne pas biaiser l'estimation de la densité apparente, l'humidité du sol devant plutôt se situer à des valeurs inférieures à la capacité au champ mais supérieures à un potentiel hydrique de -50 kPa (De Lima *et al.*, 2005). C'est particulièrement vrai dans le cas ou le tassement augmente le phénomène d'éclatement – prise en masse, la gamme d'humidité idéale pour le prélèvement d'échantillon de sol reste alors relativement large pour le sol non perturbé contrairement au sol compacté. Il est intéressant de noter que l'impact de la circulation d'un engin sur le phénomène d'éclatement – prise en masse (perte de la capacité du sol à résister aux contraintes internes liées à l'humectation – dessiccation) peut être caractérisé en une seule mesure par la courbe de retrait (Schäffer, 2007).

Résistance à la pénétration

La résistance à la pénétration du sol présente l'intérêt d'être une mesure non, ou peu, destructrice à l'échelle d'un site expérimental, pour comparer la facilité des racines à croitre dans le sol perturbé par rapport au sol non perturbé. Les appareils récents permettent, en outre, des mesures indépendantes de l'opérateur. Cependant, la valeur de la résistance mécanique du sol dépend fortement de son humidité qui est également affectée, d'une part, par les constituants et, d'autre part, par la compaction.

Notamment à CA, six mois seulement après le passage du porteur aucune différence de résistance à la pénétration ne pouvait être constatée entre les traitements T et C alors que plus d'un an après, une différence significative l'était (III). A AZ, quelle que soit l'humidité, la différence de résistance à la pénétration entre traitements était toujours significative, même si son amplitude variait fortement d'une campagne de mesure à l'autre. Cette différence de réponse de la variable résistance à la pénétration face à la dégradation physique a été attribuée à des différences de texture et d'agents structurants (notamment oxydes de fer et d'aluminium, cf. annexe 2). Le sol à CA est sensible à des phénomènes d'éclatement des agrégats quand le sol s'humidifie et de prise en masse quand le sol se dessèche, phénomènes accentués par la circulation du porteur. Ainsi à CA, l'impact du porteur sur la qualité physique des sols (densité apparente, résistance à la pénétration) peut paraître faible quand les sols sont humides au moment de la mesure (*e.g.* novembre 2008 pour la résistance à la pénétration *cf.* III, avril 2009 pour la densité apparente *cf.* I). C'est quand les sols sont secs que l'impact de la dégradation physique y est très important. Certes, les conséquences physiques du passage du porteur sont difficiles à quantifier à CA, mais cela n'empêche pas le dysfonctionnement fort (perte de stabilité structurale vis-à-vis des contraintes internes d'humectation – dessiccation) induit par la perturbation. Ce résultat est en accord avec les résultats de Schäffer (2007) obtenus par analyse des courbes de retrait de sols comprimés à différents niveaux de contraintes. Même si le passage d'un engin tend à augmenter la résistance du sol à des contraintes extérieures (*e.g.* contrainte de pré-compression, résistance à la pénétration), il tend également à diminuer sa résistance à des contraintes internes liées aux variations de potentiel matriciel. Ainsi, il n'est pas aberrant de trouver une résistance à la pénétration aussi, voire, plus faible, pour un sol compacté que pour un sol non affecté par la circulation d'engin quand les sols sont humides, même si ce n'est pas vrai pour tous les sols.

La redistribution des oxydes de fer observée sur les sols tassés des deux sites (hydromorphie, annexe 1) va probablement empêcher le retour à une stabilité structurale non perturbée par perte définitive des agents structurants. De plus, il a été montré que l'intensification de l'engorgement temporaire dans les sols tassés s'accompagne d'une augmentation de la dispersibilité des particules argileuses et de leur migration dans le profil de sol (Herbauts *et al.*, 1996), autre paramètre risquant d'accentuer la perte de stabilité structurale.

La résistance à la pénétration est un excellent indicateur de qualité physique du sol, car il est facile à mesurer et est peu destructeur. Sa mesure permet ainsi de faire une pré-analyse des propriétés physiques du sol ; que ce soit pour localiser les zones compactes à observer plus en détail (Page-Dumroese *et al.*, 2006) ou pour identifier un comportement spécifique et cibler les méthodologies à utiliser pour les prélèvements ultérieurs (*e.g.* à CA, comportement d'éclatement – prise en masse). Cependant, selon la teneur en eau du sol lors de la mesure, une différence de densité se traduira par une différence plus ou moins forte de résistance à la pénétration. Ainsi, pour caractériser une dégradation physique, il est conseillé de mesurer la résistance à la pénétration sur sols secs (Smith *et al.*, 1997). De plus la mesure de la teneur en eau à proximité de celle de la résistance à la pénétration est indispensable pour comprendre l'effet de la perturbation sur cette variable physique ainsi que pour estimer la valeur d'humidité du sol à partir de laquelle la résistance à la pénétration devient trop élevée pour la croissance racinaire (Leão & da Silva, 2004 ; Leão *et al.*, 2006). Cette valeur d'humidité entre en compte dans le calcul d'un indicateur proposé par da Silva & Kay (1997). Cet indicateur, nommé LLWR (Least limiting water range, *i.e.* étendue d'humidité la moins contraignante), est intéressant car il permet de comparer la qualité physique de différents sols en se basant sur les humidités auxquelles la croissance racinaire est contrainte. La limite inférieure de l'indicateur est la plus grande valeur entre l'humidité en deçà de laquelle la résistance à la pénétration est trop forte et l'humidité au point de flétrissement permanent (potentiel matriciel de -15 bar). La limite supérieure est la plus faible valeur entre l'humidité correspondant à 10% de porosité libre à l'air et l'humidité à la capacité au champ (valeur de potentiel variable selon les auteurs). Ainsi, LLWR correspond à une réserve utile en eau intégrant non seulement la capacité de rétention en eau du sol mais aussi des limites physiques de contraintes à l'enracinement (résistance à pénétration et aération suffisante). La réserve utile en eau (quantité d'eau comprise entre humidité à la capacité au champ et l'humidité au point de flétrissement permanent) est souvent supérieure au LLWR et les limites liées à l'aération et à la résistance à la pénétration des sols dépendent fortement de la densité apparente contrairement aux limites de la réserve utile en eau (da Silva & Kay, 1997). Le calcul du LLWR se base sur des relations entre humidité du sol et résistance à la pénétration pour calculer la teneur en eau en dessous de laquelle la résistance à la pénétration dépasse un seuil fixé à 2 ou 3 MPa selon les études. J'ai choisi le seuil de 3 MPa qui semble adapté en sols forestiers (Gartzia-Bengoetxea *et al.*, 2009). Puis j'ai calibré la relation entre résistance à la pénétration et humidité du sol sans prendre en compte l'effet « date » qui semble plus être lié à des gammes d'humidité différentes à chaque date qu'à une régénération du sol (III). Les valeurs d'humidité à la capacité au champ et au point de flétrissement permanent (succions de 0.33 et 15 bar, respectivement ; Baize, 2000) correspondent à celles du tableau 1 dans l'article III. Toutes les humidités pondérales mesurées (réserve utile) ou calculées (résistance à la pénétration) ont été converties en humidité volumique en prenant la densité apparente de la couche de sol concernée (10-20 cm) juste après tassement (II). L'humidité volumique correspondant à une porosité libre à l'air de 10% a été calculée grâce à la densité apparente juste après tassement et aux valeurs de densité de solide mesurées sur les sites.

Les limites LLWR ont été ajoutées à l'évolution de l'humidité volumique enregistrées par les sondes TDR (Figures 1a et b). On peut constater que pour les deux sites la limite inférieure d'humidité est liée à une trop forte résistance à la pénétration et que la limité supérieure correspond à la capacité au champ, ce qui est également le cas pour la majorité des sols (da Silva & Kay, 1997).

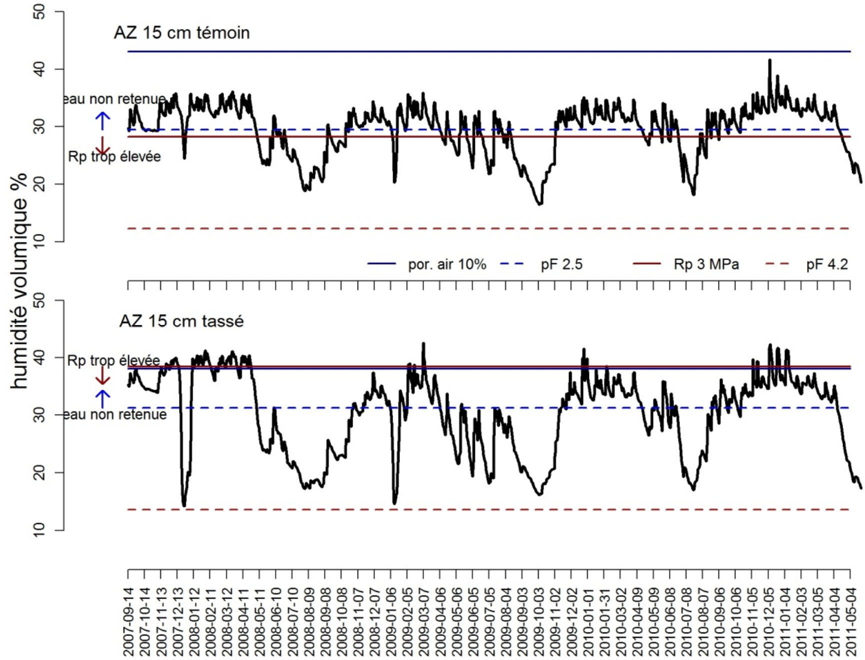


Figure 1a Evolution de l'humidité volumique à AZ à 15 cm de profondeur (sondes TDR, 5 répétitions, trait plein noir) et limites de teneur en eau correspondant au Least Limiting Water Range (traits pointillés rouges et bleu pour les limites correspondant à l'humidité au point de flétrissement permanent et à la capacité au champ, respectivement ; traits pleins rouge et bleu correspondant aux limites de trop forte résistance à la pénétration et de trop faible aération, respectivement).

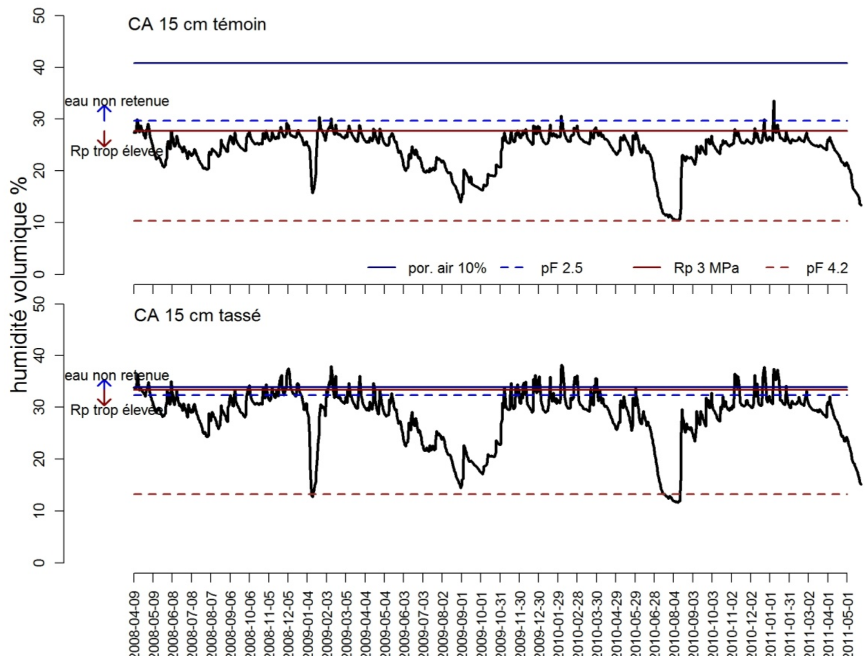


Figure 1b Evolution de l'humidité volumique à CA à 15 cm de profondeur (sondes TDR, 5 répétitions) et humidités limites correspondant au Least Limiting Water Range (même figurés que pour la figure 1a).

La valeur d'humidité correspondant à une porosité libre à l'air de 10% n'est jamais atteinte dans le traitement C des deux sites alors qu'elle est fréquemment atteinte dans le traitement T, occasionnant des conditions d'aération des sols insuffisante. De plus, au regard des limites inférieures et supérieures du LLWR, le sol témoin a une gamme d'humidité propice à la croissance racinaire certes limitée mais celle du sol tassé est inexistante pour les deux sites. En effet l'humidité en dessous de laquelle la résistance à la pénétration est égale ou supérieure à 3 MPa est supérieure à la capacité au champ et supérieure (AZ) ou légèrement inférieure (CA) à l'humidité à partir de laquelle l'aération est insuffisante. La valeur d'humidité en deçà de laquelle la résistance à la pénétration dépasse 3 MPa semble ainsi trop élevée. Il est probable que le seuil de résistance à la pénétration à partir duquel la croissance racinaire est limitée soit plus élevé avec le pénétromètre utilisé dans cette étude (PANDA®) que celui proposé dans d'autres études (2 à 3 MPa). En effet la relation entre croissance racinaire et résistance du sol à la pénétration dépend de l'essence, du type de sol considérés et du pénétromètre utilisé (Bengough & Mullins, 1990 ; Greacen & Sands, 1980). La majorité des auteurs qui étudient la relation entre croissance racinaire et résistance à la pénétration utilisent des pénétromètres dont la surface du cône inséré dans le sol a un diamètre de 1 à 2 millimètres (*e.g.* Bengough & Mullins, 1990 ; Bengough & Mullins, 1991 ; Zou *et al.*, 2001) alors que la pointe du pénétromètre utilisé dans cette étude a un diamètre de 16 mm. Par conséquent les valeurs absolues des humidités en deçà desquelles la résistance à la pénétration est trop élevée pour la croissance racinaire sont à prendre avec précautions. Pour compléter cette étude, il aurait fallu mesurer la relation entre croissance racinaire et résistance à la pénétration telle que mesurée par le pénétromètre PANDA® utilisé sur les sites d'AZ et CA (III). Par contre, la comparaison de ces valeurs entre traitements reste pertinente, le traitement n'influençant pas significativement la relation entre la résistance à la pénétration et l'humidité.

Le fait que les conditions de croissance racinaire sont moins propices dans le sol tassé que dans le sol témoin se traduit bien, au stade de croissance actuel, sur le système racinaire des chênes sessiles (photos 1a et b). Il sera particulièrement intéressant d'étudier la relation entre le développement du futur peuplement et cet indicateur.

Photo 1a Racines de chênes sessiles déracinés 4 ans après plantation. AZ, bloc II, témoin. Photos : B. Fatré et N. Bottinelli.

Photo 1b racines de chênes sessiles déracinés 4 ans après plantation. AZ, bloc II, tassé. Photos : B. Fatré et N. Bottinelli.

Si l'indicateur LLWR est prometteur car il intègre les différentes composantes des contraintes à l'enracinement, il est également intéressant de noter que cet indicateur ne tient pas compte de la dynamique temporelle de l'état des sols. Notamment le nombre de jours où l'humidité du sol se trouve en dehors des limites est également très important. Par exemple, un sol présentant une étendue de LLWR faible mais dont l'humidité n'est que rarement en dehors des limites est tout autant propice au développement racinaire qu'un sol avec une grande amplitude de LLWR mais dont l'humidité est souvent en dehors des limites. De plus, la détermination du LLWR requiert un certain nombre de mesures dépendantes de la méthodologie utilisée (*e.g.* résistance à la pénétration, correspondance potentiel hydrique – humidité) ; la validité de la comparaison des valeurs observées entre études en est fortement réduite.

Si l'indicateur LLWR est sensible au traitement appliqué sur les deux sites, il ne donne qu'une gamme potentielle de teneur en eau où la croissance racinaire est contrainte ; il ne faut pas oublier que le tassement affecte également le régime hydrique des sols. Ainsi lors du premier printemps expérimenté par les chênes sessiles dans le traitement T à AZ, le nombre de jours avec une humidité supérieure à la limite correspondant à une aération du sol suffisante a été très élevé (printemps 2008, Figure 1a) et bien supérieur à celui du traitement T à CA (printemps 2009, Figure 1a). Ceci explique peut-être en partie la forte mortalité après plantation des chênes sessiles dans le traitement T (25%) par rapport au traitement C (9%) à AZ, alors qu'à CA cette mortalité est de 8% dans le traitement T et 7% dans le traitement C.

Da Silva & Kay (1997) ont proposé des fonctions de pédotransfert permettant à partir des valeurs de densité apparente et de teneurs en carbone organique et en argile, de prédire les limites LLWR. Ces fonctions de pédotransfert ont été testées en utilisant les données mesurées sur les sites avant exploitation (densité apparente avant compaction, teneur en argile et en carbone) et sur les valeurs de densité apparente prédites par les simulations du modèle de compaction *SoilFlex* donnant les meilleurs résultats (I). On trouve à nouveau que la limite inférieure d'humidité est liée à une trop forte résistance à la pénétration et que la limite supérieure correspond à la capacité au champ. La comparaison des valeurs d'humidité volumiques trouvées avec les fonctions de pédotransfert proposées par da Silva & Kay (1997) avec celles mesurées ou calculées sur les sites est présentée dans le tableau 1.

	AZ C 15 cm		AZ T 15 cm		CA C 15 cm		CA T 15 cm	
	FPTs	mesures	FPTs	mesures	FPTs	mesures	FPTs	mesures
WC_{cc}	0.30	0.30	0.30	0.30	0.22	0.30	0.21	0.32
$WC_{\varepsilon a}$	0.46	0.43	0.37	0.38	0.44	0.41	0.29	0.34
WC_{Rp}	0.13	0.28	0.19	0.38	0.07	0.28	0.15	0.33
WC_{pfp}	0.16	0.12	0.16	0.14	0.09	0.10	0.10	0.13

Tableau 1 Humidités volumiques correspondants aux limites de l'indicateur LLWR. Calcul en utilisant les fonctions de pédotransfert proposées par da Silva & Kay (1997) (FPTs) comparé aux mesures réalisées sur les deux sites (AZ, Azerailles ; CA, Clermont en Argonne) dans les traitements C (témoin) et T (tassé). WC_{cc}, humidité à la capacité au champ (-0.33 bar) ; $WC_{\varepsilon a}$, humidité correspondant à une porosité libre à l'air de 10% ; WC_{Rp}, humidité en deçà de laquelle la résistance à la pénétration dépasse 3 MPa ; WC_{pfp}, humidité au point de flétrissement permanent.

Les fonctions de pédotransfert indiquent bien que l'impact de la circulation de l'engin se traduit principalement aux niveaux des limites liées à la résistance à la pénétration et à l'aération. Cependant on peut constater de fortes différences entre nos valeurs et celles prédites par fonction de pédotransfert concernant l'humidité à partir de laquelle la résistance à la pénétration atteint la valeur de 3 MPa. Cette différence de valeur d'humidité à partir de laquelle la résistance à la pénétration dépasse les 3 MPa peut être liée aux différentes méthodologies utilisées pour la déterminer. Da Silva & Kay (1997) étudient la relation entre résistance à la pénétration et humidité avec un pénétromètre dont la pointe a un diamètre de 4 mm (contre 16 mm dans notre cas) et, de plus, sur des cylindres de sol et non sur un profil de sol (III).

Il n'en reste pas moins que, même si la détermination de la limite du LLWR relative à la résistance à la pénétration semble dépendre fortement de la méthodologie et du pénétromètre utilisé, cet indicateur présente l'avantage de prendre en considération la majorité des conséquences physiques du tassement en un seul concept. Cet exemple montre l'intérêt de quantifier de manière précise l'impact du passage d'un engin à la fois sur la densité apparente, sur la relation entre résistance à la pénétration et teneur en eau, et sur la courbe de rétention des sols en eau. De plus, il serait intéressant de continuer le développement de fonctions de pédotransfert pour prédire l'effet de l'engin sur l'indicateur LLWR à partir du résultat d'un modèle de compaction (augmentation de densité apparente). Coupler la prédiction du LLWR, en fonction de l'état du sol et des engins utilisés, à un modèle hydrique, améliorerait probablement notre compréhension de l'impact du passage d'engins sur la croissance des peuplements et leur développement racinaire. Cela permettrait une grande avancée dans l'aide à la décision des gestionnaires forestiers.

Composition de l'atmosphère du sol

Plusieurs travaux ont démontré que la variation de densité apparente était en fait un indicateur peu sensible pour quantifier l'impact du passage d'engin sur le fonctionnement du sol (Hånkansson & Lipiec, 2000 ; Frey *et al*., 2009 ; Ampoorter *et al*., 2010). En particulier, les conditions de transferts des gaz et d'eau sont plus affectées par la diminution du volume de pores grossiers plutôt que par la diminution de porosité totale (Alaoui *et al.*, 2011). La densité apparente du sol ne donne qu'une image très imparfaite de l'impact d'un engin sur la structure du sol (Alaoui *et al.*, 2011 ; Hankansson & Lipiec, 2000). De plus, la diminution de la porosité totale est souvent associée à une réorganisation de l'espace poral, *e.g.* changement de distribution de taille de pores, diminution de la connectivité porale. Ainsi, il est probable que la dynamique des gaz de l'atmosphère du sol après tassement soit plus sensible à la dégradation physique, due à des contraintes externes, que des paramètres physiques plus globaux comme la densité apparente ou la résistance à la pénétration (Ampoorter *et al*., 2010). En effet, la concentration de l'atmosphère du sol en un gaz donné dépend de la résultante entre sa vitesse de production par rapport à celle de consommation et son transfert vers l'atmosphère. La vitesse de production/ consommation des gaz est liée à l'activité biologique qui est essentiellement déterminée par des facteurs biotiques (interactions entre organismes vivants) et abiotiques (température et humidité du sol), eux mêmes affectés par la déformation du sol. La vitesse de transfert d'un gaz donné est essentiellement déterminée par le flux de diffusion gazeuse et donc, par le gradient de concentration du gaz et le coefficient de diffusion gazeuse. Le coefficient de diffusion gazeuse est très sensible à la dégradation physique, à la fois à cause de la déstructuration de la phase solide et du changement de régime hydrique.

Sur les deux sites étudiés, l'analyse de la composition de l'atmosphère du sol a montré une forte réactivité à la dégradation des propriétés physiques du sol (IV). Elle renseigne sur la résultante des fonctions sources et puits, et par là même, directement sur les conditions d'accessibilité en différents gaz pour le développement racinaire et l'activité biologique. La méthode utilisée ne donne pas accès au dosage des gaz dissous ; quand le sol est gorgé d'eau, elle ne permet pas de ne prélever que les gaz du sol, et un vacutainer rempli d'eau est impossible à doser en chromatographie gazeuse. On a constaté que les capteurs de gaz des placeaux T étaient plus souvent engorgés que ceux des placeaux C, indicateur indirect des conditions d'accessibilité des racines au dioxygène (quasi-nulles dans ce cas précis).

En identifiant les causes de variation des concentrations de l'atmosphère du sol en différents gaz (O_2, CO_2, et N_2O), l'impact de la circulation du porteur forestier sur ces concentrations peut être analysé plus finement (IV et V). Ainsi, sur les deux sites étudiés, la porosité libre à l'air et la température du sol expliquaient une grande partie des variations des concentrations de l'atmosphère du sol en CO_2 et O_2, mais pas en N_2O (IV et V). Or le porteur a causé une diminution significative de la porosité libre à l'air sur l'ensemble de la période étudiée, qui est directement liée à une diminution de la porosité totale et une augmentation de l'humidité volumique, mais n'a pas effecté significativement la température du sol. Par conséquent, pour comparer la résultante entre consommation et production de CO_2 et O_2 entre traitements, il faut se placer à conditions de diffusion équivalentes (même porosité libre à l'air), même s'il est possible que la relation entre porosité libre à l'air et coefficient de diffusion soit également affectée par le traitement. On constate que trois (CA) ou quatre ans (AZ) après compaction, dès que les conditions de diffusion sont suffisantes (été ou printemps sec), le traitement T contient plus d'O_2 et moins de CO_2 que le traitement C. On peut donc supposer que, sur les deux sites, la circulation du porteur a occasionné une diminution de l'activité biologique consommant O_2 et produisant CO_2, ce qui a été vérifié à AZ par des mesures d'efflux de CO_2 (IV). A l'inverse, dès que les conditions de diffusion se dégradent (hiver, printemps humide) la porosité libre à l'air dans le traitement T étant plus faible que dans le traitement C, le traitement T contient moins de O_2 et plus de CO_2 que le traitement C, malgré la consommation de O_2 et la production de CO_2 moindres dans le traitement T. L'impact du tassement sur la composition de l'atmosphère du sol varie donc fortement en fonction de la saison considérée (IV et V).

Pour les deux sites, les deux gaz les plus sensibles au tassement sont CO_2 et O_2. Sur l'ensemble de la période étudiée, les deux autres gaz étudiés N_2O et CH_4 n'étaient présents dans l'atmosphère du sol qu'à des concentrations très faibles, difficilement détectables. De plus, des problèmes liés à la présence de méthane résiduel dans le système après étalonnage du chromatographe en phase gazeuse ont empêché l'analyse de l'effet du traitement sur sa concentration dans le sol. La concentration de l'atmosphère du sol en N_2O est plus faible dans les couches de surface (0-30 cm) et plus élevée dans les couches profondes (30-60 cm) du traitement T par rapport au traitement C. Deux hypothèses sont envisagées pour expliquer ces observations. La première serait que la production d'azote minéral est probablement plus faible et/ou que la consommation d'azote minéral est plus forte dans le traitement T que dans le traitement C. La seconde se base sur le fait que le N_2O pourrait être réduit en N_2 dans le traitement T où les conditions sont plus anoxiques que dans le traitement C, même si cette deuxième hypothèse semble peu probable étant donné le pH du sol.

QUEL INDICATEUR POUR QUELLE RESTAURATION?

Densité apparente

Les questions relatives à l'utilisation de la densité apparente comme indicateur d'impact des activités d'exploitation exerçant de fortes pressions sur les sols ont déjà été soulevées par de nombreux auteurs. La valeur de la densité apparente est très dépendante des constituants des sols, ce qui peut empêcher la caractérisation de l'impact d'une même pratique culturale sur différents sols (da Silva *et al*. 1997) et son évolution dans le temps (mesure destructrice qui empêche le prélèvement des échantillons successifs au même endroit). Plusieurs solutions ont été alors proposées :

- Normaliser l'impact de pratiques culturales sur la densité apparente par rapport aux constituants du sol via une régression multivariée. L'inconvénient de cette approche est qu'elle requiert une méthodologie lourde avec la mesure, pour chaque échantillon de sol, de sa densité apparente, ainsi que de ses constituants chimiques (en particulier le carbone organique) et physiques (en particulier la teneur en argile).

- Normaliser la densité apparente mesurée pour chaque cas, à la densité apparente de référence obtenue grâce à un test de compaction standard (Proctor ou compression uniaxiale d'échantillons à une pression de 200 kPa). Cette technique permet d'obtenir un degré de compaction (densité apparente/densité apparente de référence) qui ne dépend alors que des pratiques culturales et permet de discriminer ces pratiques même en comparant des sols très différents (da Silva *et al*. 1997 ; Håkansson, I., Lipiec, J. 2000). Cependant, cela nécessite un appareillage lourd pour déterminer la densité apparente de référence ou, à nouveau, la détermination des propriétés physico-chimiques pour estimer la densité apparente de référence par des fonctions de pédotransfert avec les limites de fiabilité que cela suppose (*e.g.* Keller & Håkansson, 2010).

Ces deux premières solutions permettent en effet de s'affranchir de l'effet local de la variabilité des constituants du sol sur la caractérisation de l'état de compaction de sols différents soumis à une même pratique culturale. Elles sont adaptées pour l'étude de l'effet d'un engin par exemple, mais seulement à un instant donné. Quand on veut ensuite étudier la dynamique temporelle de l'effet de cette contrainte, il faut non seulement prendre en compte la variabilité des constituants du sol mais aussi la variabilité des conditions d'humidité. En effet, des phénomènes de gonflement – retrait peuvent conduire à un biais dans l'étude de la dynamique de l'état de compaction d'un sol si les conditions de saturation en eau sont différentes (Schéma 1). C'est pourquoi l'analyse de la courbe de retrait offre un grand potentiel pour étudier la dynamique de restauration de la structure du sol.

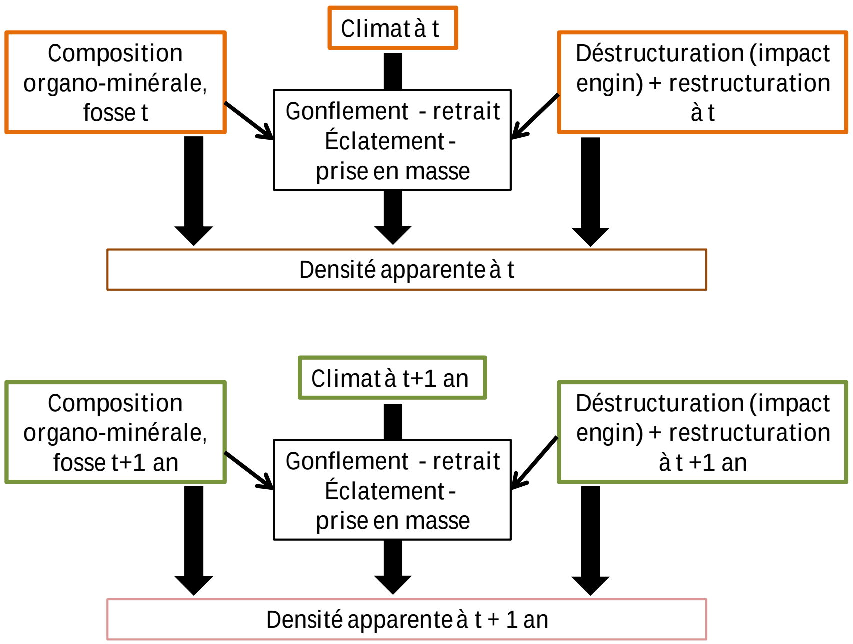


Schéma 1 Facteurs pouvant influencer la mesure de densité apparente et la détection de sa dynamique temporelle. Les couleurs des cadres des trois facteurs identifiés sur les sites étudiés (composition organo-minérale, climat et déstructuration/ restructuration) sont différentes à t et t+1 an pour indiquer de probables différences dans l'état de ces trois facteurs entre les deux campagnes de mesure de l'indicateur densité apparente. Le facteur étudié dans cette thèse (déstructuration/ restructuration) n'est pas le seul à influencer l'indicateur densité apparente, d'où l'intérêt de normaliser les mesures.

La solution présentée dans l'article II se base sur le fait qu'à un instant donné l'humidité mesurée est intimement liée aux constituants du sol. Par exemple, pour l'horizon de surface l'humidité du sol à un instant donné dépend significativement de la teneur du sol en carbone organique (I). Même si d'autres constituants et état du sol (*e.g.* succion, capacité de drainage) peuvent également influencer cette valeur d'humidité à un instant et à un endroit donné, je ne considère ici que la teneur en carbone à titre d'exemple car nous n'avons pas mesuré les autres paramètres dont dépend la teneur en eau du sol. En prenant en compte la co-variable humidité au moment du prélèvement, on peut normaliser la valeur de densité apparente mesurée à une teneur de carbone commune aux deux traitements sans avoir à mesurer cette teneur en carbone. On s'affranchit ainsi du fait que, selon la localisation des prélèvements, l'effet apparent du traitement peut être biaisé entre autres par la variabilité spatiale des constituants du sol. Cependant, cette normalisation n'intervient qu'à un instant t et il aurait fallu mesurer la courbe de retrait des sols ou mesurer le volume de sol après équilibre au même potentiel chaque année pour pourvoir comparer l'effet du traitement dans la même gamme d'humidité d'une année à l'autre. En effet, la relation entre le volume spécifique du sol et son humidité est non linéaire et dépend de la structure du sol (Boivin *et al.*, 2006 ; Schäffer, 2007). Selon le potentiel auquel on étudie l'effet du traitement sur la densité apparente, l'amplitude de cet effet ne sera pas la même. Par exemple, en mai 2011, l'effet du traitement sur la densité apparente n'était plus significatif sur la couche de surface (0–10 cm) à CA, même après normalisation par rapport à une différence en teneur en eau (I). En juin 2011, lors des campagnes de mesures de la résistance à la pénétration des sols, des prélèvements de sol par des cylindres à densité ont été réalisées pour cette même couche de surface, afin d'analyser leur teneur en carbone et en azote, leur humidité et leur densité apparente selon le même protocole spatialisé décrit dans l'article I. L'effet du traitement sur la densité apparente est alors significatif à CA (Figure 2) et a été interprété par un mécanisme de « prise en masse ». Ce mécanisme fait que le sol du traitement T est significativement plus compact que le sol du traitement C, seulement pour des humidités pondérales inférieures à 0.25 g g^{-1}. Par contre à AZ, en juillet 2011, l'interaction entre l'effet du traitement et l'effet de l'humidité n'est pas significative, l'effet de la circulation du porteur sur la densité apparente reste significatif quelle que soit l'humidité. On retrouve ici les conclusions de l'analyse de la dynamique temporelle de la résistance à la pénétration ; la structure du sol serait plus stable lors de l'humectation – dessiccation à AZ qu'à CA et le mécanisme de « prise en masse » n'interviendrait pas à AZ. Cet exemple montre bien que la normalisation des variations de densité apparente par rapport aux variations d'humidités année par année ne permet pas une comparaison entre année qui soit totalement satisfaisante, en particulier à CA à cause du comportement de d'éclatement – prise en masse des sols tassés.

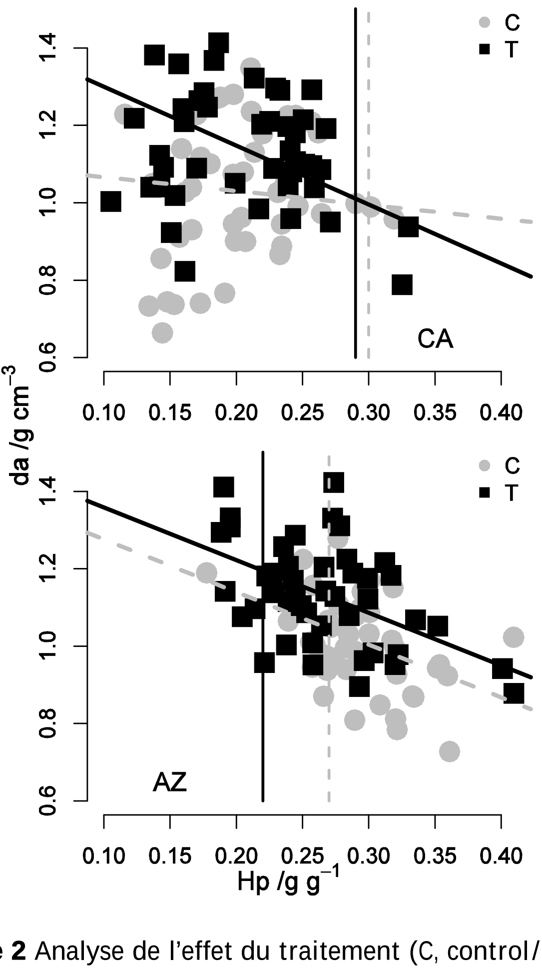


Figure 2 Analyse de l'effet du traitement (C, control/ témoin ; T, trafficked/ tassé) sur la densité du sol (da) de la couche de surface (0–10 cm) en prenant en compte l'effet de l'humidité pondérale du sol au moment du prélèvement (Hp), les points correspondent aux mesures et les lignes aux modèles expliquant da en fonction de Hp et du traitement (gris, témoin et noir, tassé). Les prélèvements ont eu lieu en même temps que les mesures de résistance à la pénétration des sols en juin 2011. Les barres verticales correspondent aux humidités lors des mesures de densité apparente sur fosses de mai 2011 (II).

Cet exemple illustre bien la nécessité de mesurer la densité de sols dans le même état hydrique d'une campagne d'échantillonnage à l'autre, pour analyser son évolution temporelle. Si les mesures interviennent une fois à l'état humide puis une fois à l'état sec, les conclusions quant à la restauration ou pas de la densité apparente du sol peuvent être biaisées. Les différentes campagnes de mesure de la densité apparente ayant toutes eu lieu quand l'humidité du sol était supérieure à 0.25 g g^{-1} à CA (Table 4 de l'article II), on peut valider l'évolution significative de l'impact du porteur sur la densité apparente de sols humides de la couche de surface. Cette régénération ne s'accompagne pas d'une disparition du comportement de prise en masse des sols tassés à CA qui témoigne de la déstructuration toujours réelle du sol compacté (Schéma 2).

A AZ, on observe également une tendance à la diminution de l'impact du porteur sur la densité apparente de la couche de surface, mais la vitesse de retour à un état non perturbé est plus faible qu'à CA qui est pourtant plus acide. La restauration du paramètre « densité apparente » est par conséquent probablement due à des processus physiques. Notamment l'amplitude des variations temporelles des teneurs en eau est plus forte dans le traitement T que dans le traitement C et pourrait expliquer un début de restructuration par le processus de gonflement – retrait (II). Nous avons émis l'hypothèse que la texture et la minéralogie des argiles jouaient un rôle primordial lors de la restructuration d'un sol tassé, surtout lorsque cette dernière est liée à des processus physiques (II). Il est certain que les résultats de cette étude ne sont que préliminaires et restent à valider sur le long terme. De plus nous émettons l'hypothèse que cette restauration sera beaucoup plus lente pour les couches inférieures moins sujettes à la bioturbation et aux cycles d'humectation – dessiccation et gel – dégel que les horizons superficiels du sol.

Nos résultats soulignent l'importance de mesurer la densité apparente d'un sol chaque année dans le même état d'humidité pour suivre sa dynamique (Håkansson & Lipiec, 2000). Cette condition est impossible à réaliser sur le terrain, d'où l'intérêt de mesurer le volume de chaque échantillon au laboratoire une fois que l'échantillon aura été rééquilibré à un potentiel choisi pour s'affranchir des variations de conditions de gonflement – retrait d'une année à l'autre. Dans le cas de sol présentant un comportement de dispersion – prise en masse, il est plus intéressant de mesurer sa densité apparente à des humidités proches du point de flétrissement permanent (*e.g.* une fois l'échantillon séché à l'air). Une fois les volumes mesurés, une mesure du poids humide puis du poids sec permet de normaliser par rapport à l'humidité et donc par rapport à la variabilité spatiale des constituants du sol d'un échantillon à l'autre. Cependant cette approche ne permet pas d'obtenir les valeurs de densité apparente quelle que soit l'humidité et notamment ne permet pas de calculer de manière très précise la valeur d'humidité correspondant à une valeur de porosité libre à l'air de 10%. Il peut alors être envisagé de faire également une mesure de volume et masse de sol frais (avant séchage à l'air). L'intérêt de ces deux mesures de volume (à l'état frais et sec) doit cependant être testé, au préalable, par rapport à des mesures de volume sur tout l'éventail d'humidité, telle que réalisé lors de la construction de la courbe de retrait.

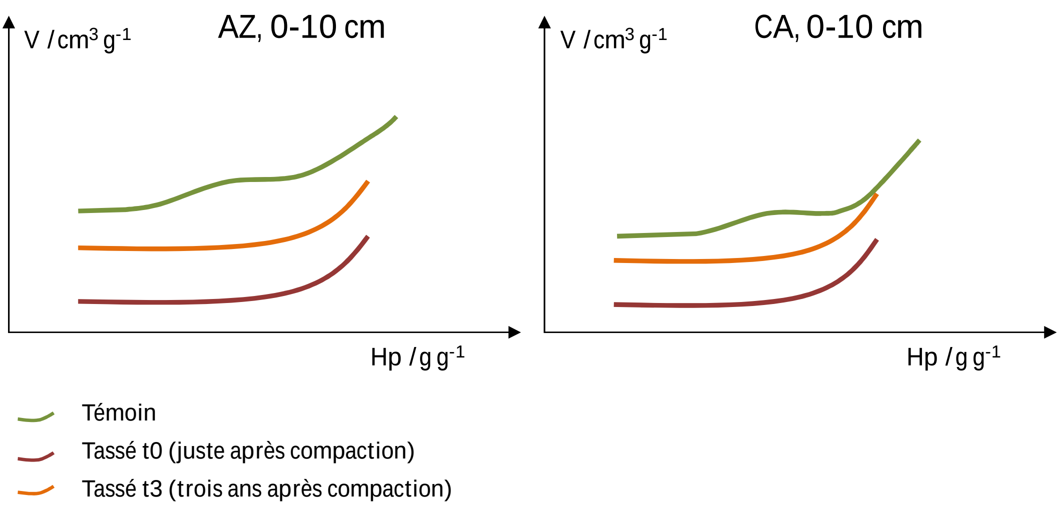
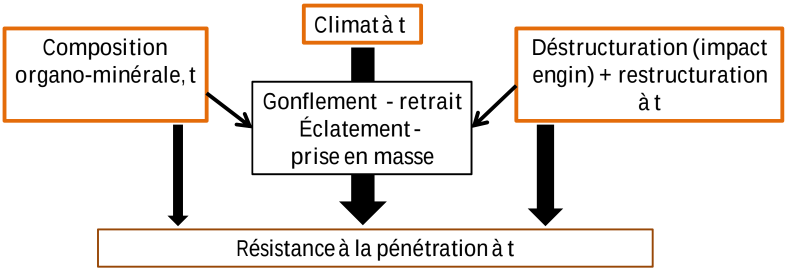


Schéma 2 Hypothèses concernant l'impact du porteur et l'évolution de cet impact sur le volume spécifique du sol (V, inverse de la densité apparente) et le comportement de gonflement – retrait à Azerailles (AZ) et Clermont en Argonne (CA). L'étendue complète des teneurs en eau (Hp) est considérée, ce schéma se base sur l'ensemble des observations faites sur les deux sites (I, II et III).

Résistance à la pénétration

L'étude de la résistance à la pénétration sur les deux sites a montré une absence de restauration de ce paramètre. Trois à quatre ans après compaction, les sols du traitement T ont toujours une résistance à la pénétration, après normalisation avec la teneur en eau, significativement plus élevée que ceux du sol C (III). L'amendement calco-magnésien n'a pour l'instant pas diminué l'impact du tassement sur la résistance à la pénétration des sols, il est probable que l'effet de la remédiation ne soit visible qu'au bout d'un laps de temps plus long (Ampoorter *et al.*, 2011). De plus, l'humidité des sols amendés (témoin et tassé) était plus faible en juin 2011 que celle des sols non amendés, probablement en relation avec une consommation d'eau plus forte et/ou une amélioration des transferts hydriques. Par conséquent, logiquement, leur résistance à la pénétration était plus élevée que celle des sols non tassés. Normalement l'apport de cations devrait favoriser l'agrégation et diminuer la résistance à la pénétration (Ampoorter *et al.*, 2011). Il sera ainsi intéressant de continuer à comparer les traitements amendés avec les traitements non amendés, pour tester si la différence de résistance à la pénétration est seulement liée à une différence d'humidité ou si un autre processus intervient.

Contrairement à la mesure de densité apparente, le problème de la variabilité spatiale des constituants du sol ne se pose pas pour le suivi temporel de la résistance à la pénétration (Schéma 3). En effet, la mesure au pénétromètre est très peu destructrice et permet un suivi systématique « aux mêmes endroits » d'une année à l'autre. De plus, cela permet également de réaliser un grand nombre de mesures par traitement, site et année. Enfin, contrairement aux différentes campagnes de mesure de la densité apparente, les sols lors des différentes campagnes de mesure de la résistance à la pénétration ont été de plus en plus secs d'une année à l'autre (III). Ainsi, la résistance à la pénétration augmente depuis le début des mesures (2008). Ce biais dans les observations n'a pas pu être complètement corrigé en normalisant la résistance à la pénétration par rapport à l'humidité, car la variabilité de l'humidité propre à chaque campagne de mesure est faible par rapport à celle des teneurs en eau toutes campagnes confondues. La relation entre la résistance à la pénétration et l'humidité du sol n'étant pas linéaire, les relations calibrées pour chaque année, profondeur et traitement ne sont, de la sorte, que peu extrapolable à l'ensemble des teneurs en eau. Par conséquent, l'effet du facteur temps depuis la compaction est encore probablement quelque peu corrélé à la variation de l'étendue des teneurs en eau explorée chaque année.



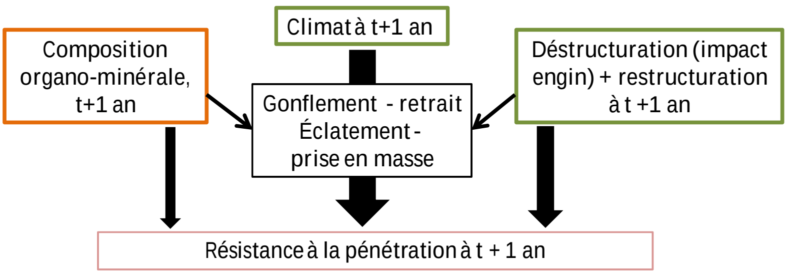


Schéma 3 Facteurs pouvant influencer la mesure de résistance à la pénétration et la détection de sa dynamique temporelle. Les couleurs des cadres de deux des trois facteurs identifiés sur les sites étudiés (composition organo-minérale, climat et déstructuration/ restructuration) sont différentes à t et t+1 an pour indiquer de probables différences dans l'état de ces trois facteurs. L'intérêt de la mesure de la résistance à la pénétration est qu'elle peut être au « même endroit » d'une année à l'autre (même composition organo-minérale, même couleur de cadre). Dans ce cas, le facteur étudié dans cette thèse (déstructuration/ restructuration) et le facteur climat influencent l'indicateur résistance à la pénétration, d'où l'intérêt de normaliser les mesures mais la normalisation est très sensible à l'étendue de teneur en eau explorée.

Pour s'affranchir de ce problème inhérent aux études de terrain, il serait nécessaire de mesurer la résistance à la pénétration plusieurs fois chaque année, de préférence lors de la même saison car la structure du sol et donc la résistance à la pénétration varient en fonction de la saison (Bormann & Klaassen, 2008), en essayant de couvrir le plus grand spectre de teneur en eau possible. Il peut également être envisagé de contrôler la teneur en eau en couvrant la surface à mesurer. La dernière option implique de renoncer à une exploration à large échelle. La première option requiert un investissement plus grand. Il doit cependant être noté que l'observation sur sols secs traduit effectivement les contraintes à l'enracinement pouvant apparaître dans le traitement T en saison de végétation. Ainsi, suivre la résistance à la pénétration en s'assurant que les sols soient très secs au moment de la mesure est probablement suffisant pour caractériser l'impact maximal sur les conditions d'enracinement et l'évolution de cet impact.

Une restauration du paramètre densité apparente a été mise en évidence dans la couche de surface (0–10 cm) des deux sites (II). Même si à CA cette restauration ne se traduit qu'à l'état humide sur la densité apparente, il est également probable qu'une restructuration du sol par des phénomènes de gonflement – retrait n'ait pas le même effet sur la densité apparente et sur la résistance à la pénétration. En effet, Horn (2004) a démontré que les cycles humectation – dessiccation étaient à l'origine de réorientations des particules de sol et ainsi d'une porosité plus stable se traduisant par une amélioration du fonctionnement hydrique. La dispersion des particules argileuses et des colloïdes, ainsi que le réarrangement au sein même des agrégats, dus aux forces exercées aux interfaces eau – sol (ménisque), renforcent la résistance mécanique des agrégats tout en diminuant leur densité. De plus, il est probable que la fissuration liée au desséchement du sol se fasse selon les plans de plus faible résistance qui sont parallèles à la surface du sol (structure lamellaire causée par le tassement), ce qui est confirmé par l'analyse des lames minces de sol prélevées tous les ans en même temps que les échantillons de densité apparente (Nicolas Bottinelli, discussion personnelle). Ainsi la fissuration n'étant pas verticale, il est probable que la résistance à la pénétration mesurée sur un axe vertical, ne soit pas affectée par ce début de restauration des propriétés physiques. Pour pouvoir tester cette hypothèse, il eut fallu mesurer la densité apparente et la résistance à la pénétration du sol à plusieurs états d'humidité chaque année, de préférence durant la même saison pour limiter les effets de l'activité biologique sur la structuration du sol (Bronick & Lal, 2005).

Atmosphère du sol

La composition de l'atmosphère du sol notamment en CO_2 et O_2 s'est avérée être un indicateur très sensible au changement de structure du sol, que ce soit à cause de la contrainte appliquée par le porteur ou après tassement, du fait de la régénération naturelle de la porosité du sol (V).

Le suivi de l'atmosphère du sol a débuté trois mois après le passage du porteur à CA et un peu moins d'un an après le tassement à AZ. Même si ce ne fut que pendant quelques mois à AZ, nous avons pu capter un effet initial très fort du tassement sur la composition de l'atmosphère du sol en CO_2 (IV). Un effet initial similaire a également été observé à CA. Pour les deux sites, cet effet a duré jusqu'à la première période de sécheresse édaphique après le passage du porteur. Pendant cette première période, l'atmosphère du sol perturbé était beaucoup plus riche en CO_2 et moins riche en O_2 que l'atmosphère du sol témoin et ceci quelles que soient les conditions de diffusion (porosité libre à l'air). Après la première sécheresse, dans l'atmosphère du sol tassé, pour une même valeur de porosité libre l'air, la teneur en CO_2 était plus faible et la teneur en O_2 plus forte que pendant la phase initiale. Ceci traduit des conditions de diffusion améliorées alors que l'espace poral où se déroule cette diffusion reste inchangé (même porosité totale et même volume de porosité occupé par l'air). Ainsi, sur les données brutes de teneur en CO_2 et O_2 à CA, on peut observer une forte diminution de l'impact du tassement au cours du temps. Cette régénération des conditions de transfert des gaz, ne se traduit cependant pas sur la porosité libre à l'air. Par conséquent, nous avons émis l'hypothèse que, la structure du sol tassé étant plus instable, des fissures se sont formées lors de la première sécheresse édaphique après compaction. Ainsi même si la porosité libre à l'air est toujours inférieure dans le sol tassé à celle mesurée dans le sol témoin, les conditions de diffusion gazeuse se sont améliorées probablement en augmentant la connectivité des espaces poraux libres à l'air. Ce résultat est conforme à l'évolution de l'impact du porteur sur la densité apparente, même si le constat d'une évolution concernant la composition de l'atmosphère du sol intervient plus subitement, par comparaison avec l'évolution plus continue de la densité apparente. La diffusion gazeuse étant beaucoup plus sensible à un changement de structure que la densité apparente, il est normal qu'un début de fissuration influence différemment l'atmosphère du sol et la densité apparente. Li & Zhang (2009) ont montré que les processus d'humectation – dessiccation n'affectaient que l'espace poral intra-agrégats. D'après leurs résultats, la restauration de densité apparente constatée dans les premiers cm du traitement T des deux sites ne proviendrait pas d'une régénération du volume poral inter-agrégats. Cela pourrait expliquer un impact toujours significatif sur le paramètre « porosité libre à l'air » alors que les conditions de diffusion gazeuse pour une même valeur de porosité libre à l'air se sont améliorées depuis la première période de sécheresse édaphique (V).

La fissuration du sol dégradé n'a pas permis le retour aux conditions d'activité biologique et de transfert de gaz qui prévalaient avant le tassement du sol. En effet, on observe toujours un impact du porteur sur la teneur de l'atmosphère en O_2, CO_2, CH_4, et N_2O. Par exemple, à AZ de la seconde année à la quatrième année après tassement, l'impact du traitement était globalement non significatif sur la teneur en CO_2 et O_2 à cause de l'alternance de périodes où l'impact était positif et de périodes où l'impact était négatif. Par contre, en prenant en compte les covariables « porosité libre à l'air » et « température » du sol, j'ai pu mettre en évidence un effet toujours significatif du traitement. Ainsi, si on considère des conditions de diffusion et de production similaires de gaz, on peut toujours observer une diminution de la teneur en CO_2 et une augmentation de la teneur en O_2 dans le traitement T par rapport au traitement C, indiquant une production de CO_2 et une consommation de O_2 toujours réduites. A CA, on peut également observer ce phénomène après la fissuration du sol. L'environnement dans lequel se développent les racines n'est toujours revenu à son état initial concernant l'aération du sol et la composition de l'atmosphère en est un excellent indicateur. L'ensemble des observations et hypothèses concernant la dynamique de la teneur en CO_2 de l'atmosphère du sol est synthétisé dans le Schéma 4.

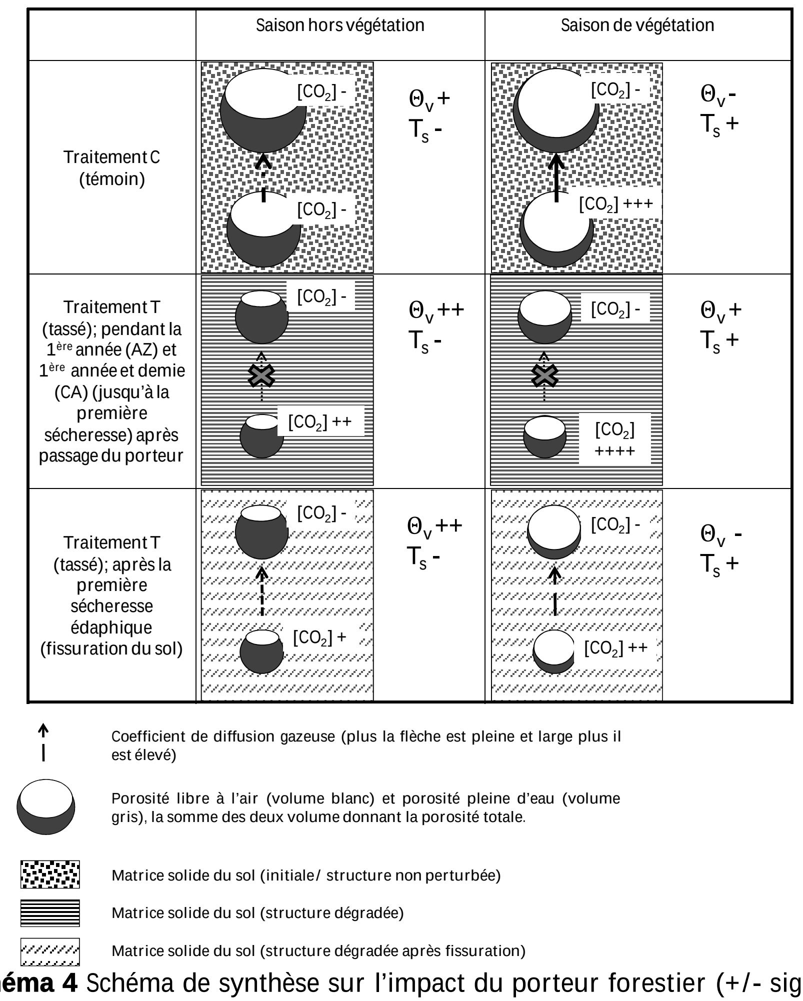


Schéma 4 Schéma de synthèse sur l'impact du porteur forestier (+/- signalant un impact positif/négatif dont l'intensité est représentée par le nombre de +/-) sur l'humidité volumique (θ_v) et la température (T_s) du sol et sur la composition de l'atmosphère du sol en CO_2 (pour O_2 inverser les signes), en fonction de la saison considérée et du moment après tassement.

RECOMMANDATIONS POUR LA GESTION DES SOLS SENSIBLES ET PISTES DE DEVELOPPEMENT

Les deux sites étudiés ont la particularité d'être très sensibles à la dégradation physique par les engins réalisant les opérations sylvicoles. En effet, la rupture texturale et la présence d'un substrat imperméable vers 50 cm de profondeur font que la couche limoneuse de surface est très souvent humide voire engorgée (traces d'hydromorphie d'intensité variable). Ainsi, même si la circulation du porteur n'a crée que des ornières peu profondes (en moyenne 5 cm, les ornières les plus profondes étant habituellement exclues des échantillonnages réalisés pour cette étude), son impact sur le fonctionnement du sol est élevé notamment en intensifiant l'anoxie et l'engorgement temporaire du sol. Sur les deux sites, l'amplitude des variations d'humidité a augmenté avec la circulation du porteur, augmentant la fréquence et la durée des périodes de stress hydrique, que ce soit en sécheresse (sol aussi, ou plus, sec que le témoin) ou en phase d'excès d'eau. De plus, le tassement a provoqué une diminution de la stabilité structurale sur le sol le plus limoneux et le plus acide (CA), ce qui a pour conséquence de diminuer l'étendue des humidités propices à la croissance racinaire (perte de structure du sol à la fois à l'état humide et à l'état sec). L'engorgement du sol provoquant une augmentation de la mobilité du fer, il est probable que la « perte », ou plus exactement la redistribution hétérogène des oxydes de fer (agent structurant) va limiter le retour vers un état structural stable et propice au bon enracinement du peuplement. Ainsi, il est primordial de protéger ces sols sensibles de toute dégradation physique en choisissant : i- le matériel adapté pour les opérations sylvicoles, ou un moyen alternatif pour les sols les plus sensibles ; ii- les conditions propices à la circulation des engins (sols secs et/ou gelés) ; et iii- un réseau de cloisonnement adapté (*cf.* guide PROSOL, Pischedda, 2009) et une interdiction formelle de circuler hors de ces pistes. Il est à noter que le sol tassé est toujours plus humide (humidité volumique) que le sol non perturbé, ce qui pose problème dans le cas de cloisonnements qui sont régulièrement utilisés par des engins, et qui doivent rester praticables. Dans ce cas, un travail du sol sur le cloisonnement peut être une solution en fonction du bilan économique de l'opération. Des simulations de déformation d'un sol donné (texture, teneur en carbone et humidité) par un matériel donné (*e.g.* poids, caractéristiques des pneumatiques) permettent de choisir l'engin adapté ainsi que les conditions climatiques adéquates pour un impact minimal. Cependant, il est nécessaire de garder à l'esprit qu'une variation de densité apparente n'est pas le seul critère à considérer, en particulier dans le cas de sol à tendance hydromorphe où une faible augmentation de densité apparente peut conduire à une augmentation conséquente de l'engorgement temporaire.

Le choix du type de mécanisation, ou d'alternative, à utiliser pour les opérations forestières doit intégrer les dégâts au sol, car même si leurs conséquences sur le peuplement ne sont pas immédiates, elles peuvent diminuer fortement la résilience des peuplements face aux stress (hydriques, tempêtes) qui pourraient augmenter de fréquence avec les changements climatiques en cours. Par exemple, les plants de chênes sessiles du traitement T à CA ont eu un meilleur taux de reprise que ceux du traitement T à AZ. Il est probable que le premier printemps des plants de CA (2009) ayant été plus sec que celui des plants de AZ (2008), la meilleure réserve utile du traitement T a permis de compenser la moins bonne qualité structurale par rapport au traitement C à CA. Cependant, au vu de nos résultats, le développement racinaire des plants à CA a de fortes chances d'être contraint dans le traitement T, ne serait-ce que suite à l'été 2011 où la résistance à la pénétration mesurée dans le traitement T a probablement fortement limité la poussée racinaire. Les conséquences à court et à long terme sur les peuplements forestiers ne sont pas les seules à prendre en considération lors du choix du type de mécanisation et des conditions d'intervention. Il ne faut en effet pas oublier les autres fonctions du sol, *e.g.* qualité de l'eau, de l'air, biodiversité, érosion.

Caractériser l'état du sol après le passage d'un engin n'est pas toujours évident, notamment dans le cas de sols peu stables comme à CA. De plus, il est souvent nécessaire de mesurer différentes propriétés du sol pour quantifier toutes les conséquences que peut avoir le passage d'un engin (aération, régime hydrique, pénétrabilité, activité biologique, biodiversité, cycle des nutriments) et réaliser un diagnostic correct. Par conséquent, il apparaît nécessaire de développer des indicateurs simples de qualité du sol, à l'aide d'études détaillées. Ces connaissances de base seront ensuite transcrites en indicateurs adaptés pour aider les gestionnaires forestiers à quantifier l'impact de leurs choix de gestion sur la productivité potentielle et la durabilité des peuplements face aux différents stress.

Parmi les pistes de développement les plus prometteuses, le type d'ornière est un indicateur de la dégradation physique sous le poids d'un engin forestier simple et peu couteux à mesurer. En effet, Frey *et al.* (2009) ont mis en relation une typologie d'ornière proposée par le WSL (institut de recherche Suisse, Lüscher *et al.*, 2009) avec des conditions dégradées d'aération et de conductivité hydraulique à saturation, l'intensité de la dégradation par engorgement augmentant avec le type d'ornière. Ils ont également travaillé sur des sols sensibles à tendance hydromorphe qui peuvent être fortement dégradés (diminution de la conductivité hydraulique à saturation et de la perméabilité à l'air) même sous des ornières de faible profondeur (type 1 à 2). Cette typologie permettrait ainsi de caractériser l'état du sol après passage d'engin forestier de manière sensible et à faible coût. De plus, à CA comme à AZ, même si la végétation a colonisé les ornières, celles-ci restent parfaitement nettes trois et quatre ans après le passage du porteur, y compris dans le détail des crampons du pneumatique. Il serait par conséquent intéressant de développer le lien entre type d'ornière et paramètres du sol affectés pour différents types de sols. Les gestionnaires pourraient ainsi facilement caractériser l'état et l'évolution de la qualité de leur sol après passage d'engins à partir des typologies d'ornières rencontrées et du type de sol.

Un autre indicateur de la dégradation du sol sous le passage d'engins forestiers est le changement de flore qui peut également être facilement constaté par des gestionnaires forestiers. Dans les sites étudiés, le changement brutal de flore après tassement perdure trois ou quatre années après l'application de la contrainte, plus marqué d'ailleurs par un changement d'abondance que par un changement d'espèces, surtout à CA (IV et V). Cependant, l'impact de la dégradation de la qualité physique du sol sur le changement qualitatif et quantitatif de flore et sa durée reste encore à valider. En particulier, les valeurs indicatrices de la flore vis-à-vis des propriétés physiques des sols ont été encore peu testées. A ce sujet, il est probable que l'indication ne concerne que les horizons les plus superficiels, alors que par exemple à CA l'impact est plus fort de 10 à 20 cm que de 0 à 10 cm de profondeur.

Enfin, concernant les mesures de propriétés des sols, tous les indicateurs de caractérisation et de suivi de la dégradation physique de sol devraient être mesurés lors d'épisodes climatiques extrêmes qui en quelque sorte servent à tester la qualité de la structure du sol, paramètre déterminant du comportement du sol (Weisskopf *et al.*, 2010). Nous recommandons en particulier, des conditions sèches pour la mesure des propriétés physiques suivies dans cette étude (résistance à la pénétration et densité apparente) et des conditions humides pour les propriétés biogéochimiques (gaz).

L'intérêt des deux sites expérimentaux étudiés est de permettre de coupler des approches physiques, biologiques et chimiques pour quantifier l'impact du passage d'un engin classique d'exploitation forestière, en tenant compte des interactions entre les différentes conséquences du tassement d'un sol. Ce sont des dispositifs en vraie grandeur, capables de répondre à des questions de recherche, mais aussi permettant de conduire à des recommandations pour la gestion. Ces sites offrent notamment l'opportunité de développer et d'améliorer des modèles qui pourront aider les gestionnaires à planifier les différentes opérations sylvicoles de manière à préserver leurs sols. Un premier pas a été accompli en testant la validité d'un modèle analytique de compaction dans la prévision des déformations lors de la circulation d'un engin sylvicole. D'autres modèles comme la prévision de la teneur en eau du sol en fonction du type de sol et du climat permettraient d'organiser les différentes opérations sans avoir besoin de relever au préalable l'humidité, frein notable aux mesures de préservation des sols. Les sols de ces deux sites présentent une forte analogie physique et morphologique, associée à des différences chimiques notables, autorisant le contrôle de cette interaction sur leur sensibilité à la dégradation physique et leur dynamique après tassement.

Le suivi de propriétés physiques (densité apparente, résistance à la pénétration et humidité du sol) et chimique (composition de l'atmosphère) du sol a démontré l'intérêt de coupler les approches pour mieux comprendre l'impact de la déstructuration du sol sur une de ses principales fonctions, *i.e.* assurer un environnement propice au développement racinaire.

Sur les deux sites étudiés, le passage du même porteur forestier a augmenté la capacité de rétention en eau des sols tout en diminuant leur capacité de drainage à saturation. Il en résulte un accès des racines à une réserve hydrique plus conséquente mais un déficit d'aération couplé à un risque accru de déficit hydrique en cas de sécheresse, estivale en particulier. En effet même si les deux sols tassés sont plus humides une grande partie de l'année, lors d'épisodes de sécheresse ils sont autant, voire plus, secs que les sols non perturbés. Cependant il doit être noté que mon travail de thèse a porté sur deux sols très sensibles au tassement, de par leur texture et leur humidité élevée une grande partie de l'année. De plus, à cause de leur fonctionnement hydrique déjà perturbé par la présence d'une rupture texturale occasionnant la présence d'une nappe temporaire, ces sols sont aptes à se dégrader très rapidement. Ainsi, sur des sols à réserve utile très faible et/ou aucun problème de drainage n'apparaît, l'effet du tassement sur l'aération du sol peut être au moins compensé entièrement par l'augmentation de la rétention en eau. Il n'en reste pas moins que l'augmentation de densité apparente et de résistance à la pénétration peuvent réduire drastiquement l'enracinement et augmenter la sensibilité des peuplements à des stress (sécheresse, forte pluviométrie, tempête). Par conséquent, même si la croissance du peuplement peut ne pas être affectée par le tassement des sols, sa résilience actuelle ou future peut en être fortement altérée.

Nous avons également montré que la caractérisation de l'impact du passage d'engin à la surface du sol n'est pas évidente sur des sols à faible stabilité structurale comme à CA. Elle est particulièrement pertinente dans des conditions extrêmes : après une forte pluie pour les conditions d'aération et la diminution d'infiltration d'eau et/ou en période de sécheresse édaphique pour la compacité (résistance à la pénétration, densité apparente). De nouvelles techniques doivent être développées pour éviter les conclusions erronées lors du suivi de sols dégradés. En effet, étant donné les nombreux paramètres du sol affectés par le tassement et les nombreuses interactions entre ces paramètres, il est nécessaire d'identifier ces interactions et de normaliser l'indicateur suivi par rapport aux paramètres l'influençant pour assurer la comparabilité des études et des mesures d'une année à l'autre.

Tous les paramètres relevés sur les sites concordent quant à l'impact du porteur sur les sols des deux sites, même si chacun apporte un éclairage distinct sur la dégradation du sol, en raison de sa sensibilité spécifique. Pendant les trois (CA) et quatre (AZ) premières années après le passage du porteur, les paramètres physiques ont évolué en surface. A CA, cela se traduit par une différence entre les traitements T et C qui n'est plus significative à l'état humide mais qui l'est encore à l'état sec, alors qu'à AZ, la différence entre ces mêmes traitements est encore significative mais a été diminuée, que ce soit à l'état humide ou à l'état sec. Cette évolution des paramètres physiques en surface se traduit par une amélioration du transfert des gaz, avec un effet seuil étroitement lié à l'apparition de la première période de sécheresse édaphique après le passage du porteur. Ainsi, des comportements similaires en restauration (fissuration) sont constatés sur les deux sites malgré une plus faible teneur en argile à CA qu'à AZ, qui est probablement compensée par le caractère gonflant des argiles à CA et pas à AZ.

La restauration d'un sol tassé peut être accélérée sous des climats contrastés (typiquement sous climat continental), suite à des phénomènes de gel – dégel et d'humectation – dessiccation, même s'il présente peu d'évidence de bioturbation. En effet, la résistance à une augmentation de succion, lors du desséchement d'un sol dégradé par tassement, est faible ; sa capacité à se fissurer est donc augmentée. Cependant cette fissuration se fait essentiellement selon des zones de plus faible résistance mécanique, donc dans le cas d'un sol compacté selon un plan parallèle à la surface du sol (structure lamellaire). De plus, il est probable qu'elle se limite aux couches superficielles du sol. Le suivi à long terme des deux dispositifs étudiés devrait apporter des réponses sur la vitesse de restauration de sols acides où la bioturbation est limitée et la restauration est essentiellement liée à des phénomènes physiques. Le site de CA devrait également permettre d'éclairer le comportement d'un sol sujet à l'éclatement – prise en masse face à une restructuration liée aux cycles d'humectation – dessiccation. En effet, il semblerait que la restructuration du sol tassé de CA liée au gonflement – retrait ne fasse pas disparaître son comportement d'éclatement – prise en masse. Il est donc indispensable qu'un suivi soit réalisé sur le moyen terme (tableau 2) pour pouvoir identifier sans ambiguïté les indicateurs et les mécanismes de la restauration. Ces sites de recherches réalisés en conditions relativement contraintes, sont les seuls aptes à permettre cette avancée dans les connaissances, en particulier grâce à leur base de données multicritères et à l'opportunité offerte d'associer des équipes de recherche variées.

Tableau 2 Proposition de suivi multicritère pour les années à venir sur les sites de CA et AZ.

Variable suivie	fréquence	intérêt	protocole
Climat: -Méso	Suivi continu	Indicateur de climat local indispensable à toute investigation	Protocole actuel; les données sont enregistrées automatiquement par la centrale d'acquisition et à récupérer une fois par mois
-Pédo (humidité et température du sol)		Un changement de structure lié à la régénération du sol va se traduire sur les transferts d'eau et de chaleur dans le sol et donc sur la différence d'humidité et de température entre les traitements T et C	
Hauteur de nappe	Suivi soit continu (sondes diver) soit mensuel (mesures manuelles)	La restructuration potentielle du sol va également se traduire sur la différence de durée et d'intensité de l'engorgement entre les traitements T et C	Les données sont également enregistrées de manière automatique (sondes diver) et récupérées une fois par mois. Cependant elles doivent être corrigées car une dérive de la pression enregistrée par les divers a lieu de manière aléatoire et variable selon le mois et le diver. Les mesures manuelles réalisées une fois par mois sont peu informative étant donné la durée de présence de la nappe souvent très brève

Tableau 2 suite

Sol : *Paramètres physiques :* -densité apparente	La fréquence peut être diminuée; tous les 5 ans au lieu de tous les ans	L'évolution de la porosité totale traduit également un changement de structure	la forte sensibilité de la mesure de densité apparente à la variabilité spatiale des constituants du sol amène à privilégier une fréquence de prélèvement plus faible mais avec une analyse plus complète (mesurer au moins la teneur en carbone organique et en argile des échantillons, leurs masse et volume humide et leurs masse et volume après séchage à l'air). Les volumes sont mesurés grâce à la méthode du sac plastique (Boivin *et al.*, 1990)
-Résistance à la pénétration	Une fois par an, voire tous les deux ans	Indicateur des conditions d'enracinement et leur évolution	Mesures pendant la saison de végétation, quand les sols sont très secs pour déterminer l'impact de la régénération du sol sur la contrainte maximale à l'enracinement
-structure du sol	Fréquence à définir	Indicateur direct des variations de structure du sol (*e.g.* relation succion-teneur en eau, lames minces, conductivité hydraulique)	Protocole à définir une fois le travail de post-doctorat de N. Bottinelli terminé
-Résistivité électrique ou magnétique	Une fois tous les 5 ans	Indicateurs peu invasifs et sensibles aux variations d'humidité et de structure du sol	La calibration de cet indicateur est indispensable pour l'interpréter en sol organique, acide, saturé par Al (*cf.* thèse de Samir Seladji, 2010)
Paramètres géochimiques -Fluides : gaz et eau du sol	Suivi mensuel	Indicateurs géochimiques très sensibles à la dégradation et à la restauration physique du sol, ils permettent en outre de quantifier l'impact sur la qualité des eaux et de l'air	Prélèvements, *cf.* articles III, IV et V
-Chimie du solide	tous les 3 ans	Indicateur peu sensible, sauf peut-être pour la spéciation du fer	
Paramètre biologique : -microbiologie	Fréquence à définir	Indicateur de l'activité biologique potentielle et de son évolution après contrainte	Protocole à définir ; un travail préliminaire a été réalisé lors du stage de M1 de C.Cosson encadré par A.Poszwa
-faune du sol		Indicateur, entre autres, du potentiel de restauration biologique de la structure du sol	Protocole à définir une fois le travail de post-doctorat de N.Bottinelli terminé

Tableau 2 suite

Végétation : -Flore	Une fois par an	Indicateur global de modifications des propriétés physique, biologiques et chimiques	La flore est toujours affectée par le passage du porteur et le changement n'est pas identique dans les deux sites. Il sera ainsi intéressant de suivre le retour à un état non perturbé s'il a lieu pour déterminer les variables du sol qui expliquent le mieux ce retour
-Plantations de chênes sessiles	Croissance, une fois par an	Suivre l'impact de la dégradation physique du sol et de sa restauration sur la productivité du peuplement. Est-ce que les modifications de qualité physique du sol se traduisent sur la hauteur dominante, indicateur de la fertilité de la station ou seulement sur le risque face aux aléas naturels?	Pour l'instant le suivi est en plein, jusqu'à la fermeture du couvert au moins
	Système racinaire, fréquence encore à définir	Indicateur de stress potentiel	Protocole à définir quand le projet réalisé par B.Fatré sera terminé
	État nutritif, une fois par an	Indicateur intégrant le statut de l'arbre	Protocole actuel (prélèvements foliaires)

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SOIL COMPACTION DUE TO HEAVY FOREST TRAFFIC: MEASUREMENTS AND SIMULATIONS USING AN ANALYTICAL SOIL COMPACTION MODEL

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Abstract

Models for predictions of soil compaction following forest traffic represent important decision tools for forest managers in order to choose the best management practices for preserving soil physical quality. In agricultural soil compaction research, analytical models are widely used for this purpose. Our objective was to assess the ability of an analytical model to predict forest soil compaction under forwarder traffic. We used the results from two experimental sites set up in north-eastern France in 2007 and 2008 to compare simulations using the *SoilFlex* model with observed bulk density following forwarder traffic. The best model-based predictions were found when considering the mean initial soil conditions and an increased rebound parameter in the upper soil layers (0-10 cm) in comparison to the deeper layers (10-50 cm). The need to increase the rebound parameter in the soil surface layer to improve model accuracy was attributed to a high soil organic matter content in the uppermost layers of forest soils. For the site where initial soil mechanical parameters were measured as a function of soil bulk density and water content, the model performance was good, with a root mean square error (RMSE) of 0.06. The model performed poorer (RMSE of 0.11), especially for the surface soil layer, for the second site that was wetter at the time of traffic and where soil mechanical properties were not measured but estimated by means of pedo-transfer functions. *SoilFlex* was found to yield satisfactory predictions, and could help forest managers estimate the risk of compaction of various forest management operations, and to select the most appropriate machinery for given soil conditions in order to preserve the soil from physical degradation during traffic in forest ecosystems. However, our study shows that simulation outputs are sensitive to the input of soil mechanical parameters, and calls for an urgent development of pedo-transfer functions for soil mechanical properties. Furthermore, our results emphasize the need for research on soil mechanical properties of forest soils, in particular on the role of soil organic matter and roots on soil compressive properties.

1. Introduction

Nowadays, forest managers have to harvest more wood with limited manpower available. Therefore, an increase in mechanization of forest operations is unavoidable. However, this poses problems in terms of soil sustainability. Heavy forest traffic has been proved to cause soil physical degradation, resulting in changes in soil chemical and biological properties (Greacen and Sands, 1980, Horn et al., 2007).

The prevention of soil physical degradation is needed in forest ecosystems as tillage and/or fertilization (as in agriculture) are missing in forest management practices. Soil compaction simulation models (Défossez and Richard, 2002) are important decision tools for prevention of excessive soil compaction following heavy traffic.

Forest soils differ from agricultural soils by their organic matter content and quality, by porosity, acidity, root and gravel content and heterogeneity. Therefore, different mechanical behaviours may be expected. Root networks increase soil resistance to deformation under heavy load (Soane, 1990; Cofie et al., 2000). However, assessment of several parameters not easily accessible to forest managers (e.g. rooting pattern, root stiffness and strength) is needed to take into account the soil reinforcement effect of roots in compaction modelling (Cofie et al., 2000). The presence of forest floor litter, the increased soil organic carbon (SOC) content and the modified organic matter quality may decrease the compactibility of forest soils in comparison to agricultural soils (Soane, 1990). However, the effect of SOC on soil compressibility has been shown to depend on soil water content at the time of loading (Soane, 1990; Pereira et al., 2007). Blanco-Canqui et al. (2005) found lower soil strength and higher SOC content in forest soils than in pasture and cultivated soils. They also observed that increasing SOC increased water retention and decreased soil strength, both increasing soil susceptibility to compaction. Yet

Mosaddeghi et al. (2000) and Williamson and Neilsen (2000) observed that an increase in SOC content lead to an increase in critical moisture content, i.e. the soil water content at which the maximum deformation is achieved for a given stress. Mosaddeghi et al. (2000) stated that increasing SOC content allowed the soil to better withstand loading especially at high water content. However, Pereira et al. (2007) found the opposite.

Keller et al. (2007) developed a model named *SoilFlex* that enables calculations of soil stresses and compaction due to agricultural field traffic. *SoilFlex* proposes different analytical procedures to calculate the soil-tyre contact area, the normal stress distribution at this interface, the vertical stress propagation through the soil profile and the resulting soil deformation (soil displacement, rut depth and bulk density). It is easy to use, requires a limited amount of parameters and also includes several pedo-transfer functions to estimate soil mechanical parameters from easily collectable soil parameters like bulk density, texture and water content. Models based on analytical procedures have been validated for agricultural soils (Défossez and Richard, 2002; Keller et al., 2007). Therefore, *SoilFlex* could represent a possibility for helping forest managers to choose the harvesting equipment that is the least likely to cause damage to the soil. However, the model requires validation for forest soils.

Our objectives were:

1. To simulate soil compaction in forest soils due to forwarder traffic by means of an analytical soil compaction model.

2. To analyze the effect of variability in soil water content and bulk density on soil compaction simulations.

3. To evaluate the impact of organic carbon on bulk density values after wheeling and on model predictions.

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2. Material and methods

2.1 Study sites

Two experimental sites to study the effects of soil compaction have been set up in Lorraine (NE part of France). They are located in the “Hauts-Bois” forest - Azerailles (48° 29′ 19″ N, 6° 41′ 43″ E), Meurthe et Moselle, and in the “Grand Pays” forest - Clermont en Argonne (49° 06′ 23″ N, 5° 04′ 18″ E), Meuse. The sites of Clermont en Argonne (CA) and Azerailles (AZ) have an elevation of 270 m and 300 m, respectively. The climate of the region is characterised by a 30-year mean annual temperature of 9° C (AZ) to 9.5 °C (CA) and a 30-year annual precipitation of 900 mm (AZ) to 1000 mm (CA).

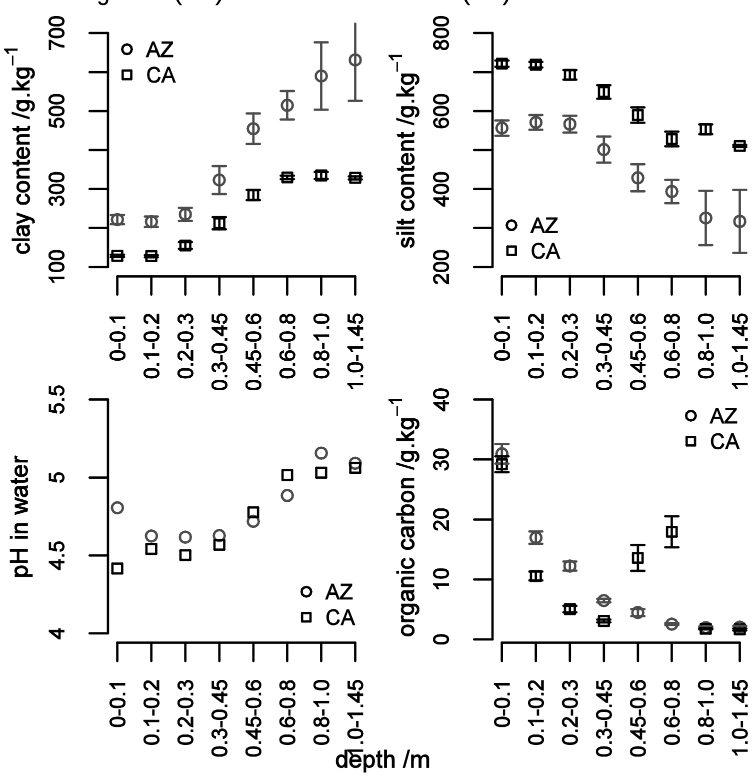
Selected soil properties of the study sites are presented in Fig. 1. The soil of both sites is classified as a neoluvisol (ruptic) (IUSS Working Group WRB, 2006) and is developed on a silt loam layer approximately 50 cm thick laying on a clayey layer (weathering of a Keuper marl for the AZ site and weathering of gaize rock for the CA site). This strong textural discontinuity caused limited temporary water logging; at the lower part of the silt loam layer hydromorphic figures could be observed.

2.2 Experimental treatment

Each site was clear-cut and timber (*Fagus sylvatica*, *Quercus petraea* mainly) was extracted using a cable yarding system to avoid soil damage. All remaining slash was removed by hand or using an iron horse to avoid soil compaction and/or sinking of the slash into the soil during traffic. Afterwards, an 8-wheel drive forwarder (1996 Valmet 840, serial number 9146, Valmet Logging, Sweden) drove on the soil for an equivalent of two passes (one forward and one rearward pass), in May 2007 at AZ and in March 2008 at CA. The wheel tracks were adjacent to each other in order to create an equally levelled compacted area of 30 m × 50 m. In each of the three blocks designed to account for spatial heterogeneity, land strips of the same surface area (30 m x 50 m) adjacent to the trafficked plots (T) remained undisturbed by the forwarder and were considered as control plots (C).

The tyres of the forwarder were 60 cm wide, had a diameter of 133 cm (600/55 × 26.5) and were inflated to a pressure of 360 kPa for both sites. The empty forwarder weighed 11.4 Mg, the four front wheels supporting 6.9 Mg and the four rear wheels supporting 4.5 Mg. In AZ, the wood-loaded forwarder weighed 23.3 Mg, the four front and the four rear wheels supporting 7.56 Mg (i.e. the empty weight on the four front wheels + 5% of the load) and 15.76 Mg (i.e. the unloaded weight on the four rear wheels + 95% of the load), respectively. In CA, we only weighed the wood load and deduced the total weight of the loaded forwarder (16.7 Mg), the loaded weight on the four front wheels (7.17 Mg) and rear wheels (9.57 Mg) according to the measurements taken in AZ.

Fig. 1. Selected soil characteristics (mean and standard error as error bars). 13 replicates in Clermont en Argonne (CA) and 18 in Azerailles (AZ)



2.3 Measurements of soil compaction

2.3.1 Initial state

For each soil layer, we measured soil gravimetric water content (WC in g $100g^{-1}$) and bulk density (BD in g cm^{-3}) to describe precisely the physical state of the soil before loading, which determined the soil mechanical behaviour.

Before setting up the experimental sites (in 2006 for both sites, under previous forest cover), 18 and 13 pits were sampled to determine BD using steel cylinders (10 cm height and 5 cm diameter) in AZ and CA, respectively. These pits were spread over the entire site area that was not disturbed by the following forwarder traffic (control). Soil bulk density was calculated as the mass of the oven-dried (105 °C over 72 h) soil sample divided by the volume of the cylinder. Soil bulk density was corrected for the rock fragment content, i.e. the volume and mass of > 2 mm soil fragments were subtracted from the soil volume and dry mass, respectively.

The day the forwarder drove on the soil, initial WC was measured in the area where the soil was going to be disturbed by heavy traffic at 28 locations × five depths in CA (0-10, 10-20, 20-30, 30-40 and 40-60 cm) and at 32 locations × three depths (0-10, 10-30 and 30-50 cm) in AZ.

2.3.2 State following the forwarder traffic

Within one month after the forwarder traffic, two pits per block (3 blocks per site) were dug in the trafficked area. In each pit two soil samples per depth were collected to measure BD. We used 250 cm^3 steel cylinders to collect undisturbed soil samples every 5 cm at 0-40 cm depth. During sampling we avoided the few very deep ruts accompanied with bulges at the edges that were formed by wheeling on the wettest soil conditions (plastic deformation; Williamson and Neilsen, 2000; Ampoorter et al., 2010).

2.4 Model simulation

2.4.1 Vertical stress distribution at the soil-tyre interface

One of the most important factors determining the performance of soil compaction models was found to be the upper boundary conditions, i.e. the soil-tyre contact area and the vertical stress distribution at this surface (Keller et al. 2007; Keller and Lamandé, 2010). Consequently, we estimated the contact area and the distribution of normal stresses at the soil-tyre contact area using the model proposed by Keller (2005) that is incorporated in the *SoilFlex* model. We did all the simulations taking into account two load steps with four wheels (two front and two rear wheels), in order to mimic the stress applied in the field experiment.

2.4.2 Vertical stress distribution in the soil profile

To simulate the propagation of the vertical major stress σ_1 through the soil, the *SoilFlex* model uses the work of Söhne (1953). According to Défossez et al. (2003), we chose a concentration factor of 6, because our soils had a low density and were moist.

2.4.3 Soil deformation

Calculations of soil deformation require soil mechanical parameters that characterize the stress-strain relationship. We chose the stress-strain relationship developed by O'Sullivan and Robertson (1996) and proposed in the *SoilFlex* model. The virgin compression line (VCL), and the recompression line (RCL) are given as (O'Sullivan and Robertson, 1996):

$$VCL: v = N - \lambda_n \ln(p) \quad (1)$$

$$RCL: v = v_{init} - \kappa \ln(p) \quad (2)$$

where: v, soil specific volume (dimensionless) = ρ_s/BD; ρ_s, density of solids (g cm^{-3}); p, mean normal stress (kPa); N, specific volume (dimensionless) at p = 1 kPa; λ_n,

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compression index (kPa^{-1}); ν_{init}, initial specific volume (dimensionless); κ, recompression index (kPa^{-1}).

The mechanical parameters were measured by Saffih-Hdadi et al. (2009) using oedometer tests on soil samples from the 10-30 cm soil layer of the AZ site. The samples were equilibrated at three WC (0.31, 0.25 and 0.2 g g^{-1}) and compressed to three initial BD values (1.1, 1.3 and 1.45 g cm^{-3}) before undergoing oedometer tests. For the AZ site, we used the data of Saffih-Hdadi et al. (2009) to determine the soil mechanical parameters N and λ_n as functions of BD and WC. We used the same relationships for the CA site, because soil texture and SOC were similar to AZ (SOC, clay and silt content between 0.9-3.3, 16-36, 40-64% respectively for the AZ site and between 0.3-1.6, 11-23, 59-74%, respectively for the CA site). For both sites, the mechanical parameters N and λ_n were estimated as functions of the initial measured values of BD and WC (see section 2.3.1) for each discriminated soil layer. We chose to perform *SoilFlex* simulations on the medium and extreme initial conditions (Tables 1 and 2), i.e.:

- at the mean BD found before compaction associated with the mean WC during compaction of the soil layer considered [case A],

-at the highest BD value found before compaction associated with the lowest WC value during compaction [case B] and

-at the lowest BD value found before compaction associated with the highest WC value during compaction [case C].

Unfortunately, Saffih-Hdadi et al. (2009) did not measure the rebound and recompression index (κ) that is needed in the model of O'Sullivan and Robertson (1996). We estimated κ in two ways. 1) By using the pedo-transfer function of O'Sullivan et al. (1999) incorporated in the *SoilFlex* model, which is given as (O'Sullivan et al., 1999):

$$\kappa = \lambda_n \left[0.119 - \left(\frac{0.082w}{17}\right) \right] . \qquad (3)$$

And 2), by calculating κ as:

$$\kappa = \frac{1}{3}\lambda_n \,, \tag{4}$$

which was suggested by O'Sullivan and Robertson (1996) for wet soils. With these two approaches, we could evaluate the impact of κ on the simulated final (i.e. after wheeling) BD.

2.4.4 Model validation

Model validation was done by comparing simulated BD with measured BD. To be as close as possible to the field conditions where the driver of the forwarder created adjacent wheel tracks, we considered the simulated values across the width of the tyre, i.e. the 30 cm on either side of the wheel centre. For each 5 cm depth increment we calculated the mean, minimum and maximum values of the predicted BD after wheeling across the wheel width.

The two replicates of BD measurements per depth and pit described in section 2.3.2 were taken at each side of the pit front (approximately one meter width). Therefore, the probabilities to have sampled at a given distance from the wheel track centre were considered as equal. By considering the *SoilFlex* predictions on this 60 cm wide stripe we assumed to have a modelled population as close as possible to the observed population.

2.5 Analysis of the effect of SOC on BD

To analyse the effect of SOC on BD of the undisturbed and trafficked soils, we sampled 10 randomly distributed soil samples per block × treatment × site from the 0-10 cm soil layer using 500 cm^3 steel cylinders in April 2009. We sampled the 30 × 50 m areas of each treatment (undisturbed and trafficked) per block. After determining the field-moist sample weight to calculate WC, a sub-sample was taken to measure the BD as described in the previous section. The rest of the soil sample was air-dried and ground (sieve size of 250 µm) for determination of total carbon

(C) and nitrogen (N) content using a CHN apparatus (ThermoQuest CE Instrument NCS 2500). The soil did not contain any carbonate; therefore, total carbon corresponds to the SOC content. The effect of SOC or C/N ratio, WC and treatment on BD was assessed through an analysis of covariance using the R software (version 2.11.1). The best model was selected after checking on the distribution of the residuals and by looking for the maximum coefficient of determination (R^2) and minimum residual standard error (RSE).

3. Results

3.1 Variability in initial BD and WC, and observed soil compaction

The variability in BD and WC before heavy traffic was high (Tables 1 and 2) for both sites. For example at 0-10 cm depth of each site, the maximal WC value was approximately twice as high as the minimal WC. This high variability in soil physical properties may have had an important impact on soil deformation following the forwarder traffic. For both sites, we found that BD was significantly correlated with WC at the sampling time (AZ: Pearson's product-moment correlation of -0.47, t = -7.5, df = 194, p-value = 2.3e-12; CA: Pearson's product-moment correlation of -0.58, t = -8.8, df = 154, p-value = 3.3e-15). Consequently, to take into account the effect of the variability in initial soil conditions on soil deformation, we considered the fact that WC and BD were not statistically independent. For example, the lowest value of BD before traffic was associated with the highest value of WC during traffic (Tables 1 and 2).

The soil of both sites was considered as highly sensitive to compaction, which was verified *in situ*. Despite the high variability in BD values, the effect of the forwarder traffic on BD was notable for both sites (Fig. 2).

Table 1 AZ initial soil parameters used for model simulations. Soil gravimetric water content (WC in g 100g^{-1}) measured the day the forwarder drove on the site on 32 sampling locations spread over the trafficked area and soil bulk density (BD in g cm^{-3}) values determined on 18 pits spread over the undisturbed area were used to calculate λ_n and N according to Eqs. 5 and 6

	case A (mean initial density)				case B (max. initial density)				case C (min. initial density)			
depth /cm	WC	BD	λn	N	WC	BD	λn	N	WC	BD	λn	N
0-10	33.9	0.97	0.25	3.14	23.9	1.12	0.23	3.09	50.5	0.78	0.27	3.08
10-30	27.3	1.23	0.18	2.74	21.9	1.5	0.1	2.31	31.9	0.88	0.29	3.41
30-50	26.6	1.35	0.14	2.5	19.9	1.6	0.07	2.14	30.9	1.16	0.2	2.8

Fig. 2. Soil bulk density measured before (Control) and immediately after heavy traffic (Trafficked). 18 and 12 replicates per depth in the AZ site and 13 and 12 replicates per depth in the CA site for Control and Trafficked, respectively. The grey shaded areas and the bold line in the middle correspond to the range and to the mean of the measured bulk densities respectively

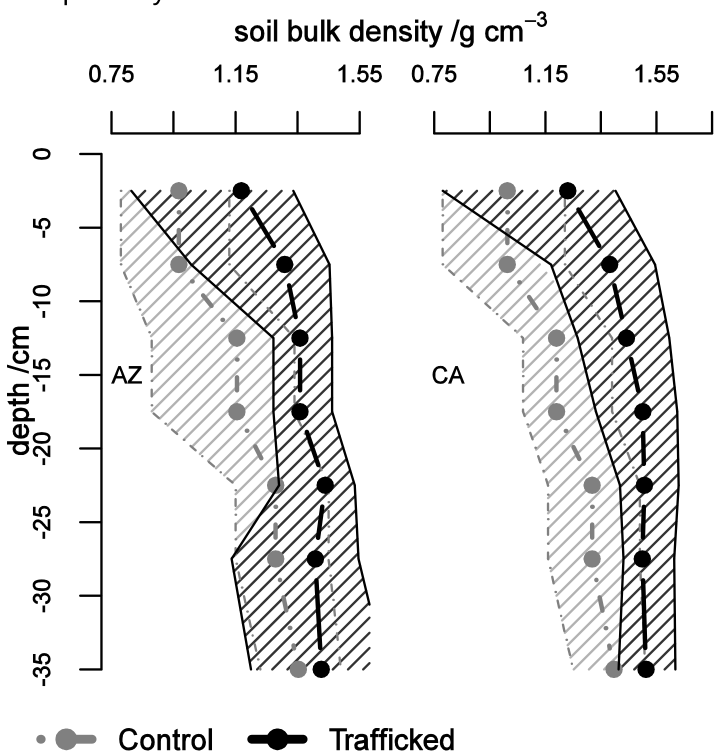

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Table 2 CA initial soil parameters used for model simulations. Soil gravimetric water content (WC in g $100g^{-1}$) measured the day the forwarder drove on the site on 28 sampling locations spread over the trafficked area and soil bulk density (BD in g cm^{-3}) values determined on 13 pits spread over the undisturbed area were used to calculate λ_n and N according to Eqs. 5 and 6

	case A (mean initial density)				case B (max. initial density)				case C (min. initial density)			
depth /cm	WC	BD	λn	N	WC	BD	λn	N	WC	BD	λn	N
0-10	48.5	1.01	0.20	2.62	32.3	1.22	0.17	2.63	66.6	0.78	0.23	2.59
10-20	34.8	1.20	0.17	2.59	25.3	1.39	0.13	2.45	44.2	1.07	0.19	2.61
20-30	28.6	1.34	0.14	2.47	16.7	1.49	0.12	2.48	36.8	1.05	0.22	2.7
30-40	28.6	1.40	0.11	2.33	23.2	1.55	0.08	2.16	46.1	1.18	0.14	2.30
40-50	25.8	1.43	0.11	2.34	13.6	1.57	0.10	2.38	30.1	1.27	0.16	2.58

3.2 Determination of mechanical parameters as a function of BD and WC

Using the data of Saffih-Hdadi et al. (2009), we found the following relationships between N and λ_n, respectively, and BD and WC (coefficient of determination of 0.97):

$$\lambda_n = 0.699 - 0.3575 \times BD - 0.0029 \times WC \quad (5)$$

$$N = 6.324 - 2.249 \times BD - 0.0296 \times WC \quad (6)$$

Using these relationships, we took into account the variability in initial WC and BD in model simulations (Tables 1 and 2), and analysed its effect on soil deformation.

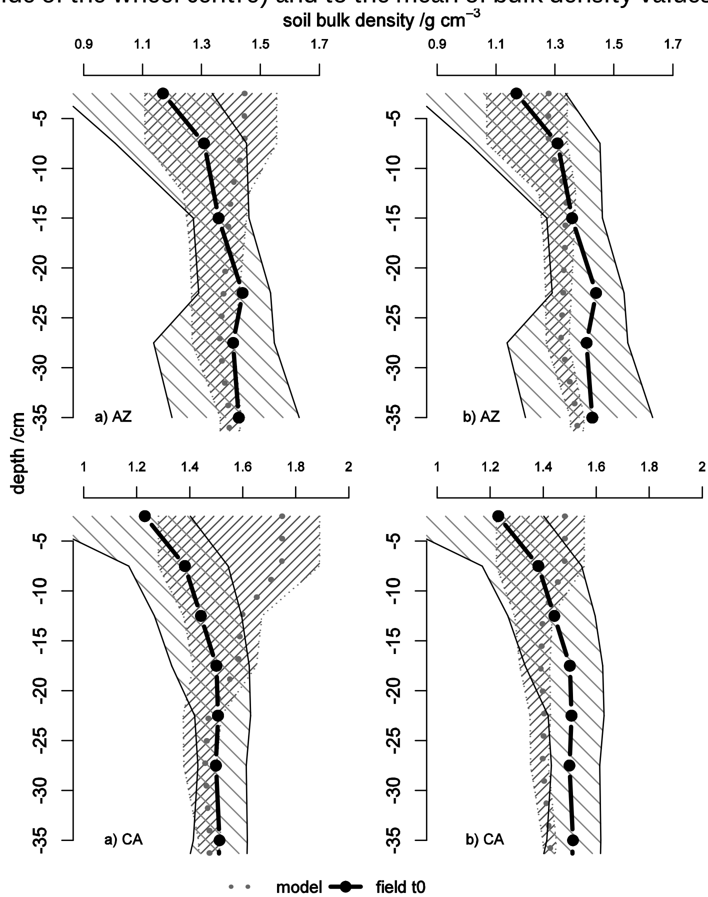
3.3 Stress simulation

The mean contact area modelled after the passage of four wheels (two front and two rear wheels) was 0.35 m² for both sites. The mean vertical contact stress was 156 and 153 kPa with a maximum value of 423 and 413 kPa for the AZ and CA sites, respectively.

3.4 Effect of the rebound parameter on model estimations

The model overestimated BD for the 0-20 cm soil layer when using κ according to Eq. 3 (Fig. 3a). The mean of the simulated BD values for the 0-20 cm soil layer was of 1.67 and 1.43 g cm^{-3}, with one of the observed values being of 1.39 and 1.30 g cm^{-3} for the CA and AZ sites, respectively. From 20 to 50 cm depth, the impact of the forwarder traffic on BD was slightly underestimated by the model. In the CA site, this tendency was larger: the differences between simulated and measured values were greater in comparison to the AZ site.

Fig. 3. Soil bulk density measured immediately following heavy traffic (field t0) and modelled considering mean initial conditions (case A, see text for details). a) simulations using Eq. 3; b) simulations using Eq. 4. The grey shaded areas and the bold line in the middle of the shaded area correspond to the range (min and max across the tyre width, i.e. 30 cm on each side of the wheel centre) and to the mean of bulk density values respectively



The model-based estimations of BD were closer to the measured values for the surface layers of both sites when applying κ according to Eq. 4 (Fig. 3 b). The improvement was particularly evident for the CA site. For the 0-5 cm soil layer, the difference between the mean of simulated and observed values decreased from 0.52 to 0.25 by using Eq. 4 instead of Eq. 3 for the CA site, whereas it decreased from 0.28 to 0.11 for the AZ site. For both sites, using Eq. 4 instead of Eq. 3 decreased the overestimation in the surface layer, but increased the underestimation in the deeper soil layers.

3.5 Effect of the variability in initial BD and WC on model-based estimations

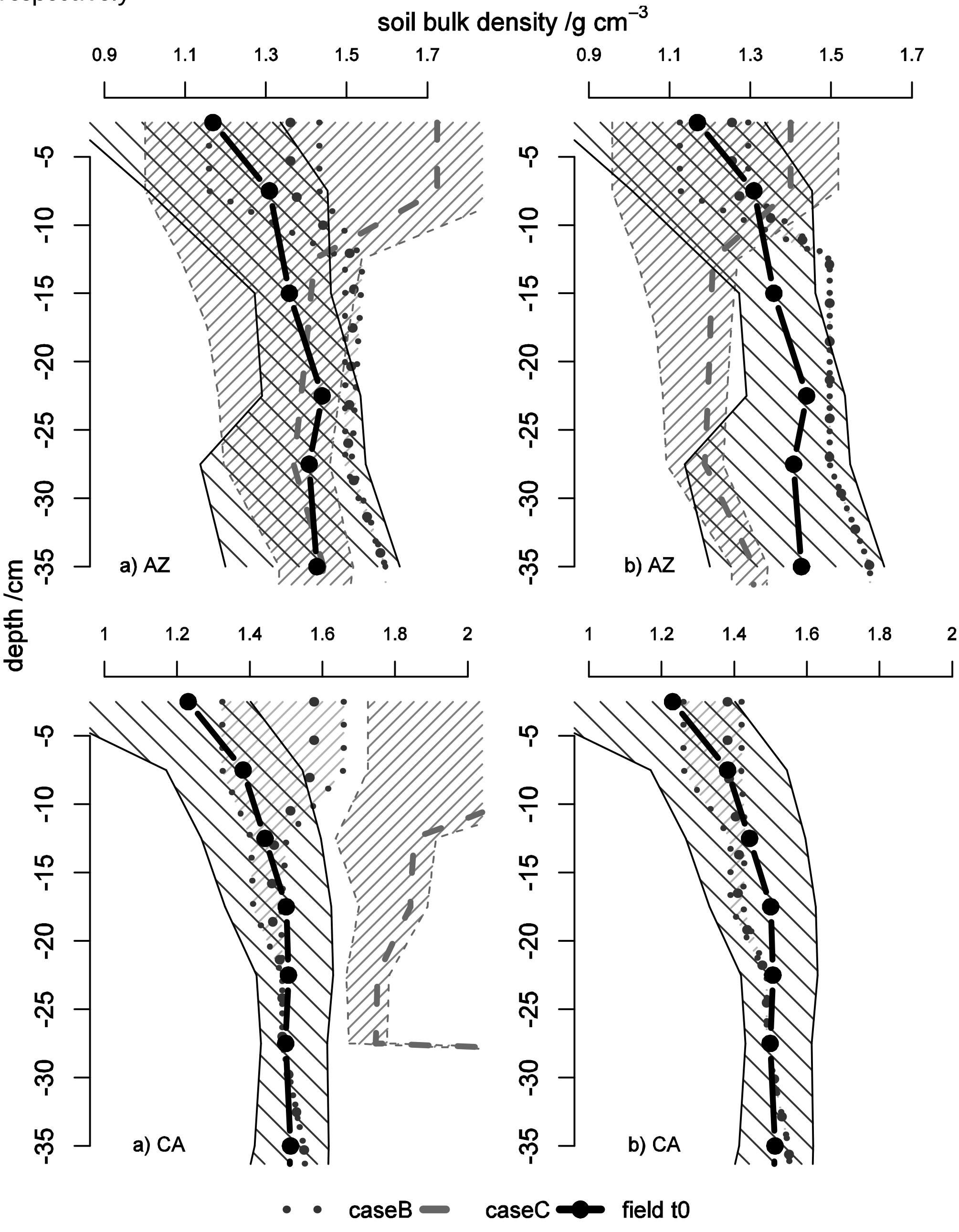
Increasing initial WC and decreasing initial BD (case C) increased the range of BD values simulated after heavy traffic (Fig. 4). Yet simulations never yielded as low values as we observed in the soil surface layer.

For the AZ site, the simulations in case C predicted higher BD in the upper soil layers and smaller BD in the deep soil layers in comparison to case B (minimum initial WC and maximum initial BD*)*. Bulk density predicted in case C using κ according to Eq. 3 was close to the observed values for depths below 20 cm.

For the CA site the simulations in case C yielded unreasonably high BD (Fig. 4a), and no deformation could be calculated using Eq. 4 in case C (Fig. 4b). For the CA site, soil mechanical parameters were not measured but estimated based on the relationships developed for AZ, as noted in Section 2.3.3. The initial WC was considerably higher for case C for CA than for any cases for AZ (Tables 1 and 2), which probably yielded inaccurate estimates of λ_n and N, and therefore inaccurate predictions of BD.

For the AZ site in case B and using Eq. 3, the model overestimated BD throughout the entire soil profile, whereas for the CA site the overestimation was restricted to the upper soil layers. For both sites in case B, using Eq. 4 improved model estimations in the upper soil layers, while the effect of κ on BD was marginal for depths below 20 cm.

Fig. 4. Soil bulk density measured immediately following heavy traffic (field t0) and modelled considering min and max initial conditions (case B and C, see text for details). a) simulations using Eq. 3; b) simulations using Eq. 4. The grey shaded areas and the bold line in the middle of the shaded area correspond to the range (min and max across the tyre width, i.e. 30 cm on each side of the wheel centre) and to the mean of bulk density values respectively



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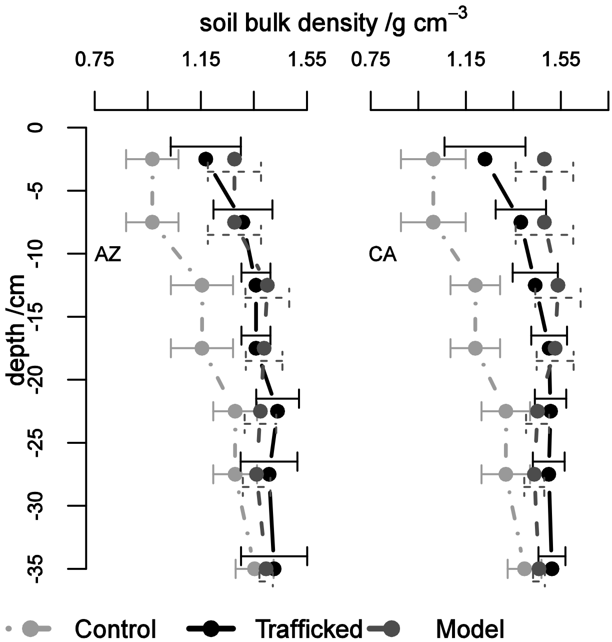
3.6 Model performance

To assess the performance of *SoilFlex* in predicting soil compaction under forwarder traffic, we considered on the one hand the predictions from case A and on the other hand the mean of the simulated values from the three cases A, B and C. Based on the results presented in Fig. 3, we calculated soil deformation by using Eq. 4 for the 0-10 cm soil layer and by using Eq. 3 for the 10-50 cm soil layer (Fig. 5). That is, we used a higher rebound in the uppermost soil layer. Comparison of the predicted values to the observed ones yielded a root mean square error (RMSE) of 0.06 and 0.11 for the AZ and CA site, respectively. The lower performance of the model for the CA site was mainly due to the overestimated impact of the forwarder traffic in the surface soil layer even when using Eq. 4.

Fig. 5. Model-based predictions considering mean initial conditions (case A, see text for details), two passes of four wheels (two front and two rear wheels) and using Eq. 3 for the 0-10 cm soil layer and Eq. 4. for the 10-40 cm soil layer versus observed soil bulk density values before (Control) and after heavy traffic (Trafficked). 18 and 12 replicates per depth in the AZ (Azerailles) site and 13 and 12 replicates per depth in the CA (Clermont en Argonne) site for the Control and Trafficked areas respectively (means and standard deviation as error bars)



If we consider that the three simulation cases (case A, B and C, respectively) were three locations on each site where we came to sample after heavy traffic, we would as measurement of the impact of forwarder traffic take the mean of the BD values per depth and site. Therefore, we calculated in the same way the mean of the three cases (AZ site) or of case A and B (CA site; unrealistic values for case C were discarded, as mentioned above). The RMSE was then 0.07 and 0.10 for the AZ and CA site respectively. These RMSE values are almost identical to the ones obtained for the case A simulation.

3.7 Effect of SOC on the impact of heavy traffic on soil BD.

For both sites in the 0-10 cm soil layer of the control plots (C-treatment), variations were best explained with a linear regression between BD and the logarithm (base e) of SOC and WC (Table 3). Bulk density increased with decreasing SOC (Fig. 6a) and with decreasing WC (Table 3). No effect of the C/N ratio could be observed. Soil water content increased with increasing values of SOC (Table 4 and Fig. 6b).

The effect of SOC on the increase in BD after wheeling was expected to be high because of its effect on initial BD and WC. According to Eqs. 5 and 6, λ_n and N are affected by both BD and WC, but the relationship between SOC and mechanical properties is complex: while SOC decreases BD, it increases WC. However, the effect of BD on λ_n (Eq. 5) and N (Eq. 6), respectively, is about one order of magnitude larger than the effect of WC on λ_n (Eq. 5) and N (Eq. 6), respectively. Therefore, both λ_n and N increase with increasing SOC. This results in a decrease of compaction at low stresses, but in an increase of soil compaction at high stresses such as during loading with the forwarder. However, no significant correlation between the effects of SOC and treatment could be detected, i.e. the increase in BD from control to trafficked soil did not increase with increasing SOC (Table 3). This is also seen from the fact that the linear regression lines between BD and log SOC for the different treatments were parallel (Fig. 6a). Hence, the larger λ_n (higher compressibility) at higher SOC has to be compensated with a larger κ (higher rebound).

Fig. 6. Partial relationships between soil bulk density and ln(*SOC*) while the second covariate is held constant at its arithmetic mean (a), and relationships between gravimetric water content (WC) and ln(*SOC*) (b). For each graph, C correspond to the control treatment (undisturbed areas) and T to the trafficked treatment; 30 samples per site and treatment. Sampling in April 2009, one (CA site) to two (AZ site) years after heavy traffic

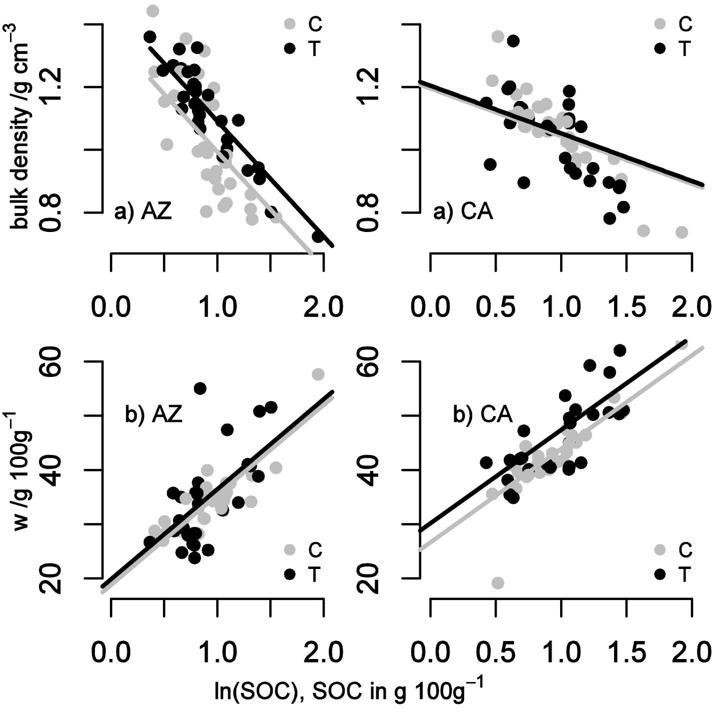


Table 3 Results of the analysis of covariance accounting for effects of treatment, soil gravimetric water content (WC, in g 100g^{-1}) and soil organic carbon content (SOC, in g 100g^{-1}) on soil bulk density (in g cm^{-3}). 30 samples per site × treatment, sampling in April 2009 (one year after heavy traffic for CA, two for AZ).

	intercept	ln(SOC)	WC	treatment T	WC: treatment T	ln(SOC) : treatmentT	ln(SOC): WC:treatmentT	R²	RSE
AZ	1.9 (0.2)***	-0.37 (0.06)***	-0.014 (0.006)*	-0.3 (0.2) n.s.	1.2 (0.5)*	n.s.	n.s.	0.72	0.1
CA	1.64 (0.04)***	-0.15 (0.03)***	-0.01 (0.001)***	0.009 (0.01) n.s.	n.s.	n.s.	n.s.	0.87	0.05

R^2, coefficient of determination; RSE, residual standard error, p-value: 0<***<0.001<**<0.01<*<0.05<n.s.

Table 4 Results of the analysis of covariance accounting for effects of treatment and soil organic carbon content (SOC, in g 100g^{-1}) on soil gravimetric water content (WC, in g 100g^{-1}). 30 samples per site × treatment, sampling in April 2009 (one year after heavy traffic for CA, two for AZ).

	intercept	ln(SOC)	treatment T	ln(SOC) : treatmentT	R^2	RSE
AZ	0.20 (0.02)***	0.06 (0.007)***	0.01 (0.01) n.s.	n.s.	0.53	0.05
CA	0.26 (0.02)***	0.06 (0.007)***	0.03 (0.02)*	n.s.	0.57	0.06

R^2, coefficient of determination; RSE, residual standard error, p-value: 0<***<0.001<**<0.01<*<0.05<n.s.

4. Discussion

4.1 Rebound may be increased in surface layers of forest in comparison to agricultural soils

For both sites and for the three initial soil conditions, no model simulation could predict the lowest BD found in the field at 0-5 cm depth after the forwarder traffic and only a slight increase in BD in the 0-5 cm soil layer in comparison to the increase at 5-10 cm depth. One explanation could be that all the analytical procedures used in the *SoilFlex* model were developed for agricultural soils and do not consider the buffering effect of the forest floor and the change in soil mechanical properties due to the high organic matter content and persistent root mat. The presence of organic matter may have increased the rebound (Soane, 1990). Indeed, a clear improvement of predictions was observed for the surface soil layers when increasing the rebound by using Eq. 4 instead of Eq. 3. In general, *SoilFlex* predictions tended to over-estimate the impact in the upper soil layers and to under-estimate it in the deeper soil layers. Therefore, even if forest floor litter and/or soil organic matter decreased the impact of the forwarder on the soil surface layers, stresses were transmitted to the deeper soil layers. In our study, the

0-10 cm soil layer had a very low density in comparison to the rest of the soil profile; the current one–layer analytical approach may therefore be a possible cause of model accuracy loss (Keller and Lamandé, 2010). Furthermore, WC of the soil surface layers were at or slightly beyond the wet end of the range at which mechanical properties were measured, and therefore, estimation of mechanical properties (Eqs. 5 and 6) may have been less accurate for these layers as compared with the deeper soil layers. It may also be possible that predictions of stress (and therefore BD) were less accurate close to the soil surface, because the contact stresses were estimated and not measured. Besides, the prediction equations by Keller (2005) were developed from measurements using agricultural tyres on arable soil. Therefore, a future improvement of analytical soil compaction models could be to take into account the effect of SOC on soil mechanical parameters, especially on the rebound parameter, but also on stress propagation.

For both sites we found, in the 0-10 cm undisturbed soil layer, a positive correlation between SOC and WC, and a negative correlation between SOC and BD (Fig. 6). Hence, according to the effect of BD and WC on soil susceptibility to compaction (Saffih-Hdadi et al., 2009), a greater amount of SOC should have increased soil compaction during forwarder traffic. Yet the statistical evaluation of the relationship between BD and SOC did not show a dependence of the treatment effect on SOC. At AZ, the linear regression lines between BD and log SOC of the undisturbed and the trafficked soils are shifted but parallel to each other (Fig. 6a). The impact of heavy traffic (i.e., the shift) was constant whatever the SOC. Williamson and Neilsen (2000) found similar results between BD and organic matter before and after 15 passes of a laden logging machine. At CA, the same conclusion could be drawn, except that the impact of heavy traffic on BD (shift) was not significant (Fig. 6a). These results underlined an additional effect of SOC on soil mechanical behaviour besides its effect on initial BD and WC. It was namely possible that soil with high SOC content was highly compressible but also recovered more after unloading (rebound) (Kuan et al., 2007). That is, SOC increases compressibility (i.e. λ_n; Eq. 1) as well as resilience (i.e. κ; Eq. 2). This agrees with our simulations, where the introduction of an increased rebound improved the model performance.

We only measured BD and were not able to analyze the effect of SOC on textural and structural pore spaces (Tranter et al., 2007; Pereira et al., 2007; Boivin et al., 2009). The effect of SOC on the resistance to stresses may have been different for the texture- and structure-related pore spaces, resulting in similar BD after heavy traffic for high SOC and low SOC locations. For example, after accounting for the SOC content effect on BD, the relationship between WC and BD was affected by treatment for the AZ site (the slope between BD and WC was -0.2 and -1.4 for the trafficked and control plots, respectively). This may be related to a change in swelling-shrinking properties after compaction and to a modification of the soil structure (Boivin et al., 2009). For the CA site, the relationship between SOC and WC was affected by treatment, which could originate from different impacts on the textural and structural pore space.

Another possible explanation for the relatively small impact of the forwarder traffic in the topmost soil layer could be that the low density due to high SOC resulted in an increased soil-wheel contact area and thus lower contact stresses. We could not test this hypothesis using *SoilFlex* as the model developed by Keller (2005) to describe stress distribution at the soil-wheel interface did not account for soil properties.

4.2 Evaluation of model performance

The predicted mean vertical contact stress fell into the range of values measured by Horn et al. (2007) for forest machinery. Taking into account the spatial variability of initial soil conditions did not improve model performance. Model simulations performed better for the AZ than for the CA site, because the relationships between soil mechanical and physical properties were measured for AZ but only estimated for CA. Different authors (Défossez et al. 2003; Keller et al., 2007) also found that the simulations of soil deformation under wheeled traffic are very sensitive to the assessment of soil mechanical properties. Considering the minimum BD and maximum WC at the time of traffic yielded unrealistic results for the surface layer of the AZ site when using Eq. 3, and for all depths for the CA site. The mechanical parameters *N* and λ_n calculated were probably not accurate in these cases. Indeed, Saffih-Hdadi et al. (2009) did not measure mechanical parameters beyond a maximum WC of 31%.

The soil was wetter at the time the forwarder drove on the CA site in comparison to the AZ site. The greater improvement of predictions by using the rebound parameter estimation for wet soils (Eq. 4) for the CA site in comparison to the AZ site could be due to these very wet conditions. Yet even by using Eq. 4, the model over-estimated the impact of the forwarder on the soil surface layer of the CA site, unlike the AZ site. AZ and CA depicted nearly identical SOC contents in soil surface layers (2.4-4.7 and 2.2-4.0 at 0-10 cm depth for the AZ and CA sites, respectively), but as they did not have the same WC at the time the forwarder drove on the site, the effect of SOC may have differed. Therefore, the very low impact of heavy traffic on the soil surface layers of the CA site in comparison to the theoretical expectations may be related to the specific role of SOC in wet conditions (Soane, 1990; Mosaddeghi et al. 2000; Pereira et al., 2007). Another explanation could be that the 0-10 cm soil layer was very wet at the time of traffic and could flow (Williamson and Neilsen, 2000; Ampoorter et al., 2010), whereas the minimum 0.1 g g^{-1} drier 10-50 cm soil layer did not. It resulted in a very low impact on BD in the surface layer with only shallow ruts (minority of ruts exceeding 5 cm depth), in accordance with Mosaddeghi et al. (2000).

5. Conclusion

For the AZ site, we achieved the best model performance (RMSE=0.06) considering the mean initial soil conditions (mean initial BD and WC) and the rebound parameter (κ) estimation for wet soil (O'Sullivan and Robertson, 1996) from 0 to 10 cm depth and the estimation of O'Sullivan et al. (1999) from 10 to 50 cm depth. The model estimations predicted well the range of BD values found after forwarder traffic. Therefore, for soil conditions (BD and WC) in the range where soil mechanical parameters were measured, *SoilFlex* yielded accurate predictions even for forest soils with a non-negligible gravel and root content. . Taking into account the variability of soil initial conditions did not improve model performance.

For the CA site, the wetter soil at the time of the compaction and the lack of accurate assessment of mechanical properties decreased the quality of the predictions. Yet using the O'Sullivan and Robertson (1996) estimate for κ from 0 to 10 cm and the O'Sullivan et al. (1999) one from 10 to 50 cm while considering the mean initial conditions led to satisfactory predictions (RMSE=0.11).

For both sites, high SOC content was associated with high WC and low BD, yet the change in BD caused by heavy traffic was not found to be influenced by SOC at 0-10 cm depth. Therefore, soil displaying higher SOC content may have been more compressible due to higher WC and lower BD, but probably recovered more than soils poor in SOC. The soil deformation predictions for the 0-10 cm soil layer of both sites were initially poor, but improved when we increased the rebound parameter. Analytical models of soil compaction need further development to take into account the (mechanical) properties, soil heterogeneity and structure of the litter and organic layers of forest soils.

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II

ASSESSMENT OF THE NATURAL RECOVERY RATE OF SOIL SPECIFIC VOLUME FOLLOWING FOREST SOIL COMPACTION

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Summary

Soil compaction is a major degradation to forest soils. To make decision on restoration, both the soil compactness after heavy traffic and the natural soil recovery rate must be estimated. We estimated the impact of heavy traffic on soil specific volume and its recovery rate on two forest sites by yearly collecting steel cylinders close to field capacity at different depths during three to four years, as currently recommended. Though collected at water contents *w* as homogeneous as possible, the comparison of sample volumes led to inconsistent results. Using *w* as covariate was necessary to quantifying the initial compaction and the soil bulk volume *V* recovery with time and depth. Moreover, compared to the soil *V* and *w* determined with shrinkage analysis, some field values were very large, suggesting an artifact due to hammering the cylinder at large *w*. The surface layer (0-10 cm) of the less compacted site showed no residual compactness three years after heavy traffic and the 10-20 cm layer compactness decreased significantly with time. The compactness of the second site decreased significantly only in the 0-10 cm layer, and the recovery was still ongoing after the third year of monitoring. This site had less swelling clays and larger clay content. The recovery of the soil volume was attributed to shrink-swell processes. Longer monitoring is required to validate these trends, and further research should evaluate the need for more accurate monitoring based on shrinkage analysis and the use of soil organic carbon and clay content as covariates.

Abbreviations: C, control; *D*, soil depth; ShC, soil shrinkage curve; T, trafficked; *Tr*, treatment; *V*, soil specific volume ; *w*, soil gravimetric water content.

Introduction

Soil compaction has been identified as one of the six main threats to a sustained soil quality in Europe (Commission of the European Communities, 2006). The growing mechanization of forest operations is increasing soil compaction, and neither tillage nor reclamation can be applied. Compacted forest soils, therefore, recover their functional and structural quality through natural processes such as freezing-thawing, wetting-drying, bioturbation, and erosion-deposition. For a given soil, climate, and compaction intensity, the estimated rates to recover initial quality depend on the soil properties considered (Croke et al., 2001; Page- Dumroese et al., 2006; Von Wilpert & Schaeffer, 2006). Soil bulk density or specific volume is one of the most frequently used parameters to assess the impact of heavy traffic. It is easy and inexpensive to estimate and directly connected to the soil compactness (Hakansson & Lipiec, 2000). The reported bulk density recovery rates after forest harvesting vary considerably from several years (Brais, 2001; Page- Dumroese et al., 2006) to several decades (Croke et al., 2001; Froehlich et al., 1985). Several authors reported a dependence of bulk density recovery rate on soil texture (Brais, 2001; Page- Dumroese et al., 2006), the slowest recovering soils being the fine-textured ones, whereas other authors observed no effect of soil type on the recovery rate (Croke et al., 2001). These very different conclusions could come from the different methodologies used to assess soil regeneration, or from the various initial impacts (different machinery, soil type and moisture content), or from the varying regeneration factors (climate, slope and soil biological activity). Besides, soil bulk volume and mechanical stability have been found to strongly depend on soil properties like soil organic carbon (SOC) content, texture, clay type and water content (e.g. Imhoff et al., 2004, Boivin et al. 2004, 2006b; Heuscher et al., 2005; Schaeffer et al., 2008). Bormann & Klaassen (2008) observed a seasonal pattern in soil bulk density variations with a decrease in spring and summer and an increase in autumn for a Stagnosol under forest. Therefore spatial and temporal variability of soil properties at field scale are susceptible of jeopardizing a reliable assessment of the impact of heavy traffic and of the regeneration rate on long-term field experimental plots (Hakansson & Lipiec, 2000). The most common recommendation is to sample the soil volume close to field capacity to get rid of the changes in soil volume due to swelling. An alternative is to measure the soil volume in the laboratory with the plastic bag method (Boivin et al., 1990) after equilibrium at a target matric potential on a sand bed. This bulk volume measurement can be performed on clods and do not require a steel cylinder, thus allowing for re-saturation of the sample. Some authors showed the potential of shrinkage analysis and covariance analysis with SOC and clay content as co variables for the diagnosis of compaction (Boivin et al., 2006b, Schaeffer et al., 2008). This approach requires, however, additional analyses, and is, therefore, more expensive to perform.

In this study, we discussed an intermediate and less expensive alternative. The compaction of two forest soils following heavy traffic was quantified based on steel cylinder volume measurement, and the recovery rate of the initial volume was monitored yearly during four years. Additionally to the bulk specific volume, the water content of the sample at the time of collection was measured, and used later on as covariate in covariance analysis to take into account the spatial variability in soil properties. The measured volumes and water contents were compared to the shrinkage curves of the same samples to discuss the quality of the data.

Materials and methods

Study sites

We worked on two experimental sites set up in north-eastern France in the "Hauts-Bois" forest – Azerailles (48° 29′ 19″ N, 6° 41′ 43″ E), Meurthe et Moselle, and in the "Grand Pays" forest – Clermont en Argonne (49° 06′ 23″ N, 5° 04′ 18″ E), Meuse. The sites of Clermont en Argonne (CA) and Azerailles (AZ) have an elevation of 270 m and 300 m above mean sea level respectively, with a maximum of 8 m variation of level through each site. The climate of the region is characterised by a 30-year mean annual temperature of 9°C (AZ) to 9.5 °C (CA) and a 30-year annual precipitation of 900 mm (AZ) to 1000 mm (CA).

Both sites were clear-cut over a 5 ha area, logs were removed using a cable yarding system to avoid soil disturbance during harvesting. The remaining branch and slash were carried out of the sites by hand or using an iron horse to limit soil surface cover heterogeneity during the following forwarder traffic. Each site was split into three blocks. In each block, the same full-loaded 8-wheel drive forwarder (23 and 17t) drove on land strips for an equivalent of two passes in May 2007 and March 2008 in the Azerailles (AZ) and Clermont en Argonne (CA) sites respectively. Each plot measured 50 × 50 m, with two undisturbed 10 × 50 m land strips (C) on each side of the 30 × 50 m trafficked area (T). Adjacent to these three plots per site, undisturbed plots of same surface area were also considered as control (C) and were used at some occasion to assess the spatial variability of soil physical properties without or after disturbance. In autumn 2007 (AZ) and 2008 (CA) the entire site surface area was planted with sessile oak (*Quercus petraea* L.) at a density of 1600 seedlings ha^{-1} for each site. In the AZ site the composition of the vegetation was immediately (within two months) modified following the forwarder traffic; the T-treatment consisted mostly of rushes (*Juncus* sp.), and the C-treatment was mostly composed of bramble (*Rubus fructicosus* L.). In the CA site the same change in vegetation composition could also be observed but it was slighter than in the AZ site.

The soils of both sites are developed on a silt loam layer (0-50 cm) laying on a clayey material (weathering of a Keuper marl for the AZ site and weathering of a Cenomanian gaize rock for the CA site). This strong textural discontinuity caused limited temporary water logging; the lower part of the silt loam layer showing hydromorphic figures e.g. ferri-manganic concretions. Some differences in the clay content and in the clay mineralogy were observed between the two sites, besides the soil at CA is more acidic than at AZ (Table 1).

Table 1 Soil pH in water, silt, clay, and organic carbon contents (mean and standard deviation in brackets, 18 and 13 replicates for the AZ and CA sites respectively) and identification of clay minerals for the AZ (Azerailles) and CA (Clermont en Argonne) sites.

	pHwater	silt	clay	SOC†	XRD‡
			%		
			AZ		
0-10 cm	4.8	55.6 (6.8)	22.2 (4.0)	3.1 (0.7)	K, I, Ch, I/C int.
10-30 cm	4.6	56.9 (6.8)	22.6 (5.2)	1.5 (0.4)	K, I, Ch, I/C int.
30-60 cm	4.7	46.5 (12.2)	38.9 (14.4)	0.1 (0.6)	K, I, Ch, I/C int.
			CA		
0-10 cm	4.4	72.2 (2.9)	12.8 (1.1)	2.9 (0.5)	K, I, V, HIC, S
10-30 cm	4.5	70.6 (3.8)	14.1 (2.7)	0.8 (0.4)	K, I, V, HIC, S
30-60 cm	4.7	61.9 (7.2)	24.8 (6.3)	0.8 (0.8)	K, I, V,HIC, S

† Soil organic carbon (SOC) content, determination using a CHN analyser

‡ identification of clay minerals using X-ray diffraction (XRD); K, kaolinite; I, illite; Ch, chlorite; I/C int., interstratified illite-chlorite ; V, vermiculite; S, smectite; HIC., hydroxy-interlayered clay.

Soil specific volume measurements

Each year, two pits were dug in the T-treatment plots of each block. One pit was located in the 10 × 50 m control area and the other was located in the middle of the 30 × 50 m trafficked area to compare treatments while limiting the effects of spatial variability. Each pit was approximately one meter wide and at each side of the pit front we sampled two profiles of undisturbed soil samples. We used 500 cm^3 and 250 cm^3 steel cylinders to collect two replicates of soil samples every 10 and 5 cm in the C- and T-treatment areas respectively for each profile. Soil specific volume (*V*) was then calculated as the cylinder volume divided by the mass of the dry soil sample (105°C over 72h) and was corrected for the rock fragment content. The first pits sampled (2007 for AZ, 2008 for CA) were randomly distributed while avoiding the few very deep ruts accompanied with bulges at the edges that were formed in some place by wheeling on the wettest soil conditions (plastic deformation; Ampoorter et al., 2010). We also avoided the large stumps remaining on the site to avoid large amount of roots into our soil samples. The pits of the following years were always dug ahead to sample soil that was only affected by treatment and not by the previous sampling. To avoid a possible interaction between the effects of season and treatment we always sampled at beginning of May whatever the climate and sampled both sites within one to two weeks. We measured soil gravimetric water content (*w*) on each sample to cope with soil moisture conditions for the different campaigns *via* shrinkage curve determination.

Shrinkage curve assessment

In February 2011, we collected soil cylinders of 100 cm^3 approximately, randomly distributed over the entire experimental site area (in the three blocks × two treatments [control and trafficked] 30 × 50 m plots). Four and three replicates per block × treatment were sampled at 5 and 15 cm depth respectively. Soil samples were then saturated with deionized water on a sand table at a water potential of -8 cm with respect to the centre of the samples. The saturated soil sample was then placed on the experimental device described in Boivin et al. (2004). The shrinkage apparatus recorded simultaneously and quasi continuously the changes in soil height (linear transducer) and mass (balance) while the soil was let to dry in controlled evaporation conditions <the top surface and the lateral faces of the sample allowed evaporation> until the soil weight remained constant. We calculated the changes in the sample water content *w* using the every 5 min recorded weight and the dry sample weights. The saturated and air-dried soil volumes were measured using the plastic bag method (Boivin et al., 1990) and used to convert the recorded sample heights changes into changes of soil volume (Boivin et al., 2004; Boivin et al., 2006b).

Measurement of the field daily changes in soil water content

A data-logger, located approximately in the middle of each site, recorded soil water content using five Time Domain Reflectometry (TDR) sensors per treatment and site inserted into the soil at 15 and 10 cm in the control and trafficked plots respectively. The depths, with reference to the soil surface, differed between treatments so that the different sensors allow investigations of the same soil layer (rut of approximately 5 cm depth). The TDR outputs (soil dielectric constant) were converted to volumetric water content using calibration curves derived from laboratory measurements of both gravimetric water content and TDR data on undisturbed soil cores (Heathman et al., 2003). For each TDR sensor, outputs were recorded every two hours, and the day mean value of soil moisture was recorded.

Statistical analysis
For statistical procedures' purpose, we used the mean *V* value of the two 250 cm^3 cylinders from the trafficked area corresponding to the same 10 cm depth increment. For each year considered, we assumed that *V* values were auto-correlated as a function of depth as we sampled on two soil profiles per treatment × block. We also assumed an auto-correlation of the data as a function of time for each soil profile sampled. Indeed two soil profiles close in time were closer in space than two profiles separated by a long time interval. Therefore we had to test for auto-correlation structure of the residuals of the two statistical models considered:
For each year:

$$y = \mu + D + Tr + Tr : D + \mathrm{Pr} + e$$

For each soil layer (0-10, 10-20, 20-30, 30-40, 40-50 and 50-60 cm):

$$y' = \mu' + Ti + Bl + e'$$

with y the soil specific volume in cm^3g^{-1}; y', difference (T minus C) in mean specific volume per block; *D*, depth in cm (centre of the 10 cm high cylinders, i.e. 5, 15, 25, 35, 45 and 55 cm, respectively); *Tr*, treatment effect (two levels: C and T); Pr, soil profile (in each block × treatment, two soil profile were sampled for specific volume measurements); Ti, time (in number of year after heavy traffic); Bl, block (three levels); e, residuals with an auto-regressive structure of lag 1 with a depth covariate within each soil profile; e', residuals with an auto-regressive structure of lag 1 with a time covariate within each block. The effect of soil profile or block was considered as a random effect inducing random variations around the fixed and reproducible effects.

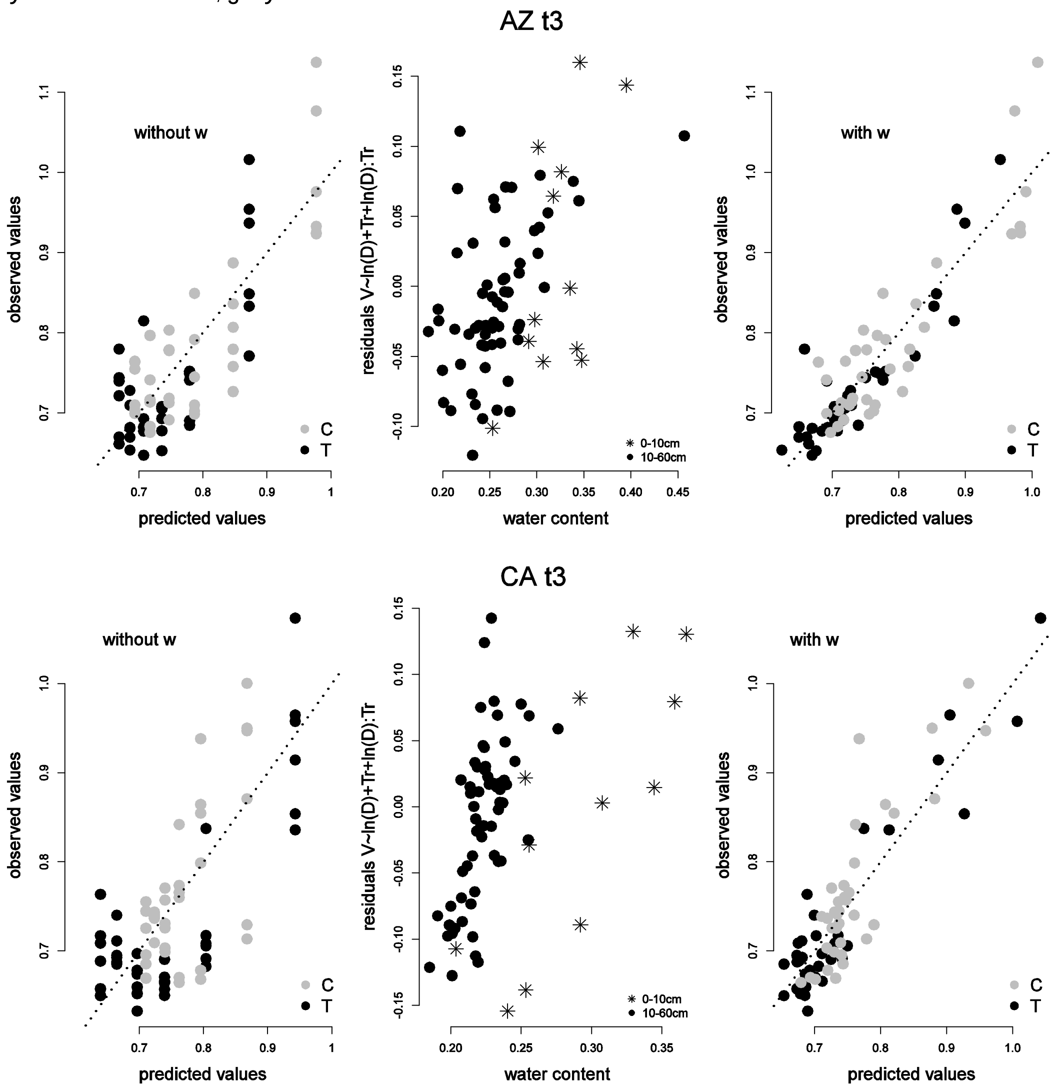
The linear mixed models were fit using the Maximum Likelihood (ML) procedure of the R software. Selection of the best model was based on the distribution of the residuals, the Aikake information criterion (AIC), Bayesian information criterion (BIC) and the likelihood ratio Chi-2 test (LRT) for nested models. Since the first model showed a significant interaction between *D* and *Tr*, the second model was tested on the values of specific volume per treatment × soil layer. Yet for the CA site, soil specific volume of the undisturbed area did vary significantly with time. Therefore, we tested the second model on the bulk density difference between T and C rather than on the density values. We found that the auto-correlation structure of the residuals of the second model was not significant whatever the soil layer considered, therefore we removed it. In the following sections, the term significant refers to a significance at a probability α-level of 0.05.

Results

The effect of the forwarder traffic on sample Volume V and its changes with depth

While considering each sampling year separately, we found that *V* decreased exponentially as a function of depth (*D*) for both sites (Table 2). The treatment (*Tr*) and *D* interaction effect on *V* was significant for most years, except for the first and the fourth year after compaction for the CA and AZ sites respectively. The coefficient associated to the effect of heavy traffic on the slope of the relationship between *V* and ln(*D*) was positive except for the third year after heavy traffic in CA. Indeed *V* decreased less as a function of *D* in the trafficked (T) treatment in comparison to the control (C).

Figure 1 Observed soil specific volume (*V*) values as a function of the predicted ones for the different selected mixed linear models. “without w” refers to the model explaining V as a function of soil depth (D, in cm) and treatment (Tr) and “with w” refers to the model explaining V as a function of soil depth (D), treatment (Tr) and soil gravimetric water content (*w,* in g g^{-1}). The residuals of the “without w” model are plotted as a function of *w* in the centre panel; dots: 10-60 cm depth, stars: 0-10 cm soil layer. The pictured data correspond to the third year after heavy traffic (2010 for the AZ site, 2011 for the CA site). Black symbols: trafficked, grey dots: control.



Impact of water content on sample Volume

At first we used the coefficients of the different linear mixed models considering the effects of *Tr* and *D* on *V* (Table 2) to assess the changes in the impact of traffic with time at a given depth.

However taking a constant depth of 1 cm (ln(1)=0) at the AZ site gave an impact of -0.23 cm^3g^{-1} after one year against -0.33 cm^3g^{-1} just after heavy traffic and -0.31 cm^3g^{-1} two years after traffic. The high standard errors associated with the parameter estimates (about one third of the estimate's value, Table 2) could explain these variations. However, we observed a correlation between the residuals of the models and the water content at the sampling time for each year (Fig. 1).

Therefore we used soil gravimetric water content (*w*) at the time of sampling as a covariate (Table 3). The effect of this covariate was significant for all the years after compaction and improved considerably the distribution of the residuals (Fig. 1). Accounting for *w* variations made the estimations of the *Tr* effect more precise, decreased the intercept estimate and decreased slightly the estimates of the *Tr*- and *D*-effects. For most sampling years, it also decreased the value of the lag 1 auto-correlation of the residuals as a function of depth (Tables 2 and 3).

Table 2 Linear mixed model coefficients and associated standard error (in brackets) explaining soil specific volume as a function of treatment and depth, standard deviation (SD) of the random effect (soil profile; Pr), residual standard deviation (RSD), value of the lag 1 autocorrelation (Phi AR(~D|Pr)), Aikake information criterion (AIC) and Bayesian information criterion (BIC). One model per year after compaction (t0 being immediately following heavy traffic, t1 being one year after heavy traffic…) and site (AZ being Azerailles, and CA being Clermont en Argonne) was fit. Selection of the significant effects was based on the AIC, BIC and p-value of the likelihood ratio Chi-2 test (LRT) for nested models after analysis of the distribution of the residuals. Only the coefficients of the significant effects (α-level of 0.05) are given; n.s. corresponding to a non-significant effect.

	μ†	$\ln(D)$	Tr	$\ln(D)$: Tr	SD Pr	RSD	Phi AR(~D|Pr)	AIC; BIC
AZ t0	1.24 (0.06)	-0.13 (0.02)	-0.33 (0.08)	0.08 (0.03)	<0.001	0.08	0.91	-128; -113
AZ t1	1.27 (0.05)	-0.15 (0.02)	-0.23 (0.08)	0.05 (0.02)	<0.001	0.07	0.87	-158; -144
AZ t2	1.28 (0.06)	-0.16 (0.02)	-0.31 (0.09)	0.08 (0.03)	<0.001	0.08	0.93	-149; -133
AZ t3	1.17 (0.05)	-0.12 (0.01)	-0.16 (0.06)	0.03 (0.02)	<0.001	0.06	0.88	-191; -175
AZ t4	1.21 (0.04)	-0.13 (0.01)	-0.05 (0.02)	n.s.	<0.001	0.07	0.93	-182; -168
CA t0	1.06 (0.03)	-0.096 (0.009)	-0.25 (0.04)	0.06 (0.01)	<0.001	0.04	0.95	-266; -250
CA t1	1.07 (0.02)	-0.09 (0.007)	-0.05 (0.01)	n.s.	<0.001	0.04	0.86	-237; -224
CA t2	1.18 (0.04)	-0.13 (0.01)	-0.14 (0.05)	0.03 (0.02)	<0.001	0.05	0.93	-236; -220
CA t3	0.97 (0.05)	-0.07 (0.01)	0.17 (0.07)	-0.06 (0.02)	<0.001	0.07	0.96	-199; -183

†coefficients associated to the following linear mixed model: $y = \mu + \ln(D) + Tr + Tr:\ln(D) + \mathrm{Pr} + e$, with y being soil specific volume ($cm^3\ g^{-1}$), μ being the model intercept (control at a depth of 1 cm), *D* being soil depth (cm), *Tr* being the T(trafficked)-treatment effect, Tr:ln(D) being the effect of the T-treatment on the slope of *y* against ln(*D*), *Pr* being the soil profile (random qualitative variable; two replicates × two treatments × three blocks), and *e* being the residuals with an auto-regressive structure of lag 1 with a *D* covariate within each *Pr.*

Table 3 Linear mixed model coefficients and associated standard error (in brackets) explaining soil specific volume as a function of gravimetric water content (*w*), treatment (*Tr*) and depth (*D*), standard deviation (SD) of the random effect (soil profile; *Pr*), residual standard deviation (RSD), value of the lag 1 autocorrelation (Phi AR(~D|Pr)), Aikake information criterion (AIC) and Bayesian information criterion (BIC). One model per year after compaction (t0 being immediately following heavy traffic, t1 being one year after heavy traffic…) and site (AZ being Azerailles, and CA being Clermont en Argonne) was fit. Selection of the significant effects was based on the AIC, BIC and p-value of the likelihood ratio Chi-2 test (LRT) for nested models after analysis of the distribution of the residuals. Only the coefficients of the significant effects (α-level of 0.05) are given; n.s. corresponding to a non-significant effect.

	μ	w	ln(D)	Tr	ln(D): Tr	SD Pr	RSD	Phi AR(~D|Pr)	AIC; BIC
AZ t0	0.81 (0.08)	1.3 (0.2)	-0.11 (0.01)	-0.29 (0.06)	0.08 (0.02)	<0.001	0.06	0.85	-155; -139
AZ t2	1.00 (0.08)	0.9 (0.2)	-0.13 (0.02)	-0.29 (0.08)	0.08 (0.02)	<0.001	0.07	0.93	-166; -148
AZ t3	0.83 (0.06)	0.9 (0.1)	-0.09 (0.01)	-0.14 (0.05)	0.03 (0.01)	0.02	0.05	0.86	-223; -204
AZ t4	1.06 (0.07)	0.6 (0.2)	-0.13 (0.01)	-0.03 (0.03)	n.s.	0.002	0.07	0.93	-187; -171
CA t1	0.65 (0.06)	0.8 (0.1)	-0.04 (0.009)	-0.04 (0.008)	n.s.	<0.001	0.03	0.17	-270; -255
CA t2	0.73 (0.06)	1.0 (0.1)	-0.07 (0.01)	-0.13 (0.03)	0.03 (0.01)	<0.001	0.03	0.91	-288; -270
CA t3	0.49 (0.07)	1.4 (0.2)	-0.02 (0.01)	0.16 (0.05)	-0.05 (0.01)	0.02	0.05	0.94	-240; -222

Table 4 Mean and standard deviation (in brackets) of soil specific volume (V, in $cm^3 g^{-1}$) and water content (w, in $g g^{-1}$) measured during the different cylinder sampling campaigns and of maximum V and w (*Vsat* and *wsat*) and minimum V (*Vres*) of the shrinkage curves (ShC). For each site × block × treatment (C and T corresponding to the 30 × 50 m undisturbed and trafficked land strips respectively), four and three replicates were sampled in February 2011 to measure soil shrinkage curves at 5 and 15 cm depth respectively. For each site × block × year × treatment (corresponding to the 10 × 50 m undisturbed (C) close to the 30 × 50 m trafficked (T) area), two replicates per depth were sampled using cylinders to determine V and w.

	5 cm				15 cm			
	C-treatment; *V*	T-treatment; *V*	C-treatment; *w*	T-treatment; *w*	C-treatment; *V*	T-treatment; *V*	C-treatment; *w*	T-treatment; *w*
AZ t0	1.04 (0.13)	0.82 (0.08)	0.32 (0.03)	0.29 (0.05)	0.82 (0.04)	0.74 (0.03)	0.26 (0.02)	0.24 (0.02)
AZ t1	1.06 (0.14)	0.90 (0.05)			0.79 (0.07)	0.70 (0.01)		
AZ t2	1.04 (0.19)	0.86 (0.11)	0.29 (0.09)	0.27 (0.05)	0.79 (0.07)	0.74 (0.02)	0.25 (0.06)	0.24 (0.02)
AZ t3	0.99 (0.09)	0.89 (0.09)	0.33 (0.02)	0.31 (0.05)	0.80 (0.06)	0.73 (0.03)	0.26 (0.02)	0.26 (0.03)
AZ t4	1.01 (0.06)	0.95 (0.11)	0.27 (0.05)	0.22 (0.05)	0.86 (0.08)	0.75 (0.04)	0.25 (0.02)	0.21 (0.02)
CA t0	0.90 (0.08)	0.75 (0.03)			0.78 (0.05)	0.69 (0.02)		
CA t1	0.94 (0.06)	0.86 (0.05)	0.41 (0.03)	0.39 (0.05)	0.83 (0.06)	0.73 (0.03)	0.28 (0.02)	0.27 (0.04)
CA t2	0.97 (0.08)	0.89 (0.04)	0.36 (0.05)	0.35 (0.04)	0.81 (0.05)	0.71 (0.04)	0.24 (0.006)	0.23 (0.01)
CA t3	0.87 (0.12)	0.93 (0.09)	0.30 (0.05)	0.29 (0.06)	0.80 (0.11)	0.72 (0.06)	0.24 (0.03)	0.21 (0.01)
	Vsat / *Vres* C	*Vsat* / *Vres* T	*wsat* C	*wsat* T	*Vsat* / *Vres* C	*Vsat* / *Vres* T	*wsat* C	*wsat* T
AZ ShC	0.88 (0.08)/ 0.79 (0.05)	0.79 (0.06)/ 0.74 (0.04)	0.42 (0.06)	0.36 (0.05)	0.78 (0.07)/ 0.74 (0.06)	0.67 (0.03)/ 0.64 (0.03)	0.32 (0.05)	0.26 (0.06)
CA ShC	0.87 (0.09)/ 0.84 (0.08)	0.76 (0.06)/ 0.74 (0.05)	0.39 (0.05)	0.32 (0.04)	0.75 (0.05)/ 0.74 (0.04)	0.69 (0.04)/ 0.68 (0.04)	0.28 (0.03)	0.25 (0.03)

The water content w of the samples varied largely between treatments and from year to year (Table 4), which may explain the variations in V due to shrink-swell properties of the soils. This was discussed based on soil shrinkages curves (ShC) assessment with reporting on these curves the w-V observed values of the samples collected in May 2009, 2010 and 2011, respectively (Figs. 2, 3 and 4). On these figures, the ShCs of the different samples are spread on the V axis, and the control samples volumes seem, in average, larger than the trafficked samples volume, whatever the water content. Moreover, the water content range of the control soil samples is larger. Most of the soil samples from the pits were close to water saturation and laid on different ShC, with some samples showing very large w-V

values. The clouds of the pits values showed a steep slope of V according to w, at the wet end of the ShCs.

Figure 2 Comparison between V measured in May 2010 and 2011 using the cylinder method (dot, corresponding to the 10 × 50 m undisturbed (C, light grey) close to the 30 × 50 m trafficked (T, black) area), the shrinkage curves (dotted lines, light grey and black corresponding to the 30 × 50 m undisturbed and trafficked land strips, respectively) measured on soil sampled in February 2011 at 5cm depth and the saturation line (grey solid line).

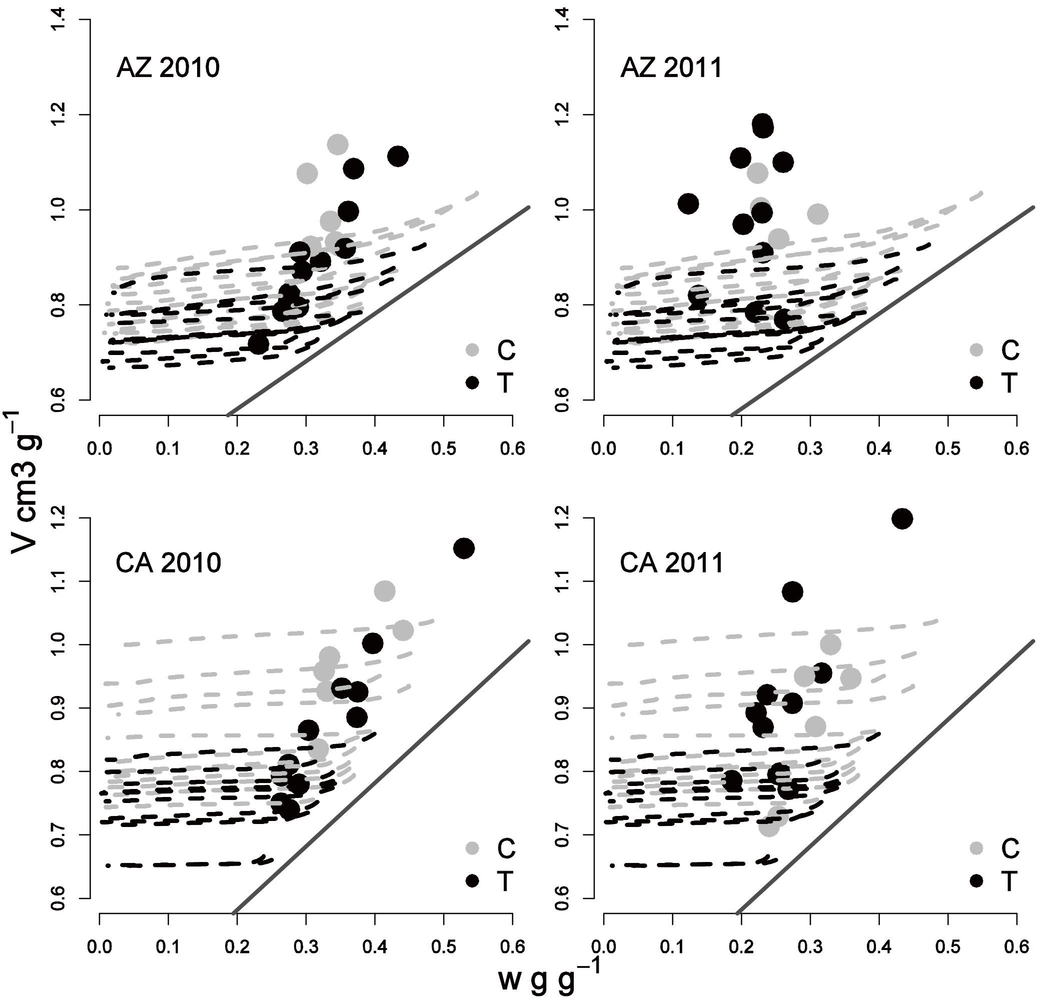


Figure 3 Comparison between *V* measured in May 2010 and 2011 using the cylinder method (dots, control (C) in light grey and trafficked (T) in black), the shrinkage curves (dotted line, C in light grey and T in black) measured on soil sampled in February 2011 at 15cm depth and the saturation line (grey solid line).

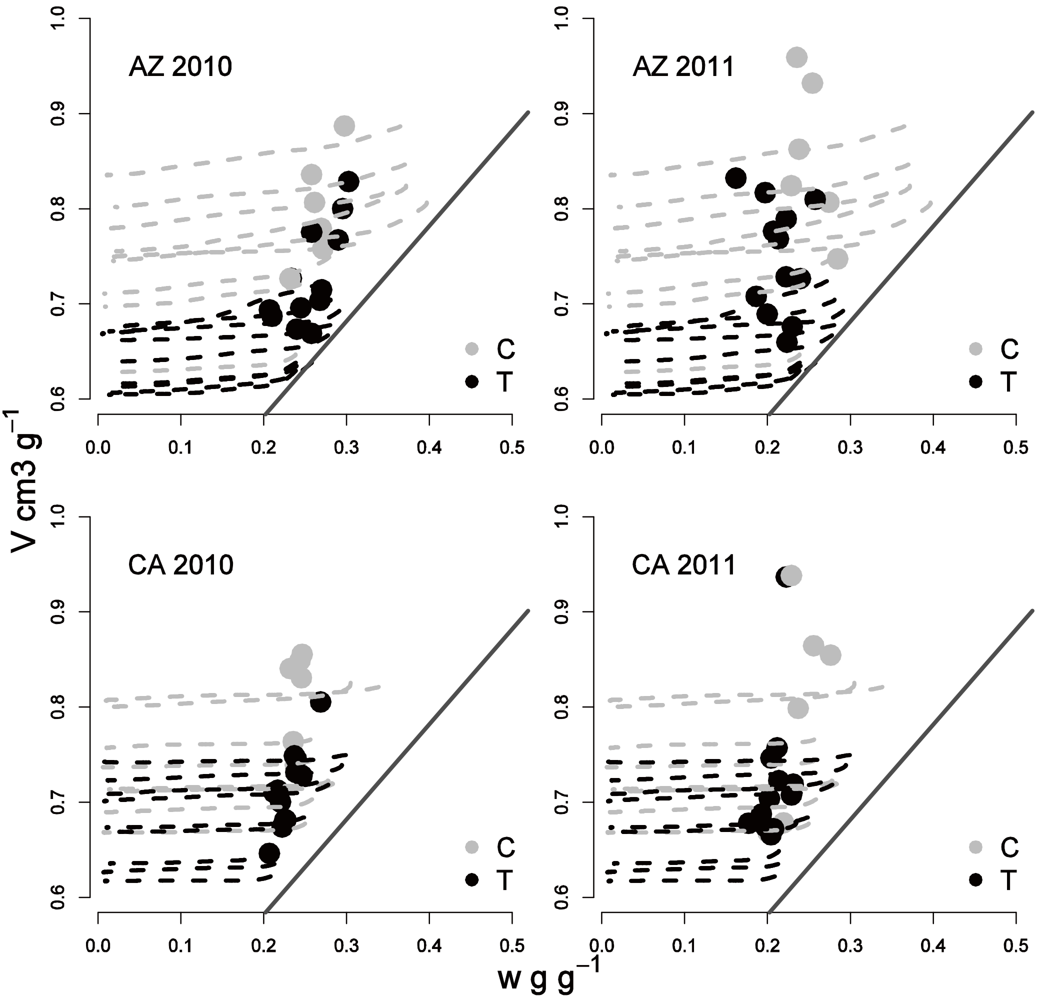
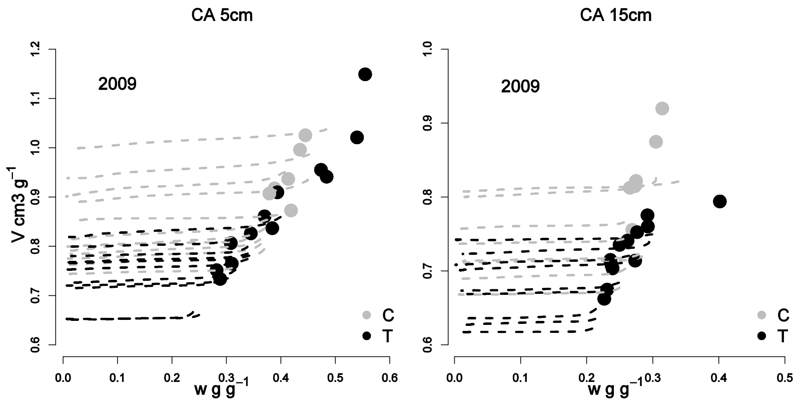


Figure 4 Comparison between *V* measured in May 2009 using the cylinder method (dots, control (C) in grey and trafficked (T) in black) and the shrinkage curves (dotted line, C in grey and T in black) measured on soil sampled in February 2011 at 5 and 15cm depth for the CA site.



In May 2009 (one year after compaction) the impact of heavy traffic on soil specific volume was small compared to the one estimated in 2010 for the CA site even after accounting for *w* variations (Tables 2 and 3). When comparing the ShC determined in February 2011 and the *w*-*V* coordinates of the samples measured in May 2009 on this site (Fig. 4, Table 4), we observed that most of the soil samples collected in May 2009 had a water content larger than the maximum water content (8 hPa) determined on the ShC of the samples collected 18 months later. A steep slope of the ShC close to water saturation is often observed with compacted soils (Schaeffer et al., 2008). More questionable is the field water content larger than the maximum value obtained in the laboratory at -10 hPa equilibrium (May 2009).

Changes in soil compaction with time

We found a significant effect of duration after compaction on the difference of soil *V* between the T- and C-treatment for the 0-10 cm soil layer at AZ and the 0-10, 10-20 and 30-40 cm soil layers at CA (Table 5; the coefficients associated with the effect of time since compaction are significant at the α-level of 0.05 unless stated otherwise). The initial impact of heavy traffic on *V* of the 0-10 cm soil layer was larger for the AZ site than the CA site (i.e. the absolute value of the difference in *V* between T and C was higher at AZ than at CA immediately after traffic, see the values of μ' of Table 5 or Fig. 5). Three years after disturbance no residual effect could be detected in the CA site whereas four years after disturbance a slight effect remained in the AZ site (Fig. 5). The effect of the difference in soil *w* on the difference in soil *V* between the T- and C-treatment was significant at all depth except for the 0-10 cm soil layer of the AZ site. The wetter the trafficked soil compared to the undisturbed soil at the time of sampling, the smaller the impact of traffic on soil *V* and *vice versa* (Fig. 5).

The CA site depicted very large volumetric water contents during most of the year in the trafficked in comparison to the undisturbed soil (Fig. 6). During the dry periods, trafficked and control soils were nearly at the same water content. Thus, more intense wetting-drying cycles were experienced in the T- than in the C-treatment. In the AZ site the same observation could be made but only during the first year after heavy traffic; afterwards the differences in the amplitude of water content variations were slighter.

Figure 5 Evolution of the difference in soil *V* between trafficked (T) and undisturbed (C) soil and selected linear mixed model (dotted lines) explaining it as a function of the number of year after heavy traffic and as a function of the difference in soil water content between T and C. In each panel the second covariate (time after compaction or water content) is held constant at its arithmetic mean. Only the results of the 0-10 cm soil layer are shown as it is the only soil layer where we observed a significant evolution with time for both sites. The different colours correspond to the blocks (I, II and III) and the different symbols to the years after disturbance (t0, t1, t2, t3, t4 relating to immediately, one year, two years, three years and four years following the forwarder traffic respectively).

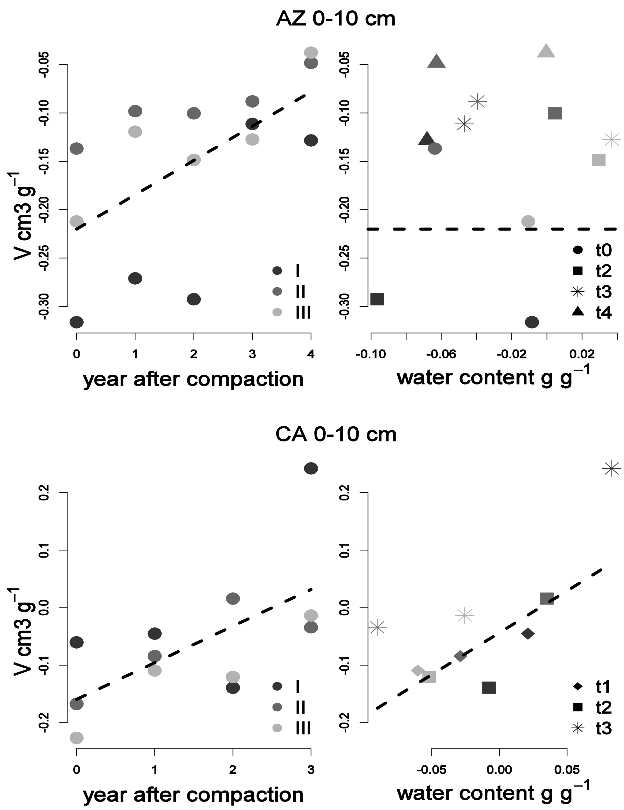
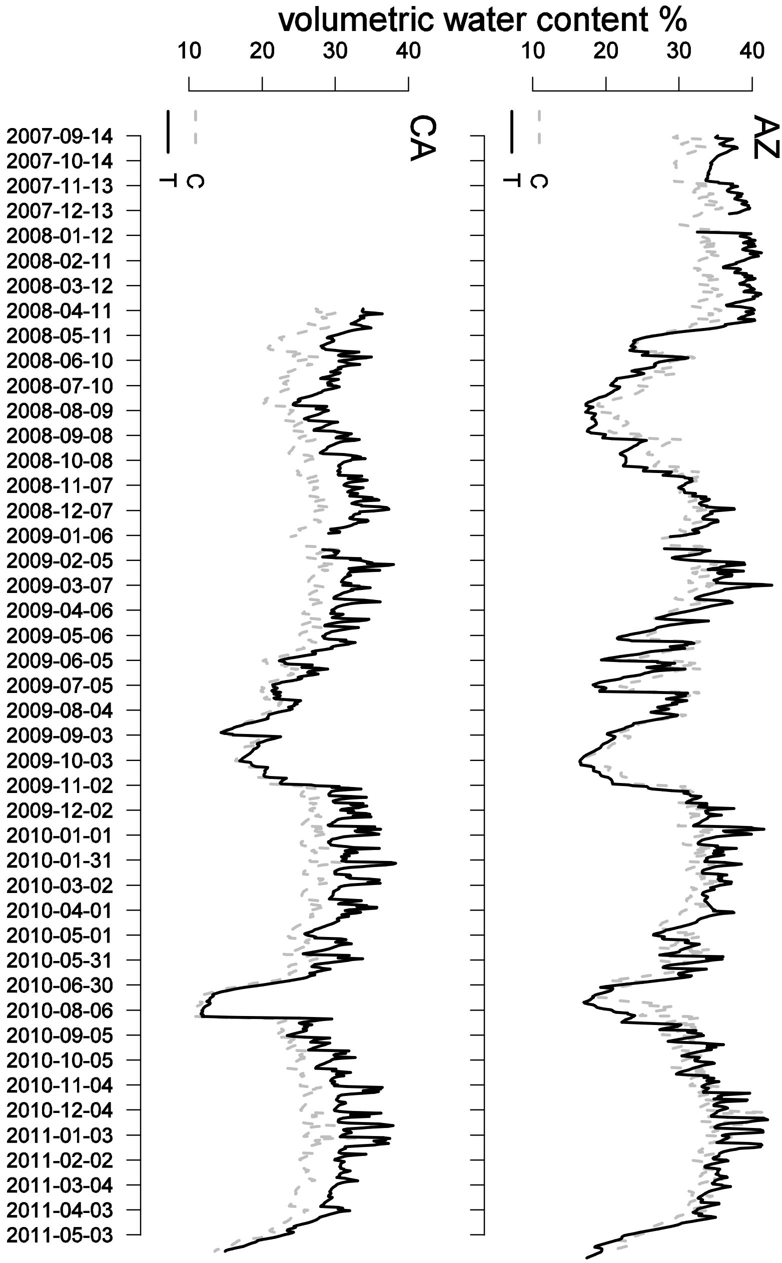


Table 5 Linear mixed model coefficients and associated standard error (in brackets) explaining the changes in the difference in soil bulk volume V between T (trafficked) and C (control) treatment as a function of difference in gravimetric water content (*w*) between treatments, and time since compaction (*Ti*), standard deviation (SD) of the random effect (block; *Bl*), residual standard deviation (RSD), Aikake information criterion (AIC) and Bayesian information criterion (BIC). One model per soil layer and site (AZ being Azerailles, and CA being Clermont en Argonne) was fit. Selection of the significant effects was based on the AIC, BIC and p-value of the likelihood ratio Chi-2 test (LRT) for nested models after analysis of the distribution of the residuals. Only the coefficients of the significant effects (α-level of 0.05) are given; n.s. corresponding to a non-significant effect.

	μ'†	w'	Ti	$w':Ti$	SD *Bl*	RSD	AIC; BIC
AZ 0-10 cm	-0.22 (0.03)	n.s.	0.04 (0.007)	n.s.	0.05	0.002	-39; -36
AZ 10-20 cm	-0.06 (0.01)	0.8 (0.3)	n.s.	n.s.	<0.001	0.03	-42; -40
AZ 20-30 cm	-0.06 (0.01)	0.5 (0.3)	n.s.	n.s.	0.01	0.03	-40; -38
AZ 30-40 cm	-0.04 (0.01)	0.7 (0.2)	n.s.	n.s.	<0.001	0.03	-39; -37
AZ 40-50 cm	-0.008 (0.01)	0.6 (0.2)	n.s.	n.s.	<0.001	0.04	-25; -25
AZ 50-60 cm	0.02 (0.02)	0.5 (0.3)	n.s.	n.s.	<0.001	0.06	-27; -25
CA 0-10 cm	-0.14 (0.05)	1.5 (0.4)	0.06 (0.02)	n.s.	<0.001	0.05	-17; -16
CA 10-20 cm	-0.10 (0.03)	0.2 (1.1)	0.02 (0.001)	0.99 (0.03)	0.04	0.001	-29; -27
CA 20-30 cm	-0.03 (0.01)	2.6 (0.7)	n.s.	n.s.	<0.001	0.03	-30; -29
CA 30-40 cm	0.03 (0.02)	2.5 (0.8)	-0.02 (0.009)	n.s.	<0.001	0.02	-34; -33
CA 40-50 cm	-0.01 (0.01)	1.8 (0.7)	n.s.	n.s.	<0.001	0.03	-30; -29
CA 50-60 cm	-0.008 (0.008)	1.1 (0.4)	n.s.	n.s.	<0.001	0.03	-33; -32

†coefficients associated to the following linear mixed model:
$y' = \mu' + w' + Ti + w':Ti + Bl + \varepsilon$, with y' being the difference in *V* between T and C ($cm^3\ g^{-1}$); *μ'* being the model intercept (difference a few months after heavy traffic); *w'* being the difference in *w* between T and C ($g\ g^{-1}$); *Ti* being the time after heavy traffic (in number of year after heavy traffic); *w'*:*Ti* being the effect of time on the slope of *y'* against *w'*; Bl, being the block (three levels); and ε being the residuals.

Figure 6 Changes in the soil volumetric water content as recorded using TDR sensors (mean per day and treatment, one measure every two hours for each of the 5 sensors per treatment).



Discussion

Our results will be commented according to two points of view; first the requirements in the methodology to assess changes in bulk density and compaction, and second the observed recovery of soil volume in the experimented fields.

Assessment soil compaction at field scale: methodological considerations

Assessing soil compaction by the measurement of soil bulk density is the most common and accessible technique. This is supported by many results showing that soil compaction occurs at the expense of the coarser pores (e.g. Schaeffer et al., 2008), thus resulting in a decrease of the soil bulk volume.

The soil bulk volume, however, is also subjected to changes according to water content (e.g. Boivin et al., 2006a). Therefore, it is recommended to sample the soil at a constant water content (Hakansson & Lipiec, 2000), to get rid of this issue. The soil pore size distribution, however, is modified by compaction (Schaeffer et al., 2008), which means that the water retention curve is no longer the same from control to compacted soil. Therefore, constant water content at sampling time is not enough to guarantee equivalent swelling of the soil, as can be seen on our results, and a standard matric potential would be a better criteria. Sampling close to water holding capacity is the recommended way to overcome this limitation. As we show, this is not always enough in forest soils where a sharp heterogeneity of the soil water content and matric potential is expected. Quantifying soil compaction with shrinkage analysis would be, therefore, recommended since it allows determining the soil bulk volume on the full water content and matric potential range, as can be seen in Figures 2 and 3, while allowing to focus on volumes targeted either on matric potential values or the limits of the shrinkage domains, independently from field conditions. Neglecting the spatial variations in moisture content may lead to wrong conclusions on either compaction or recovery rate estimation.

Moreover, several papers have recently shown that the soil pore size distribution and pore swelling properties also depend on the soil clay and/or organic carbon contents (e.g. Boivin et al., 2004, 2006a, 2009, Shaeffer et al., 2008). Schaeffer et al. (2008) showed that compaction could be quantified only after the standardization of the physical properties, e.g. bulk volume, with respect to soil organic carbon and clay content using covariance analysis. For instance, lower compaction values after forest traffic have already been found and have been attributed to the variability of organic carbon contents (Ampoorter et al., 2010).

Compaction of the studied soils

In this study, using the water content of the soil at sampling time as covariate allowed us to take into account the spatial variations in water content, which indirectly integrated part of the soil constituents' variability effect. This allowed us to estimate the yearly changes and changes with depth at lower cost than performing a full shrinkage and soil constituents analysis. A further study should discuss whether performing such full analysis would give more meaningful results.

The very large *w*-*V* values observed in the 2009 sampling at CA site could be due to changes in the soil porosity from 2009 to 2011. However, larger *w* and *V* in May 2009 than in February 2011 is not likely to occur as no additional compaction occurred, while natural recovery was expected. We suspect that the application of the soil-core method on the waterlogged soils could have induced some artefacts. When hammering the steel cylinders into the soil the water content was probably not at equilibrium, particularly in the T-treatment, and free drainage water was filling the coarser pores. The uncompressible water was then submitted to large positive pressure, which may have artificially expanded the topsoil volume. Heavy traffic has been found to increase waterlogging (Herbauts et al., 1996), therefore the cylinder sampling methodology may not be adequate for these situations. The plastic bag method (Boivin et al., 1990) allows determining the soil bulk volume on undisturbed soil clods of any shape without the need of hammering a cylinder. This method is, therefore, to be recommended in such case, since it would prevent this kind of artefact, while allowing easy equilibration to a target matric potential prior to volume measurement, and is less time consuming than the cylinder method.

Also, the effect of *w*-differences on the volumes of the control and trafficked soils for each year was highly significant; the observed changes in *V* due to traffic decreased with increasing excess of water in the trafficked compared to the control.

Recovery of the soil specific volume

The present study is based on bulk soil volume changes only. Even though soil specific volume may show some recovery three to four year after forwarder traffic, the pore-size distribution and continuity of pores may not show the same recovery (Page- Dumroese et al., 2006). For example, Von Wilpert & Schaeffer (2006) identified the early recovery of soil structure only 15 to 25 years after compaction.

For both experimental sites, we observed a trend of soil specific volume recovery in the surface soil layer (0-10 cm) propagating downwards vertically, at least for the CA site. This result was in accordance with the results of Von Wilpert & Schaeffer (2006) in terms of rooting capacity and of Brais (2001) and Froehlich et al. (1985) in terms of bulk density. On the contrary Page- Dumroese et al. (2006) found that most of the North American Long-Term Soil Productivity sites showed largest bulk density recovery rates between 10 and 30 cm depth than in the surface layer.

The soil surface layer of the CA site is more acidic than is the AZ site surface layer (Table 1). Soil pH is an important factor controlling earthworm species and demography; low pH decrease earthworm biomass and diversity, especially below a pH in water of 4.5 for anecic and epigeic species (Potthoff et al., 2008). Therefore a better biological activity for the recovering of soil structure was expected in AZ compared to CA (Bottinelli et al., 2010). Nevertheless, the observed *V* recovery rate in the 0-10 cm soil layer was higher for the CA than for the AZ site. Three years after heavy traffic no residual effect could be observed in the CA site whereas a slight effect could still be detected four years after the forwarder traffic in AZ. Higher recovery rate for coarse- than for fine-textured soils was found by Brais (2001) and Page- Dumroese et al. (2006). The CA site has smaller clay content than the AZ site. Differences in clay mineralogy could also account for the different recovery rates of the two sites. Aggregation is influenced by clay mineralogy and was found to be enhanced in smectite-rich soils (Bronick & Lal, 2005), which is the case for CA but not for AZ. Moreover, clay type and content affect soil shrink-swell properties (Boivin et al., 2004) and the CA site showed more contrasted wetting-drying cycles between the C- and T-treatments than the AZ site (Fig. 6). The shrink-swell cycles induced by changes in water content are at the origin of structuration processes (Kay, 1998) and are more intense with swelling clays (Boivin et al., 2004). For instance, Pires et al. (2008) observed significant structural change due to repeated wetting and drying cycles resulting in an increase in the structural pore diameter. Therefore we assume that such physical processes were at the origin of the high recovery rate in the acidic CA site. The larger initial impact of forwarder traffic on the 0-10 cm soil layer of the AZ in comparison to the CA site could be an additional reason for the different recovery rates in the two sites (Page- Dumroese et al., 2006). However, Froehlich et al. (1985) found no significant influence of the initial compaction state on the bulk density recovery rate.

Conclusion

This study underlines the hazards of wrong interpretation associated with the assessment of compaction *via* bulk density measurements, either from place to place or over years. Though we tried to sample close to field capacity, working with bulk volume measurement led to contradictions in comparing the control and trafficked soils, or their evolution over years. Using the gravimetric water content as covariate allowed overcoming partly this limitation, though we could not appreciate neither the part of variance due to changes in soil constituents, nor the changes in pore size distribution regardless of the volumes. Some of the volumes and water content measured with the cylinder method were extremely large compared to the shrinkage curves measured on the undisturbed samples. We strongly suspect that hammering the cylinders in the soils near water saturation introduced some artefacts.

After using the water content as covariate, the changes in bulk volume of the soils from depth to depth and over years were interpreted. The initial compaction was larger at the AZ site than at the CA site. This latter fully recovered the control soil volume of the 0-10 cm soil layer after three years. We attribute the relative high recovery rate of the CA site to the smaller clay content together with the presence of smectite clays that is to physical processes based on shrink-swell properties rather than to biological activity, which is suspected to be low in the CA acidic soil.

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III

SOIL RESISTANCE TO PENETRATION FOLLOWING TO FOREST TRAFFIC AND ITS EVOLUTION WITHOUT MECHANICAL LOOSENING AS INFLUENCED BY PH AND LIME ADDITION.

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Abstract

Natural structure regeneration following heavy traffic is assumed to require several decades, especially in acidic forest soils. Our aim was to compare the recovery rate in soil resistance to penetration (Rp) between two compacted forest soils differing slightly in pH, and after soil liming for the most acidic of both soils. We measured Rp once a year in two experimental sites northeast France, during three to four years following forwarder traffic. The impact of heavy traffic on Rp depended on soil water content (WC) at the time of measurement. For the most acidic and silty site, no difference in Rp between treatments were observed when the soils were water-saturated (November 2008), whereas the impact of the wood-loaded forwarder was high when soils were close to wilting point (June 2011). For the other site, the impact of treatment on Rp could be detected whatever the soil moisture, even if the amplitude of the effect depended on the field campaign and depth considered. The difference in Rp response to compaction between the two sites may have originated from distinct sensitivity to hardsetting. Standardizing Rp values with regards to WC offered a great potential to analyze the evolution of the impact of heavy traffic with time. No recovery in Rp could be detected three to four years after soil compaction at both sites. Liming did not increase the regeneration of Rp to undisturbed levels over the study period even if it increased soil surface pH and base saturation.

Résumé

La régénération de la structure d’un sol soumis au passage d’engin à sa surface semble requérir plusieurs dizaines d’années, en particulier pour des sols forestiers acides. Notre objectif était de comparer la vitesse de restauration de la résistance à la pénétration (Rp) de deux sols forestiers tassés. Les deux sols différaient légèrement en termes de pH et le sol le plus acide a été amendé. Nous avons mesuré Rp une fois par an, pendant trois à quatre ans après le passage du porteur forestier. L’impact de la circulation du porteur sur Rp dépendait de la teneur en eau du sol (WC) au moment de la mesure. Sur le sol le plus acide et limoneux, aucune

différence de Rp n'était observée entre traitements quand le sol était saturé en eau (novembre 2008), alors qu'un fort impact du porteur pouvait être constaté quand le sol était proche du point de flétrissement permanent (juin 2011). Sur l'autre sol, l'impact du traitement sur Rp pouvait être observé quelles que soient les conditions d'humidité, même si son amplitude variait en fonction de la profondeur et de la campagne de mesure considérée. La différence de réponse de Rp à la compaction du sol entre les deux sites pourrait être liée à une sensibilité au 'hardsetting' contrastée. Normaliser les mesures de Rp par rapport aux mesures de WC permettait une analyse potentiellement plus pertinente de l'évolution temporelle de l'impact du porteur. Aucun retour à des niveaux de Rp non perturbés n'a pu être constaté sur les deux sites. L'amendement calco-magnésien n'a pas encore affecté la vitesse de restauration du paramètre Rp, même si son effet sur le pH et le taux de saturation de la CEC en cation est déjà visible en surface.

Introduction

Forest soils with lower bulk density (Bormann & Klaassen, 2008) and soil strength (Blanco-Canqui *et al.*, 2005) than agricultural or pasture lands are very vulnerable to external mechanical stresses exerted by heavy forest machinery. The use of vehicles to harvest wood has already been proven to affect soil physical properties irrespective of machine and soil types (Horn *et al.*, 2007). Direct impacts include an increase in soil bulk density (BD) and in soil mechanical resistance to penetration (Rp) (*e.g.*, Ampoorter *et al.*, 2007), but also a diminution of soil porosity in term of quantity and quality (connectivity of voids, pore size distribution) (*e.g.*, Herbauts *et al.*, 1996; Lister *et al.*, 2004). As indirect effect, the growth and health of the vegetation may be disturbed in trafficked soils as roots will need to develop more energy to prospect deeper soil layers and will probably have disturbed nutritional, aeration and hydric status (Carter *et al.*, 2007; Herbauts *et al.*, 1996; Whalley *et al.*, 1995). In forest ecosystems soil structure regeneration occurs through natural processes (wetting-drying cycles, freezing-thawing cycles and biological activity) as soil remediation through mechanical soil loosening and/or fertilization is not frequently used in forest management practices. Therefore soil recovery rate to undisturbed levels of structure and porosity is an important parameter to account for into planning of sustainable forest management. Especially for acid forest soils where biological improvement of soil structure is expected to be poor, as soil acidity decreases macro-fauna and its activity (Potthoff *et al.*, 2008). Impeded root growth and health may also reduce soil regeneration, as roots have been found to lower bulk density, increase aeration and increase aggregate stability (Lister *et al.*, 2004).

To assess the impact of heavy traffic, one of the most frequently used soil physical parameters is the physical resistance Rp (Carter *et al.*, 2007). Besides the direct relationship between change in total porosity and Rp, this parameter has been found to indicate the facility at which roots can penetrate and grow into the soil (Whalley *et al.*, 1995). Soil resistance to penetration is mostly influenced by soil structure, the nature of soil particles cementing materials (mostly clay, organic carbon content and oxides) and the changes of both soil characteristics with water content (WC) and stresses (To & Kay, 2005). Comparison of concurrent measurements of BD and Rp pointed out possible different impact and recovery time after heavy traffic for each parameter (Brais, 2001; Page-Dumroese *et al.*, 2006). It can be partly explained by the fact that the difference in Rp caused by difference in BD increases when WC decreases (Smith *et al.*, 1997). Besides a non linear relationship between Rp and BD was found (Ampoorter *et al.*, 2007; Vaz *et al.*, 2001; Whalley *et al.*, 2005), as for high Rp bulk density seems to be constant, independent of Rp.

Two experimental sites set up in two temperate forests north-east France were monitored for change and recovery in BD (Goutal *et al.*, 2011) and Rp following traffic by the same full-loaded forwarder. One site displayed a pH in water < 4.5 whereas the other had a pH > 4.5 for the 0-10 cm soil layer. We assumed that the more acidic site would show slower restoration rate as high soil acidity has been shown to limit soil microbial and anecic earthworm species. Therefore part of the acidic site was limed to test this hypothesis ([i] lime addition increased the Rp recovery rate). Two more hypotheses were tested in this study; [ii] forwarder traffic impacted significantly Rp and this impact decreased with time in the soil surface layer as was found for BD (Goutal *et al.*, 2011) and [iii] the impact depended on WC and it was necessary to standardize this impact with respect to WC in order to be independent from field conditions when measuring the effects of soil compaction and soil recovery on Rp.

Material and methods

Study sites

We worked on two experimental sites set up in the north-east of France and described in Goutal *et al.* (2011). Both sites were clear-cut over a 5 ha surface area, logs were removed using a cable yarding system to avoid soil disturbance and the same full-loaded forwarder drove afterwards for an equivalent of two passes (one back and one forth). The wheel tracks were adjacent to each other in order to create an equally levelled compacted area of 30 m × 50 m. Undisturbed plots of same surface area and adjacent to the trafficked ones were considered as control plots. The forwarder traffic took place in May 2007 and March 2008 in the site located in the 'Hauts-Bois' forest – Azerailles (48° 29′ 19″ N, 6° 41′ 43″ E), Meurthe et Moselle, and in the site located in the 'Grand Pays' forest – Clermont en Argonne (49° 06′ 23″ N, 5° 04′ 18″ E), Meuse, respectively. The tyres were inflated to the same pressure (360 kPa) and the forwarder weighted 23 and 17 t in the Azerailles (AZ) and Clermont en Argonne (CA) sites respectively because the CA site was wetter at the time of traffic than the AZ site. The climate of the region is characterised by a 30-year mean annual temperature of 9 °C (AZ) to 9.5 °C (CA) and a 30-year annual precipitation of 900 mm (AZ) to 1000 mm (CA). The sites have an elevation of 270 m (CA) and 300 m (AZ) above mean sea level, with a maximum of

8 m variation of level through each site. In autumn 2007 (AZ) and 2008 (CA) the entire site surface area was planted with sessile oak (*Quercus petraea* L.) at a density of 1600 seedlings ha^{-1} for each site.

The soil of both sites is classified as Luvisol (ruptic) according to IUSS Working Group WRB (2006) and is developed on a silt loam layer approximately 50cm thick laying on a clayey material (weathering of a Keuper marl for the AZ site and weathering of Cenomanian gaize rock for the CA site). The soil of the CA site was more acidic than the one of AZ, with a pH in water of the 0-10 cm soil layer of 4.4 and 4.8 respectively. Soil pH is an important factor controlling earthworm species and demography; low pH decrease earthworm biomass and diversity, especially below a pH in water of 4.5 for anecic and epigeic species (Potthoff *et al.*, 2008). Therefore we expected a higher natural soil structure regeneration rate at AZ than at CA and tested the effect of liming on soil structure regeneration rate at CA. In September 2008 we applied manually 1.3 t ha^{-1} of dolomite (36% CaO, 24% MgO) and 400 kg ha^{-1} of potassium sulfate (50% K2O, 17% S). Each site was divided into three blocks, each treatment was randomly attributed into each block. Each plot (block × treatment) measured 30 × 50 m. In AZ, two treatments were considered: trafficked (T) and control (C). In CA, we considered two additional treatments; C, T, control amended (A) and trafficked amended (TA). Soil texture differed between sites but not between treatments, except for below 20 cm depth at CA (Figure 1). Soil surface organic carbon content (SOC) was not affected by the forwarder traffic and differed only slightly between sites (Figure 1). Soil water retention differed between sites and was affected by treatment, but mostly between saturation and field capacity (Figure 2, Table 1). Soil liming resulted in an increase in soil pH and base saturation (Figure 3).

Table 1 Soil gravimetric water contents determined at two water potentials (-0.33 and -15 bar) in May 2009 in both treatments (C, control; T, trafficked) and sites. Measurements were carried out using ceramic pressure plate extractors on approximately 1 cm height and 5 cm diameter cylinders; mean and standard deviation in brackets (12 replicates per water potential, depth, treatment and site)

	AZ				CA			
	-0.33		-15		-0.33		-15	
Depth	C	T	C	T	C	T	C	T
5 cm	0.30 (0.05)	0.31 (0.05)	0.13 (0.05)	0.12 (0.04)	0.30 (0.06)	0.34 (0.05)	0.14 (0.06)	0.11 (0.04)
15 cm	0.24 (0.04)	0.23 (0.02)	0.10 (0.03)	0.10 (0.04)	0.23 (0.02)	0.22 (0.02)	0.08 (0.01)	0.09 (0.05)
25 cm	0.23 (0.04)	0.22 (0.02)	0.11 (0.03)	0.11 (0.03)	0.22 (0.03)	0.21 (0.05)	0.12 (0.04)	0.10 (0.04)
35 cm	0.23 (0.08)	0.23 (0.03)	0.12 (0.03)	0.13 (0.05)	0.23 (0.03)	0.22 (0.01)	0.13 (0.03)	0.11 (0.02)

Figure 2 Impact of the forwarder traffic (C, control; T, trafficked) on the relationship between soil gravimetric water content (WC) and water potential at both sites (AZ, Azerailles; CA, Clermont en Argonne). Measurements were carried out using the Wind's evaporation method (Wind, 1968) for water potentials varying between -5 and -80 cmH_2O and on approximately 7 cm height and 20 cm diameter cylinders; 3 replicates per depth and treatment; sampling in May 2008 at AZ and in May 2009 at CA (one year after soil compaction)

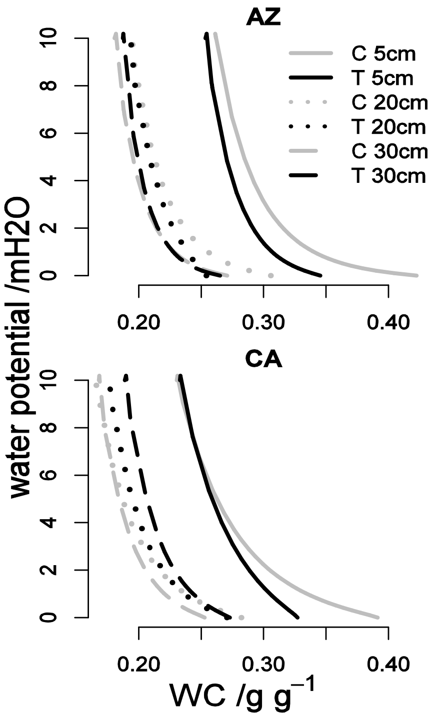


Figure 1 Soil organic carbon (SOC), clay and silt content in the different sites (AZ, Azerailles; CA, Clermont en Argonne) and treatments (C, control; T, trafficked). Mean and standard errors are plotted as error bars. 30 replicates per depth and treatment for SOC determination, 6 and 12 replicates per depth for the C- and T- treatment respectively for determination of clay and silt contents (May 2009 for both sites)

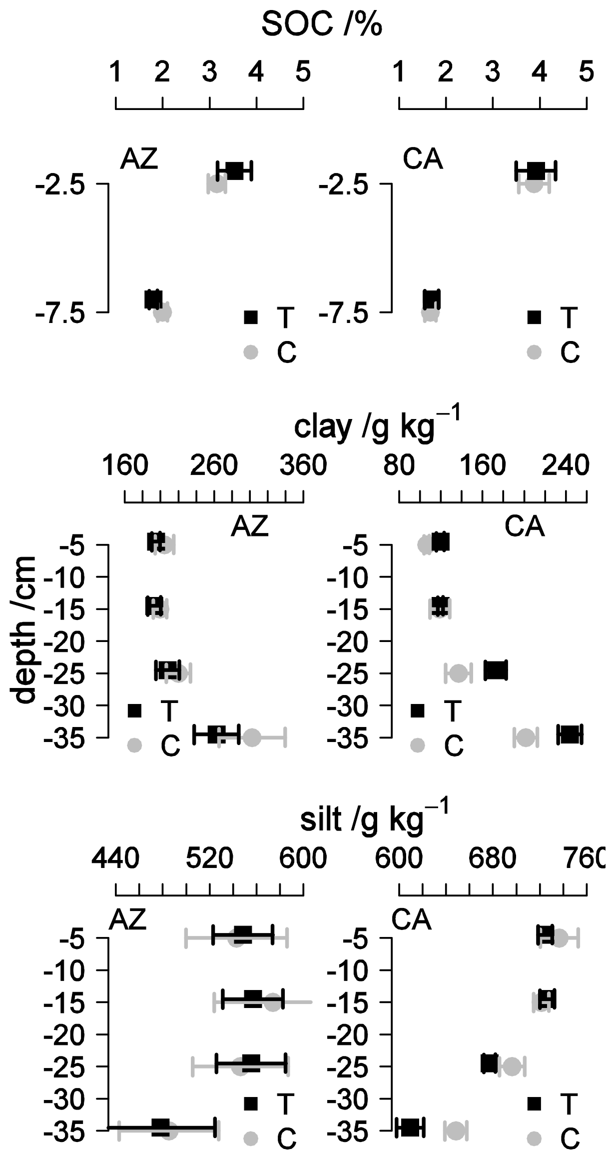
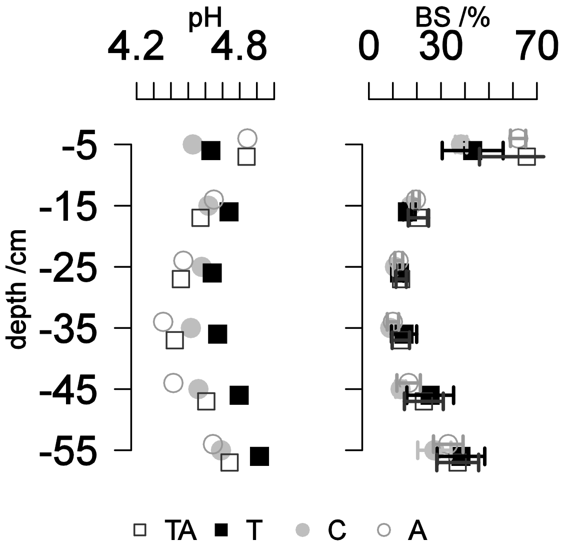


Figure 3 Soil base saturation (BS) and pH in water as a function of depth and treatment (C, control; T, trafficked; A, control amended; TA, trafficked amended) at the CA (Clermont en Argonne) site. Mean and standard errors as errors bars; 3 replicates per treatment and depth (sampling in May 2010)



Measurements of Rp

Soil penetration resistance (Rp) was measured once a year using a dynamic cone (cone basal surface area of 2 cm²) penetrometer with variable energy (PANDA ®, Sol Solution, Riom France, http://www.sol-solution.com/panda/58. Accessed: 26/09/2011). The device measures simultaneously the energy applied and the penetration of the cone into the soil for each blow. The value of Rp given by the device is the energy divided by the penetration. Therefore measurements were assumed to be operator-independent. Nevertheless we tried to limit penetration rate into the soil to a maximum of one cm increment per hammering all along the profile in order to obtain detailed Rp curves.

To compare the impact of traffic on Rp and the Rp recovery rates between sites, we always measured Rp of both sites within one week. Once a year we measured Rp on 30 profiles down to a depth of 70cm per site, block and treatment (i.e. 180 in AZ and 360 in CA), except in June 2011 where only 15 profiles per site, block and treatment were measured. Each time we measured Rp using the same grid of 30 points systematically split over the entire 30m × 50m area of each block × treatment plot. In November 2008, five randomly chosen Rp profile were sampled for gravimetric WC determination down to 40 cm depth with a 10 cm depth increment. In June 2009, 2010 and 2011, we measured WC on the same profile and in June 2011, 10 additional Rp profile were also sampled for WC determination. Samples were dried for 48h hours at 105°C. Soil WC was calculated on a dry-weight basis.

Statistical analysis

Even if we tried to push the penetrometer rod down into the soil with a depth increment of one cm, we never achieved an exact and constant one cm depth increment because of high field heterogeneity (roots, gravel, low or high density locations). Therefore we used the mean of the Rp measurements corresponding to the same 5 cm depth increment (or 10 cm depth increment in the case of comparison with WC measurements).

Within each field campaign:

For each field campaign considered, we assumed that Rp values were auto-correlated as a function of depth within each profile measured (30 profiles per treatment and block, except in June 2011 where only 15 profiles per treatment and block were measured). Therefore we had to test for the presence of an auto-correlation structure of the residuals of the statistical model considered:

$$Rp = Tr + D + Tr : D + \mathrm{Pr} + e$$

where Tr is the effect of treatment (two levels: C and T or A and TA), D is soil depth (centre of the 5 cm high soil layers considered: 2.5, 7.5, 12.5, 17.5, 22.5, 27.5, 32.5, 37.5, 42.5, 47.5, 52.5, 57.5, 62.5, 67.5), Pr is the soil profile (random effects) and e are the model residuals with an auto-regressive structure of lag 1 with a depth covariate within each soil profile.

Between field campaign:

For each soil layer (0-10, 10-20, 20-30 and 30-40 cm):

$$Rp = WC + Ti + Tr + WC:Ti + WC:Tr + Ti:Tr + WC:Tr:Ti + \mathrm{Pr} + \varepsilon$$

where WC is soil gravimetric water content (g g^{-1}), Ti is the effect of the date considered (four levels: Nov. 08, June 09, June 10 and June 11), ε are the model residuals (the auto-correlation structure of the residuals with a time covariate within each profile was not significant whatever the soil layer considered, therefore we removed it).

The effect of soil profile was considered as a random effect inducing random variations around the fixed and reproducible effects. The linear mixed models were fit using the Maximum Likelihood (ML) procedure of the R software. Selection of the best model was based on the distribution of the residuals, the Aikake information criterion (AIC), Bayesian information criterion (BIC) and the likelihood ratio Chi-2 test (LRT) for nested models. In the following sections, the term significant refers to a significance at a α-level of 0.05 probability.

Results

Within each field campaign, effect of treatment and depth on Rp

Soil resistance to penetration had a log-normal distribution in both sites and at all field campaign. At AZ the interaction between the effects of treatment and depth was significant for most sampling campaign (three out of four) whereas at CA it was the opposite, this interaction was only significant for one measurement campaign out of four (Table 2). However the variability in Rp values was high and the quality of the statistical models was poor despite the high number of replicates (Table 2, Figure 4). We observed that the statistical estimation of the intercept which represent the Rp value for the control or control amended at the soil surface, varied considerably from one campaign to another (Table 2). Especially in November 2008 the natural logarithm of Rp was estimated as non significantly different from zero at the soil surface of the control plots at CA whereas it was estimated as superior to one in June 2011. Besides at CA the impact of heavy traffic on Rp (*e.g.*, increase of 14% relative to the value in the C-treatment at 5-10 cm depth) was not significant whatever the depth in November 2008, whereas seven months latter it was significant throughout the entire soil profile (0-70 cm) (*e.g.*, increase of 32% relative to the value in the C-treatment at 5-10 cm depth).

Figure 4 Natural logarithm of soil resistance to penetration (Rp in Mpa) measured (dots) and modelled (lines) as a function of treatment and depth. Measurements in June 2010 with 90 replicates (profiles) per treatment; AZ, Azerailles; CA, Clermont en Argonne; C, control; T, trafficked; A, control amended; TA, trafficked amended.

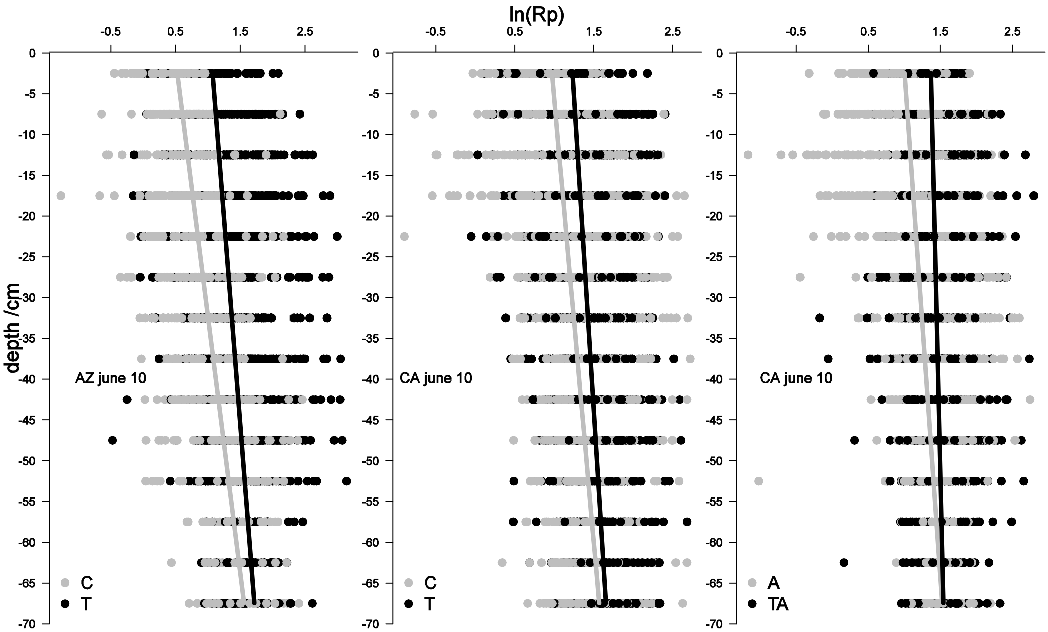


Table 2 Linear mixed models explaining soil resistance to penetration (Rp in MPa) variations as a function of treatment (*Tr*, two levels: control, C and trafficked, T or control amended, A and trafficked amended, TA) and depth (D, 14 levels: increments in soil layer thickness of 5 cm, from the soil surface to 70 cm depth). Selection of the significant effects was based on the AIC, BIC, and p-value of the likelihood ratio Chi-2 test for nested models after analysis of the distribution of the residuals. Only the coefficients of the significant effects (α-level of 0.05) are given; n.s. corresponding to a non-significant effect

	Ln(Rp)[a]	μ[b]	*Tr*:*D*	*Tr*	*D*	AIC	BIC	SD *Pr*	RSD	Phi AR(~*D*|*Pr*)
AZ nov.08	√	-0.26 (0.05)	-0.005 (0.001)	0.68 (0.07)	0.023 (0.001)	2873	2913	0.30	0.48	0.88
AZ june09	√	0.26 (0.04)	n.s.	0.36 (0.04)	0.017 (8×10^{-4})	2566	2601	2.0×10^{-4}	0.55	0.93
AZ june10	√	0.49 (0.05)	-0.006 (0.002)	0.56 (0.07)	0.016 (0.001)	1550	1590	1.1×10^{-4}	0.5	0.94
AZ june11	√	0.71 (0.07)	-0.011 (0.002)	0.59 (0.09)	0.019 (0.001)	710	746	1.8×10^{-4}	0.47	0.95
CA nov.08, C *vs.* T	√	n.s.	n.s.	n.s.	0.026 (5×10^{-4})	2378	2401	0.10	0.54	0.93
CA june09, C *vs.* T	√	0.31 (0.04)	n.s.	0.36 (0.04)	0.019 (7×10^{-4})	1780	1814	1.4×10^{-4}	0.46	0.93
CA june10, C *vs.* T	√	0.95 (0.04)	-0.003 (0.001)	0.27 (0.06)	0.009 (0.001)	1016	1057	5.0×10^{-5}	0.44	0.95
CA june11, C *vs.* T	√	1.28 (0.07)	n.s.	0.25 (0.08)	0.008 (0.001)	646	678	0.08	0.55	0.96
CA nov.08, A *vs.* TA	√	0.24 (0.04)	n.s.	0.18 (0.04)	0.027 (7×10^{-4})	1950	1985	1.6×10^{-4}	0.48	0.93
CA june10, A *vs.* TA	√	0.98 (0.04)	-0.006 (0.002)	0.38 (0.06)	0.008 (0.001)	1134	1173	4.6×10^{-5}	0.44	0.94
CA june11, A *vs.* TA	√	1.55 (0.07)	-0.006 (0.002)	0.36 (0.1)	0.008 (0.002)	611	647	6.8×10^{-5}	0.49	0.96

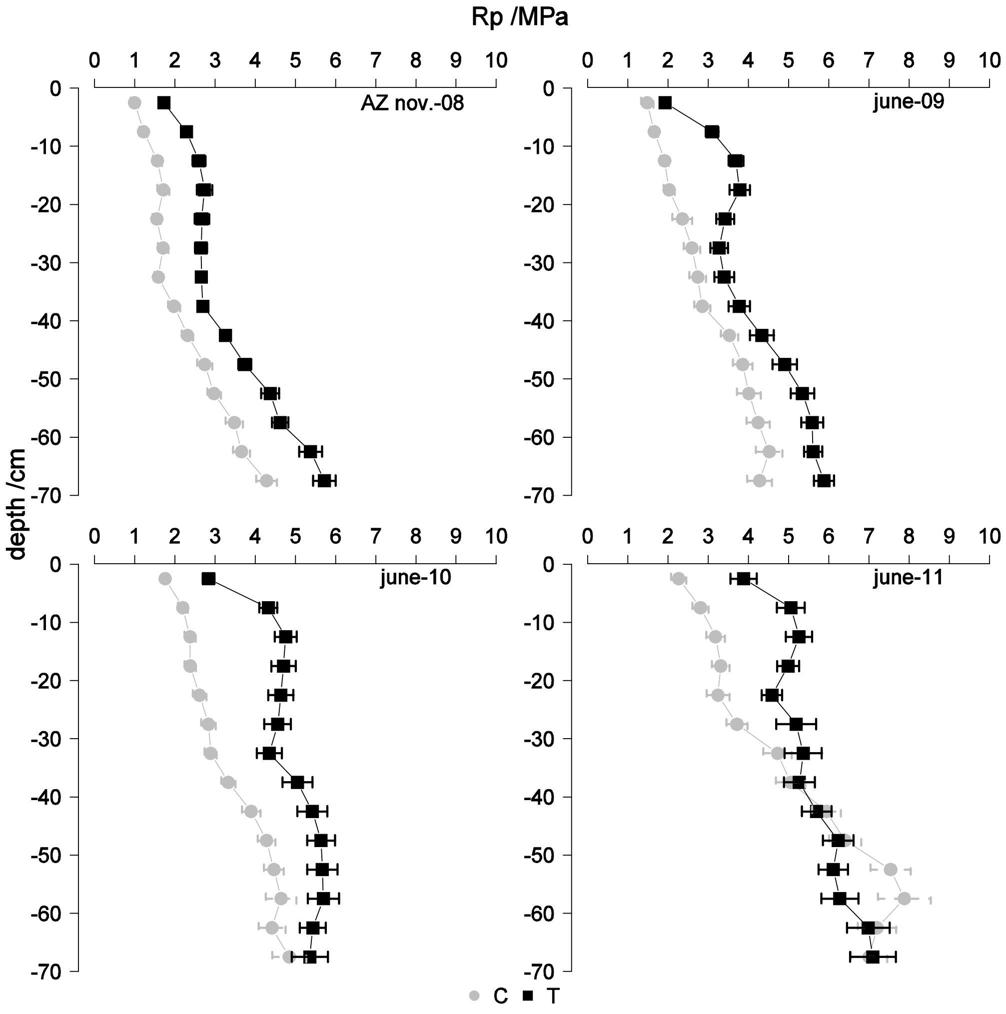
[a] log-transformation of Rp needed with respect to the distribution of the residuals

[b] coefficients and standard errors (in brackets) associated to the following linear mixed model:

$Rp = \mu + Tr + D + Tr:D + \mathrm{Pr} + e$, with μ being the model intercept (C- or A-treatment at the soil surface), *Tr*:*D* being the interaction term between the *Tr* and *D* effects, *Pr* being the soil profile (inducing random variations around the fixed effects; 30 replicates × two treatments × three blocks for each campaign, except in 2011 where only 15 replicates × two treatments × three blocks were measured), and *e* being the residuals with an auto-regressive structure of lag 1 with a *D* covariate within each *Pr.* AIC, Aikake information criterion; BIC, Bayesian information criterion; SD *Pr*, standard deviation of the random effect; RSD, residual standard deviation; Phi AR(~*D*|*Pr*), value of the lag 1 autocorrelation.

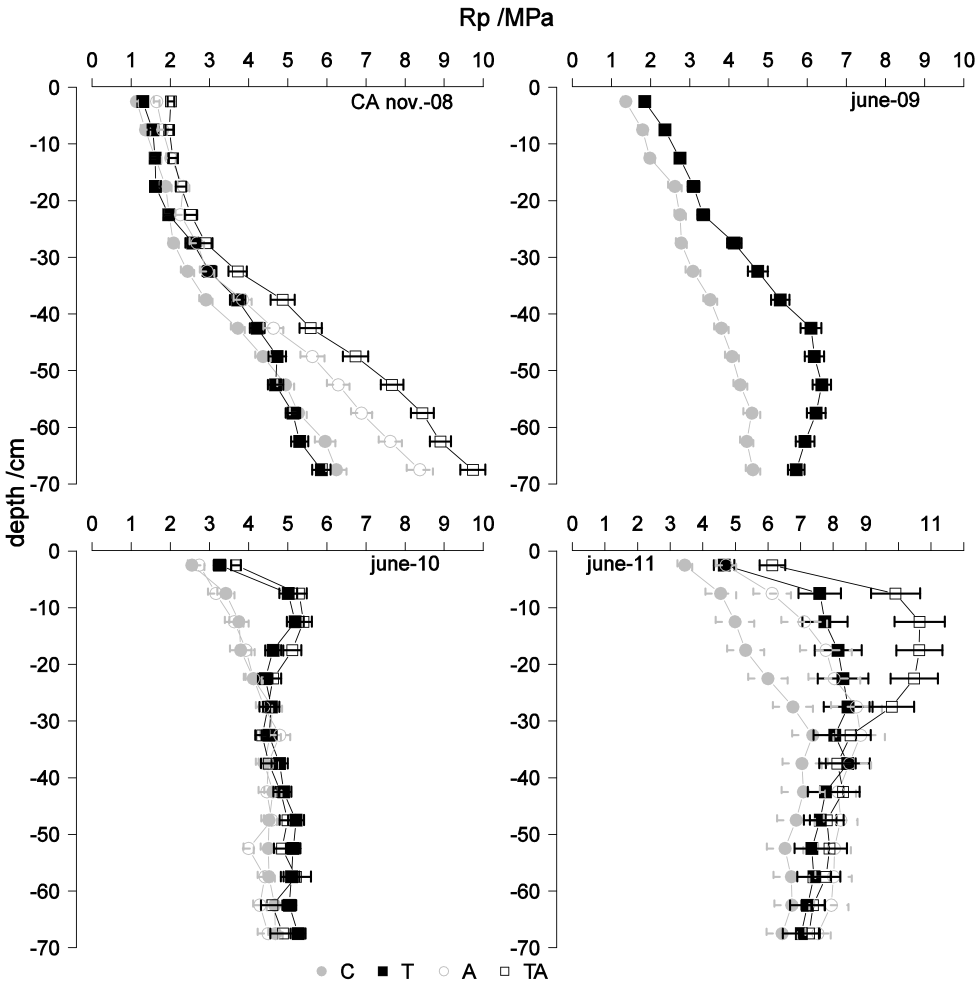
At AZ the impact of treatment on Rp was strong and obvious for every field campaign (Figure 5) even if the magnitude of the impact at a given depth varied with the campaign considered (*e.g.*, increase of 87% and 97% relative to the value in the C-treatment at 5-10 cm depth in November 2008 and June 2010 respectively).

Figure 5 Soil resistance to penetration (Rp) as a function of depth, sampling time and treatment (C, control; T, trafficked), measured at the AZ (Azerailles) site (90 replicates per treatment, except in June 2011 where only 45 profiles were measured per treatment). Mean and standard errors as errors bars



On the contrary, at CA and at a given depth, the impact of heavy traffic and lime addition on Rp varied greatly with the campaign considered (Figure 6). The A- (TA-treatment respectively) had higher Rp than the C- (T-treatment respectively) except in June 2010. This difference was exacerbated for the deepest soil layers (40-70 cm) in November 2008 and for the upper soil layers (0-30 cm) in June 2011. Soil strength was higher at CA than at AZ (Figures 5 and 6, intercepts of Table 2).

Figure 6 Soil resistance to penetration (Rp) as a function of depth, sampling time and treatment (C, control; T, trafficked; A, control amended; TA, trafficked amended), measured at the CA (Clermont en Argonne) site (90 replicates per treatment, except in June 2011 where only 45 profiles were measured per treatment). Mean and standard errors as errors bars



Between field campaign, effect of WC and treatment on Rp

The effect of WC on Rp was significant (Table 3). Treatment did not influence significantly the relationship between the natural logarithm of Rp, ln(Rp), and WC. The relationship between ln(Rp) and WC differed between sites; the slope of the relationship between ln(Rp) and WC varied between -1.9 and -5 at AZ and between -1.4 and 8 at CA (Figure 7, Table 3). At both sites the relationship between ln(Rp) and WC was less steep in the surface (0-10 cm) than in the underlying soil layers.

In November 2008, the effect of treatment was significant at CA whatever the depth considered if we accounted for WC variations. The interaction between the effect of treatment and sampling time was not significant except at CA and at 15 cm depth. The latter significant interaction was due to the fact that the impact of treatment at that depth was really low in November 2008 in comparison to other sampling dates. Yet the coefficients for the interaction term did not differ between the three remaining sampling dates (June 09, 10 and 11). At CA, the change in Rp caused by soil amendment in the deepest soil layers in November 2008 could not be linked to WC variations. Indeed we only measured WC from soil surface to 40 cm depth where WC did not differ between C- and A- or between T- and TA-treatments (Table 4).

Table 3 Linear mixed models explaining soil resistance to penetration (Rp in MPa) variations per depth (4 soil layers) as a function of treatment (*Tr*, two levels: control, C and trafficked, T or control amended, A and trafficked amended, TA), water content (WC in g g^{-1}) and date (*Ti*, four levels: November 2008, June 2009, 2010 and 2011). Selection of the significant effects was based on the AIC, BIC and p-value of the likelihood ratio Chi-2 test for nested models after analysis of the distribution of the residuals. Only the coefficients of the significant effects (α-level of 0.05) are given; n.s. corresponding to a non-significant effect.

	μ[a]	WC	June 09	June 10	June 11	Tr	Interactions	AIC	BIC	SD Pr	RSD
AZ 5 cm	0.7 (0.3)	-1.9 (0.7)	0.3 (0.1)	0.4 (0.1)	0.7 (0.1)	0.52 (0.08)	n.s.	237	262	0.18	0.42
AZ 15 cm	1.4 (0.3)	-3 (1)	0.2 (0.1)	0.3 (0.1)	0.5 (0.1)	0.39 (0.09)	n.s.	225	251	0.20	0.40
AZ 25 cm	1.9 (0.3)	-5 (1)	0.3 (0.1)	0.2 (0.1) n.s.	0.4 (0.1)	0.31 (0.08)	n.s.	245	270	0.10	0.45
AZ 35 cm	1.8 (0.3)	-5 (1)	0.4 (0.1)	0.4 (0.1)	0.8 (0.1)	0.16 (0.07)	n.s.	209	234	0.13	0.40
CA 5 cm, C *vs.* T	0.7 (0.3)	-1.4 (0.6)	0.2 (0.1) n.s.	0.7 (0.1)	0.9 (0.2)	0.35 (0.07)	n.s.	257	283	0.04	0.47
CA 15 cm, C *vs.* T	2.7 (0.4)	-8 (1)	-0.1 (0.2) n.s.	0.4 (0.2) n.s.	0.1 (0.2) n.s.	-0.6 (0.2)	June09:Tr, 1.2 (0.3); June10:Tr, 0.7 (0.3); June11:Tr, 1.0 (0.2)	303	338	2.7×10^{-5}	0.53
CA 25 cm, C *vs.* T	2.5 (0.4)	-8 (1)	0.5 (0.1)	0.7 (0.1)	0.7 (0.2)	0.17 (0.08)	n.s.	264	289	0.09	0.47
CA 35 cm, C *vs.* T	2.9 (0.3)	-8 (1)	0.3 (0.1)	0.4 (0.1)	0.5 (0.1)	0.16 (0.07)	n.s.	244	270	3.2×10^{-5}	0.46
CA 5 cm, A *vs.*TA	1.2 (0.3)	-1.5 (0.6)		0.2 (0.2) n.s.	0.6 (0.2)	0.42 (0.08)	n.s.	191	212	0.06	0.44
CA 15 cm, A *vs.*TA	2.4 (0.4)	-6 (1)		-0.02 (0.2) n.s.	0.3 (0.2)	0.4 (0.1)	n.s.	227	247	0.36	0.39
CA 25 cm, A *vs.*TA	3.0 (0.4)	-8 (1)		-0.1 (0.2) n.s.	0.05 (0.2) n.s.	0.18 (0.09)	n.s.	183	204	0.27	0.36
CA 35 cm, A *vs.*TA	2.2 (0.4)	-3 (2)		0.02 (0.1) n.s.	0.4 (0.2)	0.009 (0.09) n.s.	n.s.	166	187	0.31	0.31

[a] coefficients and standard errors (in brackets) associated to the following linear mixed model: $\ln(Rp) = \mu' + WC + Ti + Tr + \text{int}\,eractions + \text{Pr} + \varepsilon$, with μ' being the model intercept (C- or A-treatment at a WC of 0 and during nov. 08), and ε being the residuals (the auto-correlation of the residuals as a function of time within *Pr* was not significant). AIC, Aikake information criterion; BIC, Bayesian information criterion; SD *Pr*, standard deviation of the random effect (Pr); RSD, residual standard deviation.

Figure 7 Soil resistance to penetration (Rp) measured and modelled (lines) as a function of treatment, sampling time and water content (WC) in the 0-10 cm soil layer. For each graph of Rp as a function of WC and treatment (left side) the lines drawn correspond to the relationship between Rp and WC for each treatment in June 2009. For each graph of Rp as a function of WC and sampling time (right side) the line drawn correspond to the C-treatment. AZ, Azerailles; CA, Clermont en Argonne; C, control; T, trafficked; A, control amended; TA, trafficked amended

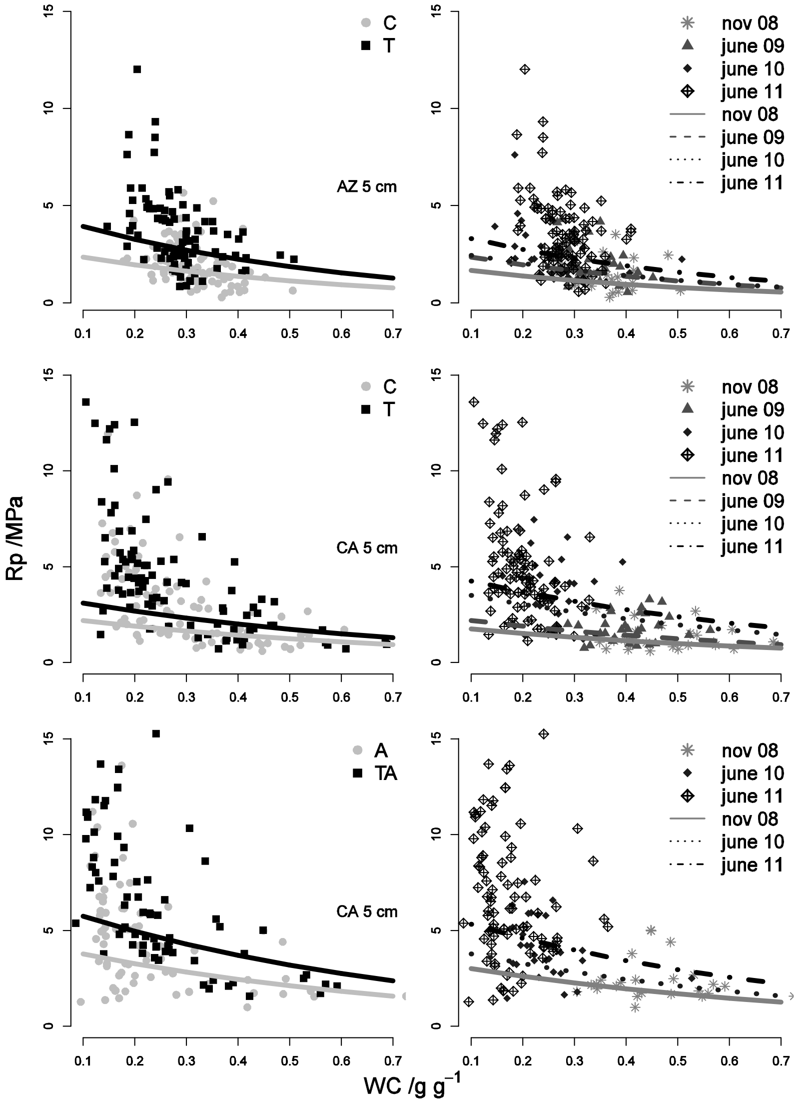


Table 4 Soil gravimetric water contents (in g g^{-1}) (means and standard erros in brackets; 15 replicates per treatment, depth and date, except in June 11: 45 replicates) measured concurrently to soil resistance to penetration. AZ, Azerailles ; CA, Clermont en Argonne ; C, control; T, trafficked

	Nov. 08		June 09		June 10		June 11	
AZ								
	C	T	C	T	C	T	C	T
5 cm	0.38 (0.01)	0.34 (0.02)	0.35 (0.01)	0.31 (0.01)	0.27 (0.01)	0.27 (0.02)	0.29 (0.006)	0.27 (0.008)
15 cm	0.30 (0.008)	0.25 (0.009)	0.28 (0.008)	0.24 (0.006)	0.26 (0.008)	0.23 (0.01)	0.25 (0.006)	0.21 (0.004)
25 cm	0.29 (0.006)	0.25 (0.007)	0.28 (0.007)	0.24 (0.007)	0.23 (0.007)	0.22 (0.01)	0.22 (0.005)	0.22 (0.005)
35 cm	0.26 (0.007)	0.23 (0.006)	0.25 (0.005)	0.23 (0.006)	0.23 (0.007)	0.23 (0.01)	0.22 (0.006)	0.22 (0.005)
CA								
	Nov. 08		June 09		June 10		June 11	
	C	T	C	T	C	T	C	T
5 cm	0.47 (0.02)	0.49 (0.03)	0.40 (0.02)	0.41 (0.02)	0.26 (0.01)	0.25 (0.02)	0.21 (0.007)	0.19 (0.007)
15 cm	0.29 (0.008)	0.26 (0.01)	0.25 (0.007)	0.27 (0.006)	0.22 (0.006)	0.20 (0.005)	0.18 (0.007)	0.16 (0.004)
25 cm	0.27 (0.004)	0.26 (0.01)	0.25 (0.004)	0.25 (0.005)	0.21 (0.008)	0.21 (0.005)	0.18 (0.005)	0.17 (0.005)
35 cm	0.24 (0.005)	0.24 (0.007)	0.23 (0.005)	0.23 (0.003)	0.22 (0.007)	0.21 (0.004)	0.18 (0.004)	0.18 (0.005)
	A	TA	A	TA	A	TA	A	TA
5 cm	0.47 (0.03)	0.44 (0.03)			0.24 (0.01)	0.24 (0.02)	0.16 (0.004)	0.19 (0.01)
15 cm	0.31 (0.009)	0.29 (0.01)			0.20 (0.005)	0.20 (0.003)	0.14 (0.004)	0.14 (0.006)
25 cm	0.29 (0.005)	0.29 (0.01)			0.20 (0.007)	0.21 (0.003)	0.14 (0.003)	0.14 (0.005)
35 cm	0.25 (0.005)	0.26 (0.007)			0.20 (0.004)	0.21 (0.003)	0.16 (0.003)	0.15 (0.005)

From 0 to 40 cm depth, in June 2011, WC was lower in the A- than in the C- and in the TA- than in the T-treatment, whereas, the differences in WC between treatments were only slight in June 2010 (Table 4). These differences in WC were well related to a higher Rp in the A- than in the C- and in the TA- than in the T-treatment from 0 to 40 cm depth in June 2011 and not in June 2010 (Figure 6). From one Rp measurement campaign to the next one, WC decreased more at CA than at AZ (Table 4). At AZ, Rp measurements took place when soils were close to field capacity (June 2010 and 2011) or close to saturation (November 2008 and June 2009) (Figure 2, Tables 1 and 4). At CA, soils were close to saturation in

November 2008 and June 2009, close or above field capacity in June 2010 and close to the permanent wilting point in June 2011. Besides at CA in November 2008 and June 2009, WC measured close to the Rp profiles were superior to the WC found at saturation (Figure 2) in the 0-10 cm soil layer. This discrepancy could originate from the slightly different sampling depths between the Rp campaign and the cylinders to determine soil water retention. Indeed we did not exclude the upper and more organic two cm while sampling for WC during Rp measurement campaign whereas for the determination of soil water retention these more organic upper cm were excluded. Under previous forest cover, the humus forms varied from Mull mesomull at AZ to Mull dysmull and moder at CA. After the clear-cut, most of the litter layer disappeared, yet at CA the upper two centimetres were mostly organic materials whereas at AZ this organic upper layer did not exist. Therefore the exclusion of the uppermost mineral soil layer at CA may have lead to greater difference than at AZ concerning the comparison of measurements.

Discussion

Assessment of change in Rp and recovery trend

For most field campaign we observed an effect of treatment on Rp even at 70 cm depth although the effect of treatment on BD could not be detected below 50 cm depth (Goutal *et al.*, 2011). Nevertheless the variability in Rp values was high. We tried to minimize the effects of variability in soil characteristics by measuring each time Rp in the three blocks, on the same grid covering the entire block surface area. Therefore we captured most of the variability in soil characteristics by taking into account the random effect of position and the structure of the residuals due to repeated measurements into our statistical analysis. Yet, assessment of the effect of treatment on Rp varied considerably from one field campaign to another especially at CA, despite the high number of soil profiles measured for Rp at both sites. The comparison with other study with regards to the impact of heavy machinery traffic on Rp of forest soils was difficult. Indeed we found an impact of soil compaction on

Rp varying considerably over the entire measurement period (2008-2011) and depths (0-70 cm) (Figures 5 and 6). According to the high variability in Rp values, especially as a function of the measurement period, no clear trend in Rp changes since the forwarder traffic could be detected while considering raw data. However, we found that the effect of heavy traffic on Rp was still significant in June 2011, indicating poor restoration at both sites.

Soil organic carbon content and texture are the main factors influencing Rp besides BD and WC (Smith *et al*., 1997; To & Kay, 2005). Treatment application had no significant effect on soil organic carbon and texture (Figure 1), therefore between treatments variability could mainly be explained by BD and WC or water potential variations. Page-Dumroese *et al.* (2006) pointed out the need to measure WC when assessing change in impact of heavy traffic on Rp with time. For example, the effect of treatment on Rp was more obvious when soils were dry like in June 2011 as already shown by Smith *et al*., (1997). Besides standardizing Rp with respects to WC also partly normalize this parameter with respects to other soil characteristics (Goutal *et al*., 2011). In our study, the effect of WC on Rp was significant, and accounting for WC variations made the effect of treatment become significant in November 2008 at CA in comparison to the analysis of bulk Rp values without covariates.

Differences in the relationships between Rp and WC for the two sites may be explained by the different clay contents and nature, and/or by the different BD (To & Kay, 2005). As we had at both sites more than 20 % clay + silt, our results agreed with the study of Smith *et al*., 1997; increasing clay content by 10% (AZ>CA) reduced the rate at which Rp increased as WC decreased (Table 3). Besides the compacted soils at CA showed a hard-set soil behaviour, whereas the compacted soil at AZ did not. Indeed when the CA soils were wet, the T-treatment had as low or lower Rp than the C-treatment, whereas the differences in Rp between the two treatments were high when the soils were dry. The occurrence of hardsetting behaviour may have been linked to soil compaction at CA (Fabiola *et al*., 2003). Yet soil compaction did not imply soil hardsetting behaviour at AZ, probably because the CA site was more sensitive to hardsetting as it contained simultaneously more silt (Figure 1) and less Fe(III) oxides (Breuer & Schwertmann, 1999) than the AZ site. Indeed for the 0-50 cm soil layer (silt loam layer), the amount of free Fe (Mehra & Jackson, 1960) was of 7.6 g kg^{-1} at CA, whereas it was of 21.5 g kg^{-1} at AZ. Besides, the clay minerals composition of the CA site included swelling clay minerals (smectite), unlike the AZ site.

The relationship between Rp and WC was only significantly shifted consequently to forwarder traffic as the interaction between the effect of WC and treatment was not significant. The lack of effect of soil compaction on the slope of the relation between Rp and WC may be due to the log-transformation of the data. Indeed when Rp values were not transformed (Figures 5 and 6), high WC (November 2008) corresponded to slight differences in Rp and low WC (June 2011) meant high differences in Rp between compacted and non compacted soil. Furthermore a compensatory phenomenon could have occurred between an increase in BD and a change in pore size distribution. It is probable that we would have observed an influence of soil compaction on the relationship between water potential and Rp (To & Kay, 2005; Whalley *et al.*, 2005). Indeed same WC in both treatments meant different water potential in particular between field capacity and water saturation (Figure 2).

As the range in WC measured differed from one Rp campaign to the other, the relationship between Rp and WC could not be perfectly characterized for the entire WC range on each site (over the four campaigns). Indeed the amplitude in WC values for each campaign was narrow in comparison to the total range of the combined campaigns, especially at CA. By adjusting a WC-Rp relationship for one campaign we were limited in terms of extrapolation of this relationship to the other campaigns. In particular, the relationship between Rp and WC seemed to be steeper when considering the entire data set (the four sampling dates) in comparison to the fit done for each individual campaign (Figure 7). Yet because we studied the effect of time since soil compaction (soil regeneration), we could not adjust the relationship between Rp, WC and treatment without taking into account the effect of time, even if it would have allowed for a better fit. Therefore we could adjust Rp measurements with respects to WC (Table 3) but the remaining effect of the sampling date could originate from the fact that we did not calibrate the WC-Rp relationship over the entire WC range and not only from an effect of time since compaction. We should have measured the relationship between Rp and WC over the entire WC range each year (Smith *et al.*, 1997) to robustly adjust Rp with regards to difference in soil conditions between sampling dates. Nevertheless the absence of significant time by treatment interaction while using WC as covariate contradicted the hypothesis of a beginning of Rp recovery despite the BD recovery in the 0-10 cm layer of both sites (Goutal *et al.*, 2011). Page- Dumroese *et al.* (2006) also observed some level of BD recovery following forest harvesting on a silt loam soil whereas no sign of Rp recovery could be detected.

The BD recovery observed at both sites was attributed to physical processes leading to soil cracking (freezing/thaw, wetting/drying). The increased capacity to form cracks following to soil compaction should have reduced the contribution of water potential to Rp (To & Kay, 2005), but we found the opposite at CA. Indeed according to To & Kay (2005) and the fact that we found evidence of soil cracks formation in the T-treatment, the difference in Rp due to soil compaction should have increased as the soil dried but at the dryer end should have remained constant. Yet when soils were close to the wilting point at CA (June 2011), the differences in Rp due to compaction were the highest (Figure 6). Furthermore Goutal *et al.* (2011) observed in May 2011 no significant difference in BD between C- and T-treatment in the surface layer at CA. Therefore, even if a small difference in BD could lead to a large difference in Rp when the soil was dry (June 2011), it was unexpected that no difference in soil surface BD could still be concurrent to a significant difference in Rp. Another factor than BD and water potential may have caused the high difference in Rp despite low or absent change in BD following heavy traffic. It may well be the hardsetting behaviour of the compacted soil at CA causing BD and Rp to be high when the soil is dry even if BD has recovered when the soil is close to field capacity. However, we could not test this hypothesis as BD was measured at CA in May 2011 when soils were wetter than for the Rp measurements in June 2011.

Liming did increase Rp but did not increase Rp recovery rate

The impact of treatment in the amended plots was higher than in the non-amended plots at CA (Table 2). Yet Rp was also higher in the A- than in the C-treatment. Besides, the design of the study sites (random assignment of treatments within each blocks) ensured that the A- and TA-treatments did not correspond to another soil type or at least corresponded to systematic differences in Rp and that they were compacted the same way as the C- and T-treatment. Therefore the higher Rp found in the A- than in the C-treatment and in the TA- than in the T-treatment were attributed to differences in WC effectively observed (table 4). Ampoorter *et al.* (2011) found the opposite (*i.e.* soil liming decreased Rp). Yet as they did not measure WC or water potential concurrently to Rp assessment, we could not compare the effect of soil liming on Rp between those studies. Besides differences in clay mineralogy may explain further the divergence of our results (higher Rp at same WC values in the A- than the C-treatment and in the TA- than in the T-treatment, see Figure 7) with theirs about the effect of calcium addition on soil structural stability. Indeed Wuddivira & Camps-Roach (2007) found that Ca^{2+} addition did not have the same soil structure stabilization effect on expanding (like at CA) than on low activity clay minerals.

Lime addition increased soil pH and base saturation in the surface layer (Figure 3). We expected that these changes would increase Rp recovery rate by improving biological activity, especially of earthworm fauna (Ampoorter *et al.*, 2011; Potthoff *et al.*, 2008). Nevertheless, the impact of heavy traffic on Rp was still significant three years after compaction and soil liming. Lime addition did not yet result in a faster reduction of compaction degree as assessed by Rp. This result agreed with the study of Ampoorter *et al.* (2011) as they did not find an increase in Rp recovery rate two years after heavy traffic and soil liming. The absence of effect of lime addition on the rate of Rp restoration may not necessary imply that soil liming did not affect soil structure improvement. In particular in June 2011, the lower WC in the A- than in the C-treatment and in the TA- than in the T-treatment may originate from different soil water consumption and/or from different soil structure.

Conclusion

Even if we assessed the impact of a wood-loaded forwarder on Rp during only a short time period after disturbance (three years), we stated several methodological issues for long-term field monitoring of soil quality.

At one sampling time, we should bear in mind that both treatment may not have the same water status and therefore the contribution of water potential to Rp may not be the same. The analysis of the raw Rp data presented the advantage to state the actual rooting conditions as affected by heavy traffic. Yet the disadvantage of this approach was to confound the impacts of heavy traffic on soil water regime and on soil pore space. For example, in November 2008, at CA soil total porosity was affected by treatment but as the disturbed soils were water saturated or more, no impact on Rp could be stated, which did not mean that roots would not eventually experiment high Rp in the T-treatment when dry.

To compare the impact of treatment on Rp between years, Rp measurements should be standardized with regards to WC and/or water potential. Indeed even when we measured Rp during the same month (June), soil water status and the impact on Rp varied greatly from one year to the other. Besides calibrating the relationship between Rp and WC as affected by treatment can allow calculation of the water content corresponding to restricted root growth due to high Rp for each treatment and for each site.

No recovery in Rp could be detected even by accounting for WC variations, three (CA) to four (AZ) year after compaction. Lime addition did not increase the rate of Rp recovery, yet it did often decrease WC and consequently increase Rp probably because of a change in soil water regime (water consumption and/or change in soil structure). Soil strength assessment may be more sensitive to soil degradation to heavy traffic and therefore offer a greater potential as indicator of soil physical quality than BD (significant BD recovery in the surface layer of both sites whereas no significant trend in Rp improvement). Yet the dependence of Rp on BD, WC, texture and SOC among others may impede the definition of threshold values applicable to different soils, except if relationships among these soil characteristics were calibrated. More researches are needed to allow the use of Rp in soil quality monitoring procedure.

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IV

SOIL CO_2 CONCENTRATION AND EFFLUX AS AFFECTED BY HEAVY TRAFFIC IN FOREST IN NORTHEAST FRANCE

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Summary

An experimental site was set up in northeast France on a Luvisol (ruptic) soil to examine the duration of physical, chemical and biological disturbances in the soil following mechanized forest harvesting. Soil carbon dioxide (CO_2) efflux (SE) and concentration ([CO_2]) in the silt loam layer (0–50 cm) were measured in March 2008–March 2010 in the trafficked (T) and control (C) plots of this site. This study aimed to validate these two measurements as indicators for long-term soil monitoring following disturbance by heavy traffic in 2007. Throughout the sampling period, SE in the T-treatment was significantly reduced relative to that in the C-treatment. The response of [CO_2] to traffic depended on the season; it decreased during summer and increased during winter and spring. The combination of the two measurements indicated an increase in the frequency and duration of anoxic conditions owing to poor gas diffusion after heavy forest traffic. The relationships between soil climatic properties (temperature, water content, and water table level) on one hand and SE or [CO_2] on the other, demonstrated a strong control of SE by soil biological activity and a double control of [CO_2] by gas production/consumption and gas transfer. Our findings suggest that [CO_2] and SE are sensitive to soil degradation by forest harvesting and that the impact of soil compaction provides complementary information on the processes involved in regulating CO_2 production and efflux. However, their use as a simple indicator is questionable as the impact varied with time and was probably dependent on the soil type.

Introduction

Environmental sustainability is one of the aims of forest management. Further, soil protection is also an important goal. However, mechanization of forest operations is unavoidable as it provides benefits in lowering costs and reduction in manual labour and efficient production of timber. This in turn is necessary for meeting the current requirements in Europe for both industrial and energy purposes. However, the resulting heavy traffic in managed forests increases the risk of soil physical degradation, which in turn would affect soil productivity and all functions of the ecosystem (Gaertig *et al.*, 2002). In fact, the use of heavy machinery could potentially result in a reduction in, and re-arrangement of, soil total porosity and an increase in soil strength. These modifications would result in changes in water infiltration, water-holding capacity and content, aeration (gas exchange between the soil and the atmosphere) and temperature of soil (Ball *et al.*, 1999; Page-Dumroese *et al.*, 2006). The impacts of heavy traffic on soils are often inter-dependent, but only a few studies have evaluated the long-term effects of forest harvesting on soil productivity by taking into account the various functions affected by heavy traffic (Marshall, 2000; Page-Dumroese *et al.*, 2006).

Plant roots and microbial activities are much affected by soil compaction (Whalley *et al.*, 1995; Marshall, 2000; Bekele *et al.*, 2007). Shifts in the composition of both plant and microbial communities have been reported in an aspen forest (Mariani *et al.*, 2006) which favour species able to thrive under hypoxic or anoxic conditions (Schurr-Pütz *et al.*, 2006). In the steady state, soil carbon dioxide (CO_2) production is a result of both root respiration and soil organic matter decomposition (Epron, 2009). Therefore, changes in root and microbial activities possibly make the soil carbon dioxide (CO_2) efflux (SE) sensitive to heavy traffic. Vincent *et al.* (2006) found that in a temperate deciduous forest, the spatial variation of the optimal soil water content for SE was related to the spatial variation in the bulk density of hydromorphic soils. The impacts of soil compaction on the roots or microbe metabolism will also be felt by SE; similarly, changes in physical soil properties that alter gas diffusion will also modify the relationship between the SE and the soil CO_2 concentration ([CO_2]). Therefore, heavy traffic will probably modify the soil atmosphere composition and soil gas efflux, particularly [CO_2] and SE (Ball *et al.*, 1999; Bekele *et al.*, 2007). Conlin & van den Driessche (2000) reported an increase in [CO_2] with soil compaction by measurements made over three years in Central British Columbia; they found that the response of [CO_2] to compaction was different in all three years. However, they could not draw conclusions about the cause of these inter-annual variations because they did not measure soil characteristics such as water content. During a study of podzols in Belgium, Ampoorter *et al*. (2010) found that [CO_2] was an indicator of soil physical degradation caused by forest mechanization and was more sensitive to soil compaction than bulk density or resistance to penetration.

With this background, our objective in this study was to test [CO_2] and SE as accurate and reliable variables to quantify soil physical degradation and regeneration after heavy forest traffic. We tested the following hypotheses: (i) heavy traffic leads to soil compaction and affects soil temperature, water content, [CO_2] profile and SE; (ii) the relationship between [CO_2] and SE is influenced by soil disturbance; and (iii) the relationship between soil climatic conditions (soil temperature, soil air-filled porosity, water table) with both the [CO_2] concentration profile and SE influenced the response of [CO_2] and SE to soil degradation during measurements for two years.

Materials and methods

Study site

To study the short- to long-term effects of soil compaction on several soil properties, an experimental site was set up in the northeast of France, in the 'Hauts Bois' state-owned forest (48°29′19″ N, 6°41′43″ E) near Azerailles, Meurthe et Moselle. The site elevation is 300 m above sea level with a maximum level variation of 8 m through the site, thus limiting the spatial variations in soil characteristics. Data acquired from the nearest weather station in Nancy (approximately 50 km from the site) for 30 years (1971–2000) indicate that the mean annual precipitation is 765 mm with minimum temperatures averaging 5.6° C and maximum temperatures averaging 14.4° C (http://france.meteofrance.com/france/climat_france. Accessed: 07/01/2011).

Experimental design

The previous high forest dominated by *Fagus sylvatica* L. and *Quercus petraea* L. as the secondary species was clear-cut. Timber was extracted in March 2007 by using a cable yarding system in order to avoid soil damage. In May 2007, a 8-wheel drive forwarder (a log-carrying articulated tractor, VALMET 840) was driven for an equivalent of two passes on land strips large enough to allow long-term monitoring (30 m × 50 m each), in three blocks to account for spatial variability. In each block, other 30 m × 50 m strips, adjacent to the trafficked plots (T), were left undisturbed by the forwarder and were considered as control plots (C). The tyres of the unloaded forwarder were 60-cm wide, diameter of 133 cm (600/55 × 26.5) and were inflated to a pressure of 350 kPa. The total weight of the eight-wheeled forwarder was 25 tonnes (the maximum weight allowed for this kind of equipment), applying a mean vertical contact stress of approximately 160 kPa. The mean volumetric water content of the soil surface was 0.33 $m^3 m^{-3}$; at the time the forwarder drove on the soil, this content ranged from 0.27 to 0.40 $m^3 m^{-3}$. Heavy traffic resulted in ruts averaging 5 cm in depth. Deeper ruts were formed at some points of the site, because of the spatial heterogeneity of the soil water content and strength; these very deep ruts were avoided during sampling.

Soil description

The soil is classified as Luvisol (ruptic) (IUSS Working Group WRB, 2006) developed from an approximately 50-cm thick fluviatile silt loam layer, overlaying a heavy clayey material (weathered from Keuper marl). This strong textural discontinuity causes limited temporary water logging; the silt loam layer did not show any hydromorphic figures, except at depth. The soil of the Azerailles site is consequently considered as being very sensitive to compaction. Before setting up the experimental site, selected soil characteristics were measured in samples taken from 15 pits distributed over the entire area (see Table 1).

Table 1 Selected mean characteristics of soil of the 'Hauts Bois' forest site (15 pits, sampled in 2006).

Depth /m	pH H_2O	Gravel (>2mm) /%	Clay /g kg^{-1}	Silt /g kg^{-1}	C org[a] /g kg^{-1}	C/N	Exch. Ca[b] /cmol$_c$ kg^{-1}	CEC[b] /cmol$_c$ kg^{-1}	BS[c] /%	CBD Fe[d] /%	P Duch[e] /g kg^{-1}
0–0.1	4.81	7.1	221.5	556.3	26.7	15.1	2.17	5.7	63	1.8	0.21
0.2–0.3	4.62	3.5	235	566.4	10.8	14.6	0.46	4.5	24	1.5	0.18
0.4–0.6	4.72	4.4	454.8	428.7	3.6	9.8	3.38	12	43	2.7	0.16
0.8–1	5.16	1.9	589.7	325.7	1.5	5.7	5.95	14.2	70	2.4	0.21

[a] soil organic carbon content (CHN apparatus, zero carbonates)
[b] Exchangeable calcium and cation exchange capacity, Ciesielski & Sterckeman (1997)
[c] Base saturation, sum of base cations divided by the cation exchange capacity
[d] CBD iron, Mehra & Jackson (1960)
[e] plant available phosphorus, Duchaufour & Bonneau (1959).

Ground vegetation cover and diversity

The ground vegetation cover was estimated visually with Braun-Blanquet cover-abundance values, along 8–26-m permanent transects perpendicular to the direction of traffic (Brêthes A., unpublished data). The Braun-Blanquet values were then transformed into the corresponding cover percentage values (Godefroid & Koedam, 2004). The composition of the vegetation was immediately (within two months) modified following the forwarder traffic; the T-treatment consisted mostly of rushes (*Juncus* sp.), and the C-treatment consisted mostly of bramble (*Rubus fructicosus* L.). In April 2008, the normalized cover percentages of *Anemone nemorosa* L., *Juncus* sp., *Glyceria striata* Lam., and *Rubus fructicosus* were 40%, 1%, 2% and 57% for the C-treatment and 6%, 80%, 10%, and 4% for the T-treatment, respectively.

The entire site area was planted in autumn 2007 with 1600 plants ha^{-1} of sessile oak (*Quercus petraea* L.).

Analysis of soil physical and chemical properties

In June 2007, after the forwarder had driven on the site, nine pits were dug, six within the T areas (two per block) and three within the C areas (one per block); the three pits dug in each block were separated by a maximum distance of 5 m. Undisturbed soil samples from four depths (0–10, 10–20, 20–30 and 30–40 cm) were collected in 500 cm^3 steel cylinders to measure soil bulk density (four replicates per depth × pit). The soil bulk density (ρ_B) was then calculated as the mass of the oven-dried (105° C over 72 hour) soil sample divided by its volume and was corrected for the rock fragment content (Page-Dumroese *et al.*, 1999).

In April 2009, randomly distributed soil samples from two soil layers (0–5 and 5–10 cm) were collected (120 samples in total: ten samples per treatment × depth × block), dry-sieved (2 mm), and ground (250 μm) for determining the total carbon (C) and nitrogen (N) contents for conventional analysis. The analysis was performed using a CHN apparatus (2500 NCS, ThermoQuest, Finnigan CE Instruments, Waltham, MA, USA).

Soil climatic conditions

Rainfall, soil temperature and soil water content were recorded continuously (every two hours) on two data loggers (DL2e Data Logger, Delta-T devices Ltd., Cambridge, UK, for rainfall and soil temperature; Trase 'B.E.' by Soil Moisture, Sols Mesures, Elancourt, France, for soil water content) located approximately at the centre of the site. Time domain reflectometry (TDR) probes and temperature sensors (five replicates per depth × treatment) were inserted at two depths (15 and 30 cm in the undisturbed soil; 10 and 25 cm in the compacted plot) into the soil layer that was more susceptible to soil degradation by forest harvesting (the silt loam soil layer). The depths from the soil surface were varied between treatments to facilitate investigations of the same soil layers by different sensors. The intervals between the five replicates varied between 2 and 20 cm. The soil dielectric constant values obtained by the TDR probes were converted to volumetric water content using calibration curves according to Heathman *et al.* (2003). The curves were derived from laboratory measurements of both the gravimetric water content and the TDR data on intact soil cores subjected to evaporation. In winter, soil temperatures dropped considerably to below 0° C, causing the soil water content measurements to be non-coherent (Magnusson, 1992). The corresponding measurements were deleted to avoid misleading interpretations. Twelve sensors (Diver by Van Essen Instruments, SDEC, Reignac sur Indre, France) that recorded the soil water table level (every four hours from June 2009) were distributed over the entire site area in both C and T treatments (six sensors per treatment).

The mean volumetric water content per day and mean bulk density values were used to calculate the daily air-filled porosity, defined as the ratio of volume occupied by air to the volume of soil:

$$\varepsilon_A = 1 - \frac{\rho_B}{\rho_S} - \theta_V \quad , \qquad (1)$$

where, ε_A is the soil air-filled porosity in $m^3\ m^{-3}$, ρ_B is the soil bulk density in g cm^{-3}, ρ_S is the density of the solid in g cm^{-3}, and θ_V is the soil volumetric water content in $m^3\ m^{-3}$. The density of the solid was measured on dried and ground (200 µm) soil samples (one per block × depth) using a pycnometer (multivolume pycnometer 1305, Micromeritics, Norcross, USA); ρ_S averaged to 2.58 (standard deviation (SD): 1×10^{-3}), 2.62 (SD: 1×10^{-3}), and 2.66 (SD: 2×10^{-3}) for the 0–10, 10–20 and 30–40-cm soil layers, respectively.

Soil air-filled porosities less than zero were observed, probably resulting from the soil's heterogeneity because soil water content and bulk density were not measured at the same place (sometimes more than a few metres apart). Therefore, ε_A was considered as a comparative indicator of aeration conditions between the T- and the C-treatments: the greater the air-filled porosity, the greater is the gas diffusion (Buckingham, 1904).

Analysis of [CO_2] and SE

From March 2008 to March 2010, SE was measured at least once a month on the same day that the soil air was sampled. This measurement was performed using a dynamic closed-path system equipped with a 10-cm-diameter respiration chamber (SRC-1) coupled to an infrared gas analyser (CIRAS 1 by PP Systems, Hansatech Instruments, Ltd., Norflolk, UK). One week before commencement of the SE measurements in March 2008, four PVC collars in the C-treatment and eight in the T-treatment (half on the top of the ruts and half at the bottom) were installed in each block and near to installed gas collectors (at maximum distance of 5 m).

In February 2008, 36 gas collectors were installed at the site; each collector was made of a stainless-steel tube (1.59 mm internal diameter) that connects a sub-surface soil air equilibration chamber to a sampling port at the soil surface. The chamber was made of a 3.4-ml stainless-steel cylinder perforated with 24 holes (diameter, 2 mm) to allow soil air to diffuse into the sampler from the surrounding air-filled pore space (prototype from Renault P., EMMAH, INRA, Avignon, France). The gas collectors were inserted vertically into the soil so that the chambers reached the depths sampled at 10, 25, 35 and 50 cm in the C-treatment and 5–10 cm shallower at each depth in the T-treatment. In both treatments, the difference between the depths of the gas collectors from an equivalent soil layer depended on the actual depth of the ruts where the gas collectors were installed. Three replicates per depth and treatment (one in each block) were permanently established and sampled monthly, except for the first depth in the compacted soil (five replicates). The gas collectors in two out of three blocks were at a minimum distance of 50 m, whereas those of the remaining block (two for the C-treatment and three for the T-treatment) were at a maximum distance of 20 m from the location of the soil physical measurements (temperature and moisture). Five more replicates per treatment were inserted near the soil climate sensors (at a maximum distance of 15 m) and at a depth of 35 cm (25 cm) in the C-treatment (T-treatment), to better describe the relationship between soil climate and [CO_2]. The interval between the collectors was approximately 0.5 m. From March 2008 to March 2010, soil air samples were collected once a month, using a double-sided needle in a 15 ml vacutainer in the morning. Soil air composition (N_2, O_2, CO_2, CH_4, and N_2O) was then determined in the laboratory (at 20° C) with a micro-GC 4900 gas chromatograph (Varian, Les Ullis, France), usually on the day after sampling. The gas sample was simultaneously injected into two columns (PoraPLOT Q and Molecular Sieve 5A) with a helium carrier and was detected using a thermal conductivity detector (TCD).

To link the soil SE and soil air [CO_2], the gradient in soil air [CO_2] was used, as gas diffusion is related to the gradient of the gas concentrations rather than to the gas concentration itself:

$$\Delta[CO_2] = \frac{[CO_2]_{layer2} - [CO_2]_{layer1}}{depth_{layer2} - depth_{layer1}} \times 100 \times \frac{P_{atm}}{R \times T}, \qquad (2)$$

where [CO_2] is the soil air CO_2 concentration (in %v); $\Delta[CO_2]$, the gradient in [CO_2] (in µmol m^{-2}); *depth*, the depth of the two soil layers expressed in cm; P_{atm}, the atmospheric pressure (in Pa); R, the ideal gas constant (=8.31441 J mol^{-1} K^{-1}); and T, the temperature (taken as 293.15 K°, the temperature at which the gas sample was analysed).

Data analysis

All statistical treatments of the data were conducted with R software version 2.11.1 (R Development Core Team, 2010). One of our objectives was to estimate the effect of the treatment (two levels: 'C' for control and 'T' for trafficked) and depth on different soil parameters while considering the effects of block and time that may in turn influence the effects of treatment and depth. We also wanted to determine a statistical model that could handle unbalanced data sets with missing values resulting from water collecting in gas collectors) and be adapted to both longitudinal data (SE, [CO_2], ε_A and temperature) and non-longitudinal data (C, N, and ρ_B). Therefore, mixed-effect models (package 'lme4') were used to estimate the effects of soil disturbance and depth (fixed predictive part) on soil ρ_B, N and C contents, SE and [CO_2], temperature and ε_A; the variables 'block' and 'time' were considered to have random effects. To evaluate the random effect of the sampling time, the data were grouped into four seasons: spring (March–May, increasing temperature), summer (June–August, high temperature), fall (September–November, decreasing temperature), and winter (December–February, low temperature). This grouping was made for each of the years after compaction (March 2008 to March 2009 being the second year after compaction, and March 2009 to March 2010 the third year after compaction). Any interaction terms between the random effects and fixed effects were also expressed as random effects. The best models were selected after analysing the residuals, comparing the Aikake information criterion (AIC) and the Bayesian information criterion (BIC), and using the likelihood ratio Chi-2 test for nested models.

Bivariate non-linear models (package 'nls') were fitted on SE data to test for pedoclimatic influence in each treatment. We used the function described in Vincent *et al.* (2006) because our data also showed an exponential relationship between SE and soil temperature, with residuals showing a relationship with the soil water content:

$$SE = a\exp\left[-\left(\frac{\ln(\theta_V / c)}{d}\right)^2\right]\exp\left[b\left(Temp. - 10\right)\right], \quad (3)$$

where SE is the soil CO_2 efflux in µmol m^{-2} s^{-1}; θ_V is the soil volumetric water content in m^3 m^{-3}; $Temp$ is the soil temperature in °C; and a, b, c, and d are the regression coefficients. The relationships between the soil air [CO_2] and the soil climatic variables did not show non-linearity; we thus chose to fit linear models to the data. After checking for the absence of bias in the distribution of the residuals, the best model was selected by comparing the coefficients of determination (R^2) and the residual standard errors.

Results

Forest heavy traffic led to soil compaction and affected soil water regime and soil CO_2 movement

The bulk density was significantly ($P \leq 0.001$) affected by the treatment and soil depth. The effect of the treatment decreased slightly with increasing depth; however, the interaction between these two effects was not significant ($P \geq 0.05$) (Tables 2 and 3).

Table 2 *P* values associated with the likelihood ratio Chi-2 test for nested linear mixed models

soil characteristics[a]	Fixed effects[b]					Random effects			
	Treatment	Depth	Treatm-ent : Depth	Block	Block : Fixed effect	Season in year	Season in year : Fixed effect	Year	Year : Fixed effect
ρ_B /g cm^{-3}	<0.001	<0.001	n.s.	n.s.	n.s.	n.d.	n.d.	n.d.	n.d.
Corg or N content /g kg^{-1}	n.s.	<0.001	n.s.	n.s.	n.s.	n.d.	n.d.	n.d.	n.d.
ε_A/m^3 m^{-3}	n.d.	n.d.	< 0.001	n.d.	n.d.	<0.001	<0.001	n.s.	n.s.
Temp / C	n.s.	n.s.	n.s.	n.d.	n.d.	<0.001	n.s.	n.s.	n.s.
[CO_2] /%	n.d.	n.d.	<0.001	n.s.	n.d.	<0.001	<0.001	n.s.	n.s.
SE /µmol m^{-2} s^{-1}	<0.001	n.d.	n.d.	n.s.	n.s.	<0.001.	<0.001	n.s	n.s.

[a] ρ_B, soil bulk density; Corg, organic carbon; N, nitrogen; ε_A, soil air-filled porosity; *Temp*, soil temperature; [CO_2], soil air CO_2 concentration; SE, soil CO_2 efflux.
[b] n.s., non-significant ($P > 0.05$); n.d., non-determined; ':', interaction term between the two factors.

We observed no significant ($P \geq 0.05$) difference in soil surface total C and N contents between the two treatments (Table 2). Values for both variables had a large spatial variability that could mask the effect of compaction (Table 3).

Table 3 Mean and standard error (in parentheses) of several soil properties as a function of depth and control (C) and trafficked (T) treatments on the ‘Hauts Bois’ forest site

<table>
<tr><th></th><th>C</th><th>T</th><th>C</th><th>T</th><th>C</th><th>T</th><th>C</th><th>T</th><th>C</th><th>T</th><th>C</th><th>T</th></tr>
<tr><th>depth</th><th>ρ_B [a]</th><th>ρ_B</th><th>Corg</th><th>Corg</th><th>N</th><th>N</th><th>ε_A</th><th>ε_A</th><th>Temp</th><th>Temp</th><th>Nb w.t.</th><th>Nb w.t.</th></tr>
<tr><th>/cm</th><th>/g cm^{-3}</th><th>/g cm^{-3}</th><th>/%</th><th>/%</th><th>/%</th><th>/%</th><th>/m^3 m^{-3}</th><th>/m^3 m^{-3}</th><th>/ C</th><th>/ C</th><th></th><th></th></tr>
<tr><td>0–5</td><td rowspan="2">0.98 (0.06)</td><td rowspan="2">1.24 (0.07)</td><td>3.16 (0.18)</td><td>3.53 (0.36)</td><td>0.22 (0.01)</td><td>0.24 (0.02)</td><td rowspan="3">0.23 (0.002)</td><td rowspan="3">0.24 (0.003)</td><td rowspan="3">10.47 (0.23)</td><td rowspan="3">10.43 (0.22)</td><td rowspan="2">0</td><td rowspan="2">5</td></tr>
<tr><td>5–10</td><td>1.99 (0.10)</td><td>1.80 (0.09)</td><td>0.15 (0.01)</td><td>0.14 (0.01)</td></tr>
<tr><td>10–20</td><td>1.23 (0.03)</td><td>1.40 (0.04)</td><td></td><td></td><td></td><td></td><td>0</td><td>39</td></tr>
<tr><td>20–30</td><td>1.30 (0.04)</td><td>1.41 (0.05)</td><td></td><td></td><td></td><td></td><td rowspan="2">0.22 (0.002)</td><td rowspan="2">0.15 (0.002)</td><td rowspan="2">10.54 (0.21)</td><td rowspan="2">10.42 (0.22)</td><td>6</td><td>85</td></tr>
<tr><td>30–40</td><td>1.34 (0.04)</td><td>1,42 (0.06)</td><td></td><td></td><td></td><td></td><td>32</td><td>103</td></tr>
</table>

[a] ρ_B, soil bulk density (two pits per block for T and one pit per block for C, sampled in 2007); Corg, organic carbon content and N, nitrogen content (10 samples per block × depth × treatment); ε_A, soil air-filled porosity and *Temp*, soil temperature (01-03-2008 to 31-03-2010 period, one measurement every two hours, five replicates per depth × treatment); Nb w.t., number of days of presence of the water table at a given depth (06-01-2009 to 31-03-2010 period, *i.e.* 450 days, two sensors per block × treatment).

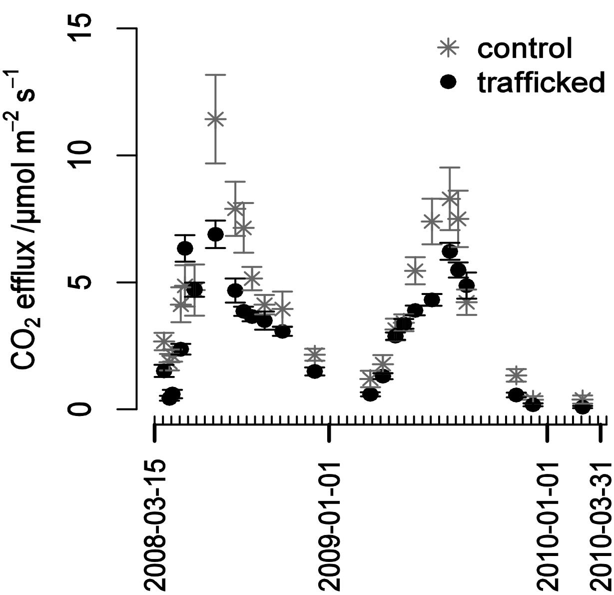
The soil air-filled porosity depended on the treatment applied, but this effect varied significantly ($P \leq 0.001$) with soil depth and with the season considered, whereas the soil temperature only varied significantly ($P \leq 0.001$) with the season (Tables 2–4). The two treatments differed in how long and how deep the water table was maintained throughout the third year after soil disturbance (Table 3). In the C plots, the water table did not reach the top 20 cm of soil, whereas in the T plots, the water table appeared more frequently and at shallower depths. Soil CO_2 efflux was greatest during the vegetation period and least in the dormant period (Figure 1). Each year, it increased from early April to late June–early July and then decreased continuously until the following April.

Table 4 Mean and standard error (in parentheses) of soil properties as a function of season and control (C) and trafficked (T) treatments on the 'Hauts Bois' forest site

	C	T	C	T
	ε_A[a]	ε_A	*Temp*	*Temp*
	/m^3 m^{-3}	/m^3 m^{-3}	/ C	/ C
spring t2[b]	0.21 (0.003)	0.15 (0.005)	10.27 (0.32)	11.24 (0.39)
summer t2	0.28 (0.002)	0.26 (0.005)	17.75 (0.07)	18.35 (0.09)
fall t2	0.22 (0.002)	0.19 (0.004)	9.56 (0.24)	9.19 (0.25)
winter t2	0.20 (0.003)	0.14 (0.004)	2.40 (0.13)	1.79 (0.14)
spring t3	0.23 (0.002)	0.20 (0.004)	10.80 (0.27)	9.96 (0.26)
summer t3	0.25 (0;003)	0.23 (0.004)	17.10 (0.09)	16.40 (0.08)
fall t3	0.27 (0.004)	0.23 (0.006)	10.53 (0.22)	10.61 (0.20)
winter t3	0.20 (0.002)	0.14 (0.003)	2.61 (0.09)	2.71 (0.07)

[a] ε_A, soil air-filled porosity; *Temp*, soil temperature.
[b] t2, second year after forwarder traffic (March 2008 to March 2009); t3, third year after forwarder traffic (March 2009 to March 2010).

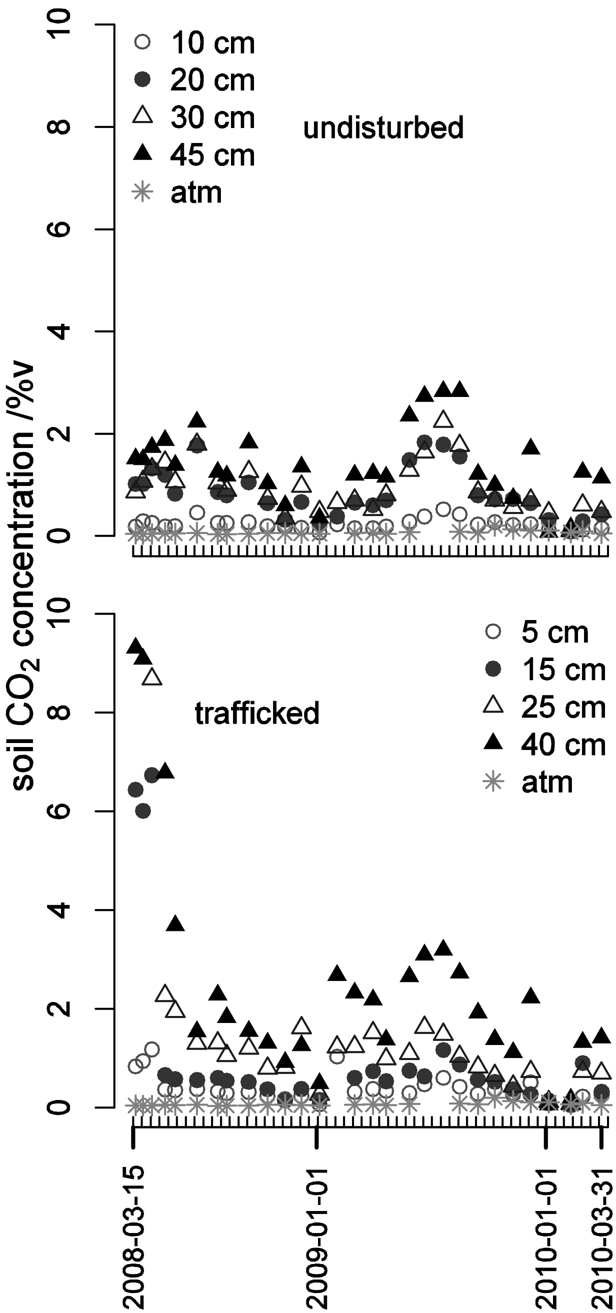
Figure 1 Impact of forwarder traffic on soil CO_2 efflux of the 'Hauts Bois' forest site from March 2008 to March 2010; each symbol represents the mean of 12 (C-treatment)/ 24 (T-treatment) measurements (standard error are plotted as error bars); tick marks indicate 14-day intervals from the start of the experiment (2008-03-15)



The difference between SEs of the two locations in the T-treatment (bottom and top of the rut) was not significant ($P \geq 0.05$), even though SE from the bottom was always slightly less than that from the top of the ruts (data not shown). The data from both locations were therefore pooled to characterize the effect of compaction on SE. Over the entire study period (March 2008–March 2010), SE was not significantly dependent ($P \geq 0.05$) on the position in the experimental design (block effect), but did show a dependence on the treatment applied or on the season (Table 2). Although the effect of the forwarder traffic depended on the season considered, traffic always resulted in a reduction in soil SE, and the amplitude of this reduction varied with time.

Over the entire study period of March 2008–March 2010, soil air [CO_2] was not significantly dependent on the block ($P \geq 0.05$), but it did depend significantly ($P \leq 0.001$) on the season and on interactions between the effects of the soil compaction and depth (Table 2). In turn, the effects of soil compaction and depth on [CO_2] depended significantly ($P \leq 0.001$) on the season (Table 2). Both the amplitude and the direction (decrease or increase) of impact of the forwarder traffic on [CO_2] varied with time (Figure 2). Generally during winter and spring, [CO_2] was greater in the T-treatment than in the C-treatment but this trend was reversed during summer. During spring 2008 (second year after compaction), the difference in [CO_2] between the T- and the C-treatments was noticeably larger than in spring 2009.

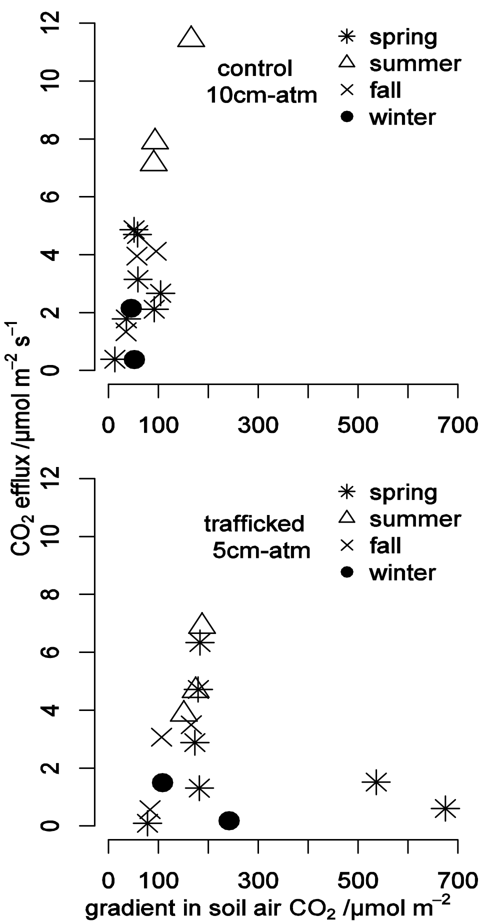
Figure 2 Impact of forwarder traffic on soil CO_2 concentration of the 'Hauts Bois' forest site from March 2008 to March 2010; each symbol represents the mean of three (or less because of water-filled gas collector) measurements; tick marks indicate 14-day intervals from the start of the experiment (2008-03-15)



Change in soil CO_2 diffusion

Soil CO_2 efflux correlated well with the gradient in [CO_2] in the C plots (especially for the upper layers), but the correlation was less obvious and more complex in the T plots (Figure 3). In the upper soil layers of the T plots, two different populations of CO_2 efflux could be distinguished; the first with a constant coefficient of diffusion (linear increase in SE with the gradient in [CO_2]) and the second in which the increase in the gradient in [CO_2] was associated with a decrease in the calculated gas diffusion coefficient (mostly in winter and spring).

Figure 3 Soil CO_2 efflux of the 'Hauts Bois' forest site from March 2008 to March 2010 as a function of $\Delta[CO_2]$: see Equation (2)



Influence of soil climatic conditions on [CO_2]) and SE

Over the entire sampling period (March 2008–March 2010), and for both treatments, SE was significantly ($P \leq 0.001$) dependent on the soil temperature and water content in the upper 20 cm (Table 5). However, it was better correlated with soil temperature than with the soil water content. There was no significant correlation between SE and the water table level in the control plots, but there was a significant ($P \leq 0.001$) correlation with the water table level in the T plots.

Table 5 Selected linear regressions fitted to explain [CO_2] and selected non-linear models fitted to explain SE using the function described in Vincent *et al.* (2006) (see Equation 3). All the model coefficients are given with their standard errors in parentheses: coefficients *a*, *b*, *c* and *d* refer to those of Equation (3) and coefficients *e*, *f* and *g* refer to those of the multivariate regression of [CO_2] on soil air-filled porosity(ε_A, in $m^3\ m^{-3}$) and temperature (*Temp*)

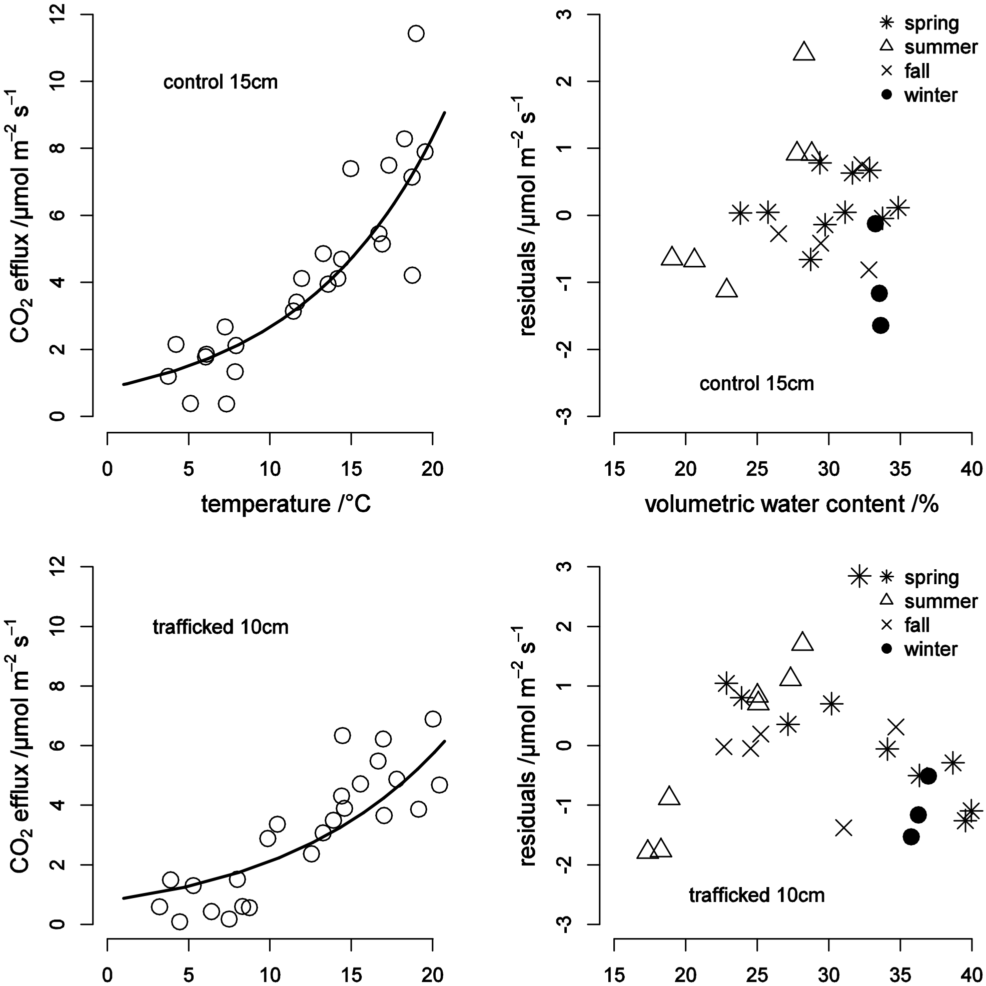
Treat-ment[a]	Dependent variable [b] (soil layer considered)	Independent variable: soil layer considered	Univariate regressions[c]			Non linear bivariate models according to Vincent *et al.* (2006), cf. equation [3]						
			θ_V /m^3m^{-3}	*Temp* /°C	w.t. /cm	*a*	*b*	*c*	*d*	RSE	dof	
C	SE	15 cm	*** RSE: 2.3; R^2: 0.37	*** RSE: 1.4; R^2: 0.76	n.s. RSE: 2.8; R^2: 0.14	2.77 (0.35)	0.12 (0.02)			1.35	23	
T	SE	10 cm	*** RSE: 1.6; R^2: 0.44	*** RSE: 1.1; R^2: 0.75	*** RSE: 1.2; R^2: 0.73	2.48 (0.31)	0.12 (0.02)	0.27 (0.02)	0.55 (0.09)	0.88	21	
						multivariate regressions [CO_2] = $e + f\ \ \varepsilon_A +\ g\ \ Temp$						
						e	*f*	*g*	R^2	p-value	RSE	dof
C	[CO_2] (10 and 20 cm)	15 cm	n.s. RSE: 0.29; R^2: 0.02	** RSE: 0.22; R^2: 0.47	n.s. RSE: 0.32; R^2: 0.14	1.07 (0.22)	-4.68 (1.12)	0.06 (0.009)	0.67	<0.0001	0.18	24
T	[CO_2] (5 and 15 cm)	10 cm	** RSE: 0.81; R^2: 0.22	n.s. RSE, 0.90; R^2, 0.007	n.s. RSE: 0.25; R^2: 0.03	2.68 (0.54)	-13.95 (3.57)	0.11 (0.04)	0.40	0.002	0.73	24
C	[CO_2] (30 cm)	30 cm	n.s. RSE: 0.50; R^2: 0.009	*** RSE: 0.37; R^2: 0.46	n.s. RSE: 0.52; R^2, 0.12	1.99 (0.33)	-10.81 (1.97)	0.12 (0.01)	0.75	<0.0001	0.26	24
T	[CO_2] (25 cm)	25 cm	n.s. RSE: 1.58; R^2: 0.08	n.s. RSE: 1.6; R^2: 0.006	n.s. RSE: 0.47; R^2: 0.06	3.37 (1.12)	-24.67 (10.43)	0.16 (0.16)	0.20	0.08	1.49	22

[a]C, 'control'; T, 'trafficked'.
[b]SE, soil CO_2 efflux in µmol $m^{-2}\ s^{-1}$; [CO_2], soil air CO_2 concentration in %.
[c] ***, $P \leq 0.001$; **, $P \leq 0.01$; n.s., non-significant ($P > 0.05$); θ_V, soil volumetric water content; w.t., depth of water table; RSE, residual standard error; R^2, coefficient of determination; dof, degrees of freedom (the dof values of the univariate regressions were equal to 10 for w.t. *vs.* SE and 15 for w.t. *vs.* [CO_2]; to 23 for Temp. or θ_V *vs.* SE and 25 for Temp. or θ_V *vs.* [CO_2]).

In the T-treatment, SE was best explained by a non-linear model that combined the effects of soil temperature and water content (Table 5). Soil CO_2 efflux increased exponentially with temperature, and the residuals (observations minus predictions) showed a bell-shaped relationship with the water content (Figure 4), thus validating the effectiveness of the bivariate model proposed by Vincent *et al.* (2006). However, in the C-treatment, soil respiration was best described by an exponential function of soil temperature; the residuals did not show any relationship with the soil water content (Table 5 and Figure 4).

In the C-treatment, [CO_2] was strictly dependent on soil temperature. In the T-treatment, [CO_2] was significantly ($P \leq 0.01$) correlated with the soil volumetric water content in the upper soil layer, but it did not show any significant ($P \geq 0.05$) dependence on soil temperature or water table level in either of the soil layers considered (Table 5). In the C-treatment, 'air-filled porosity' did not significantly ($P \geq 0.05$) influence [CO_2] on its own, but combined with the soil temperature, it significantly ($P \leq 0.001$) increased the percentage of variation in the dependent variable explained. In the T-treatment, the same behaviour was observed for 'soil temperature'. The improvement in the R^2 value by using both independent variables ('air-filled porosity' and 'temperature') was particularly significant for undisturbed soil (Table 5). The regression coefficients associated with the effects of soil temperature and air-filled porosity on [CO_2] differed with treatments and depths.

Figure 4 Non-linear relationships between mean soil CO_2 efflux and mean soil temperature of the 'Hauts Bois' forest site from March 2008 to March 2010 (left-hand side), and absence or presence of relationship between the residuals of the predicted soil respiration and soil water content (right-hand side)



Impact of harvesting on [CO_2] and SE

The forwarder traffic significantly increased soil bulk density, by 6 (30–40 cm) to 27% (0–10 cm), confirming the effectiveness of the large-scale compaction treatment applied on the site. This range agreed with the results of measurements at long-term soil productivity sites reported by Page-Dumroese *et al.* (2006) on a large-scale field plot. The significant soil compaction and the probable change in soil structure led to the modifications of soil water regimes but not of soil temperature regimes. These results support the hypothesis that forest heavy traffic has an impact on soil CO_2 production/consumption and transfer through effects on soil moisture.

The range of SEs found in this study was comparable to values measured in other poorly drained forests soils in northeast France (Vincent *et al.*, 2006). The large SE found in the C-treatment could be explained by the fact that CO_2 production in the soil is greater under aerobic than under anaerobic conditions (Ball *et al.*, 1999; Vincent *et al.*, 2006). This assumption was partly confirmed by the more frequent and shallower water table recorded in the disturbed plots and by the shift in under-story plant composition towards more water logging and compaction-tolerant plant species (*Juncus* sp., *Glyceria striata* Lam.). Another explanation could be that during the summer dry periods, the compacted soil was drier than the undisturbed one. Drier conditions could inhibit CO_2 production in soils because of the limitation of soluble organic C-substrate diffusion as reported by Davidson *et al.* (2000) or Epron *et al.* (2004). The decrease in SE caused by heavy traffic did not result from differences in soil C and N contents between treatments. Nevertheless, a change in C quality (data not published) could be another factor explaining the reduction in SE as suggested by Epron *et al.* (2004). Two years after treatment, soil organic C quantity was not affected by treatment despite the shift in vegetation and to the modified soil biological activity (as indicated by SE). A longer treatment period may be required to detect changes in soil organic C and N contents, especially when the large heterogeneity in these two soil characteristics is considered. For example, Mariani *et al.* (2006) found a significant increase in mineral soil C and N contents three to seven years after heavy compaction, which was correlated with microbial respiration.

The range of [CO_2] (0.004–10.225% v) found in the experimental site was consistent with the range reported by Bekele *et al.* (2007). However, we find both smaller and larger values of between 0.035 and 7% v. The smallest values of [CO_2] were recorded in the deep soil layers and were recorded in winter, when there was little biological activity (small rates of soil respiration) and the water table often shallow; at this time, above ground production was minimal and groundwater was a sink for CO_2 (Magnusson, 1992). Contrastingly, the largest values of [CO_2] were recorded in the deep soil layers during spring, when the biological activity increased and gas diffusion conditions varied considerably (large variations in soil water content), causing occasional accumulation of soil CO_2. The same trends in variations of [CO_2] according to season and depth have been reported by numerous studies of forested sites (Fernandez *et al.*, 1993; Certini *et al.*, 2003, Bekele *et al.*, 2007). The trend of increase in [CO_2] with depth is consistent with probable deep rooting activity and microbial production of CO_2. Even if the deep biological activity produces less CO_2 and at a slower rate than in the soil surface layers, the very slow rate of gas exchange between the sub-soil and the atmosphere causes accumulation of CO_2 in the soil air of the deep layers (Certini *et al.*, 2003). The forwarder traffic had a complex impact on [CO_2], as it depended on the depth, season and year considered. Conlin & van den Driessche (2000) also found that the impact of soil compaction on [CO_2] varied with depth and year. However, as only soil atmosphere composition during the growing season was measured, no seasonal trend could be identified. In our study, we found that the soil air CO_2 concentration both increased and decreased because of forwarder traffic, depending on the season at which gas sampling occurred. Therefore, caution should be taken when defining sampling time for quantifying the effect of soil compaction on the soil atmosphere composition.

Impact of forwarder traffic on the relationship between [CO_2] and SE

The significant decrease in SE, observed throughout the year and caused by compaction, probably supports the fact that [CO_2] was greater in the C-treatment than in the T-treatment plots during the dry periods (summer). During the dry periods, gas diffusion was not very restricted in the T-treatment, because of the drier soil and perhaps also because of soil cracking (a decrease in the hydro-structural stability following compaction as assessed by Schäffer *et al*., 2008). The reduced biological activity (confirmed by the small SE) resulted in less CO_2 in the T-treatment than in the C-treatment, explaining the difference in concentrations. During the wet periods (mostly in winter and spring), gas diffusion was very limiting in the T-treatment because of a reduced soil air-filled porosity than in the C-treatment. Thus, even if the biological activity produced less CO_2, it still accumulated in the T-treatment. Gaertig *et al.* (2002) also found large [CO_2] in compacted soils with reduced respiration rates, indicating poor gas transfer conditions.

Relationships between soil climatic conditions and [CO_2]/SE

Pedoclimatic variables that had the greatest correlation with SE and [CO_2] differed between treatments and depended on soil depth. These correlations (or their absence) can be partly explained by the fact that the water table and the air-filled porosity of the upper soil layers varied less in the C-treatment than in the T-treatment. For example, the more frequent and shallower water table in the T-treatment made the correlation between SE and the water table level obvious. In contrast, in the C-treatment, the water table was neither frequent nor shallow enough to induce variations in SE. With [CO_2], the large variations of air-filled porosity in the compacted soils may have partly masked the effect of soil temperature. Low temperatures were associated with not only less CO_2 production but also a large water content/small air-filled porosity (poor diffusion) resulting in more CO_2 accumulation in the T-treatment plots.

During the two years of monitoring in the T-treatment, SE was predicted well by a bivariate non-linear model, with an optimal soil water content of 27.5 %. In the non-disturbed soils, no optimal value of soil water content for soil respiration seemed to stand out. Vincent *et al.* (2006) observed a dependence of the optimal soil water content for soil respiration on soil bulk density and N content. In our case study, we only observed differences in bulk density, probably explaining the absence of influence of soil water content on soil respiration in the C-treatment. Moreover, in the T-treatment, during winter when the soil temperature and therefore the biological activity were reduced, greater water content than that in the C-treatment inhibited further CO_2 production. During summer, when the soil temperature was increased, reduced water content also led to a decrease in SE. At the reduced and increased water contents, the disturbed soils deviated from the optimal soil water content for soil respiration but the undisturbed soils did not: this perhaps partly explains why no optimal water content for soil respiration could be observed in the C-treatment. The fact that the T-treatment more often offered extremes in water conditions for soil biological activity may not have resulted from the increased bulk density. The normalized soil respiration at both 10° C and optimal soil water content (a coefficient of Equation 3) had good potential as an indicator of soil disturbance after heavy traffic. Indeed, this normalized soil respiration depended only on treatment, integrating the influence of soil temperature and water content on SE.

With [CO_2], the double control exerted by the soil biological activity (CO_2 production/consumption) and by diffusion was validated by the statistical analysis of the data set. Soil temperature and soil air-filled porosity explained most of the variations in [CO_2]. Bekele *et al.* (2007) also found that using soil temperature and volumetric water content as independent variables increased the determination coefficient (R^2) and that this improvement was site-dependent. They further reported that the regression coefficients associated with soil temperature and water content differed with treatment and depth. When we used the same factors in both treatments, we obtained a more significant multivariate linear regression with better R^2 and small residual standard errors for the C-treatment than for the T-treatment. Additional factors may be needed in the T-treatment to increase the percentage of variation explained. For example, heavy traffic may have altered the continuity and stability of soil pores, both factors being important for gas transfer (Hakansson & Lipiec, 2000).

The correlations of SE with soil climatic conditions were stronger than those of [CO_2]. This could be attributed to the fact that SE results from respiration of root and soil living organisms (Gaertig *et al.*, 2002) whereas [CO_2] results from a balance between the soil biological activity and gas transfer. The interactions between both these processes may lead to a more complex interpretation of the contribution of the different variables to [CO_2]. It would probably be easier to detect a change in the impact of heavy forestry traffic on SE than on [CO_2]. Thus the larger increase in [CO_2] after the harvesting traffic observed in spring 2008 than in spring 2009 could be caused by either the beginning of soil restoration two years after the heavy traffic, or it could by the wetter spring in 2008 than in 2009 (190 and 100 mm of rainfall, respectively). The relationship between [CO_2] and the soil climatic variables is weak, particularly in the disturbed soil. We do not have enough data to separate the effects of the pedoclimatic variable control from those of soil regeneration on the CO_2 production/consumption and transfer. Our SE measurements over two years did not indicate any change in the impact of the forwarder traffic on SE: as the measured SE data did not provide clues about changes in the soil gas transport, we could not draw any conclusion on the beginning of soil recovery to the undisturbed levels of this transport. Only a long-term monitoring of [CO_2] and SE and also soil climatic conditions will aid in testing the hypothesis that soil restoration results in undisturbed levels of gas production/consumption and transfer.

Conclusion

Monitoring a Luvisol (ruptic) soil in northeast France for two years has shown that mechanized harvesting of the forest significantly decreased soil CO_2 efflux. This decrease is attributed to the reduction in soil porosity observed a few weeks after the traffic. Soil compaction led to an increase in the frequency and intensity of periods where the soil was water-filled or water-logged and thus where poor aeration conditions prevailed. When the soil was dry and well-aerated, the decrease in soil respiration resulted in a decrease in [CO_2] in the T-treatment compared with the C-treatment. Nevertheless, as [CO_2] is a result of the balance between the gas production/consumption and gas transfer, it may be larger in the compacted than in the undisturbed soils despite the reduced respiration values as soon as water filled the pore space.

Soil CO_2 concentration and efflux thus were very sensitive to soil degradation caused by forest harvesting and provide complementary information on the impact of soil physical degradation on the CO_2 production/consumption and transport in the soil. The monitoring duration of two years for these two variables and soil climatic properties at the experimental site in the ‘Hauts Bois’ forest was not long enough to whether soil recovery to undisturbed levels had begun. Nevertheless, even though soil CO_2 concentration and efflux perform well as tools to assess soil degradation, their use as simple indicators is doubtful because the impact of soil degradation on these two soil parameters strongly varied with time and was also probably dependent on soil type.

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V

FORWARDER TRAFFIC IMPACTED OVER AT LEAST FOUR YEARS SOIL AIR COMPOSITION OF TWO FOREST SOILS IN NORTHEAST FRANCE

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Abstract

Soil atmosphere composition results from a balance between biological activity and gas transfer, both likely to be affected by soil compaction following heavy traffic. We monitored soil atmosphere composition, temperature and moisture once a month during three years in the trafficked (wood-loaded forwarder) and undisturbed plots of two sites in the NE part of France. Our aim was to assess the impact of compaction on soil air composition and to test if soil restoration resulted in undisturbed levels of gas production/consumption and transfer. Soil air oxygen, O_2 and carbon dioxide, CO_2 contents were the two soil gases the most sensitive to compaction and climatic variations. During at least the first year following compaction at 30 cm depth, heavy traffic resulted in an increase in soil atmosphere CO_2 concentration ($[CO_2]$) whatever the air-filled pore space. Following the first soil drought experimented at both sites, this initial impact disappeared toward an effect alternating between an increase in $[CO_2]$ when water filled pore space and a decrease when the soil was dry. The same interchanging impact was observed for soil air O_2 content but with opposite trends. We assumed that soil cracks formed in the trafficked treatment due to lower resistance to stresses when the soil dried out drastically during summer droughts, resulting in an increase in soil gas diffusion while considering same soil temperature and air-filled porosity. However three to four years following heavy traffic soil air-filled porosity was still significantly decreased and gas production/consumption seemed to be still affected by compaction.

Abbreviations: C, control; T, trafficked; $[O_2]$, soil air O_2 (oxygen) content; $[CO_2]$, soil air CO_2 (carbon dioxide) content; $[N_2O]$, soil air N_2O (nitrous oxide) content; ε_A, soil air-filled porosity; T_s, soil temperature.

Introduction

Gas exchanges between the soil and the atmosphere (soil aeration) play an important role in sustaining a favorable soil environment for root growth and biological activity (Gaertig et al. 2002; Lipiec and Hatano 2003; Von Wilpert and Schäffer 2006). Besides poor gas transfers through the soil may cause high nitrous oxide (N_2O) (Ball et al. 1999) and methane (CH_4) fluxes (Teepe et al. 2004), changes in nutrient transformation and availability through modifications in soil pH and redox potential (Stępniewski and Stępniewska 2009) and alteration of soil physico-chemical environment (Herbauts et al. 1996).

Heavy traffic can cause drastic changes in soil porosity and structure, i.e. a diminution of the total volume of soil voids, and a change in pore size distribution and relations (e.g. Herbauts et al. 1996; Richard et al. 2001). In particular, by decreasing the volume of macropores and soil pores connectivity, traffic using heavy machinery may impact soil water and gas transfers, affecting directly soil aeration. Indeed, soil compaction affects air permeability and gas diffusion coefficient by decreasing the volume of pore space occupied by air and pore continuity (Lipiec and Hatano 2003). Besides saturated soil hydraulic conductivity was found to be reduced (Richard et al. 2001) and soil water retention was found to be increased by compaction (Frey et al. 2009) increasing the risk of water logging following heavy rain events.

The changes in soil structure and gas and water transfers are likely to modify soil biological activity (Teepe et al. 2004; Frey et al. 2009). The optimal soil water content for aerobic microbial activity depends significantly on soil bulk density among others soil parameters (Schjønning et al. 2003). Carbon and nitrogen mineralization have been found to depend on the volume of pore space occupied by water (Schjønning et al. 2003) which is increased by soil compaction (Teepe et al. 2004; Frey et al. 2009). Decreased rooting intensity due to heavy forestry traffic (Von Wilpert and Schäffer 2006) will reduce soil respiration (Hanson et al. 2000).

Schnurr-Pütz et al. (2006) found changes in soil microbial communities due to compaction; compacted soils exhibiting higher potential in anaerobic microbial activities than the undisturbed soils. According to their results CH_4 and N_2O production may be enhanced following heavy traffic. Besides, anaerobic soil conditions have been found to decrease carbon dioxide (CO_2) production and oxygen (O_2) consumption as the mineralization of organic matter is reduced (Simojoki and Jaakkola 2000; Stępniewski and Stępniewska 2009). Even if CO_2 production may be decreased by soil compaction, soil air CO_2 concentration can be increased due to decreased gas-diffusion coefficient (Gaertig et al. 2002). In the same way frequent periods of low soil air O_2 concentration were found following compaction and were associated to recurrent periods of lower air-filled porosity (Weisskopf et al. 2010).

Soil air composition results from a balance between gas production/consumption and gas transfers through the soil (Stępniewski and Stępniewska 2009). Therefore the relationships between soil deformation beneath heavy machinery and soil air composition may be a relevant tool to quantify the impact of soil compaction on its capacity to provide “habitats” and to help us understanding the resulting soil gas effluxes. Besides monitoring soil atmosphere composition in relation with soil climatic conditions (temperature and moisture) was found to be sensitive to deformation caused by heavy traffic and give information about changes in soil gas production/consumption and transfer conditions (Goutal et al. 2011a). Knowledge about the initial impact of heavy traffic on soil air composition and the time needed to regenerate suitable levels of soil aeration and biological activity are needed to evaluate the entire risk of soil degradation following traffic of heavy machinery. Especially in forest ecosystems, where tillage and/or fertilization (as in agriculture) are missing in forest management practices, the knowledge of the time needed to recover suitable level of soil biological activity and gas transfer will help the decision making process.

We worked on two experimental sites set up under similar conditions to assess the impact of a wood-loaded forwarder on soil air composition during the first four years following compaction and with neither soil alleviation by tillage nor soil liming to improve soil biological structure regeneration. The soils of both sites were very sensitive to compaction according to their texture, but also because a textural discontinuity at 50 cm depth caused limited temporary water logging which could be easily further degraded. Indeed, at one of both sites, the forwarder traffic resulted in changes in soil air CO_2 content due to hypoxic conditions (Goutal et al. 2011a). Our aim was to test if soil restoration following disturbance by heavy traffic resulted in undisturbed levels of gas production/consumption and transfer. To fulfil this objective we assessed the impact of the same wood-loaded forwarder on soil air composition and its evolution, and decomposed this impact using the relationships between soil atmosphere composition and soil climatic (temperature and moisture) conditions.

Material and methods

Study sites

The two sites studied are located in two state-owned forests in northeast France; near Azerailles (AZ), Meurthe et Moselle and near Clermont en Argonne (CA), Meuse. They were described in Goutal et al. (2011a and b). The two sites presented similar soil morphology classified as Luvisol (ruptic) (IUSS Working Group WRB 2006). Both soils were developed on a silt loam layer approximately 50 cm thick laying on a clayey layer; this strong textural discontinuity causing limited temporary water logging (iron-manganese concretions at the lower part of the silt loam layer). Yet some differences in soil characteristics especially clay content and mineralogy, pH and base saturation offered the potential to assess differential soil behavior under and after identical heavy machinery constraints. The soil at AZ had a pH varying between 4.6 and 5.2 throughout the soil profile (0–1 m) with 22 to 32 and 50 to 56 % of clay and silt respectively from the surface to a depth of 50 cm and with 45 to 59 and 32 to 43 % of clay and silt respectively from 50 cm to 1 m depth. Whereas at CA soil pH varied between 4.4 and 5.1 throughout the soil profile (0–1 m), with 13 to 21 and 65 to 72 % of clay and silt respectively from the surface to a depth of 50 cm and with 28 to 33 and 55 to 59 % of clay and silt from 50 cm to 1 m depth. At AZ, soil base saturation decreased with increasing depth from 61 to 31 % (0–50 cm) and increased again to 94 % (50–100 cm), whereas at CA it decreased from 29 to 13 % (0–50 cm) and increased again to 56 % (50–100 cm). Clay minerals composition was kaolinite and illite at both sites, associated with chlorite and interstratified illite-chlorite at AZ and vermiculite, smectite and hydroxy-interlayered clay at CA. The humus forms varied from Mull mesomull at AZ to Mull dysmull and moder at CA.

The two previous forest stands dominated by beech (*Fagus sylvatica* L.) and oak (*Quercus petraea* L.) were clear-cut over a 5 ha surface area for each site and timber was extracted using a cable yarding system. All remaining slashes were removed by hand or using an iron horse. Afterwards, in May 2007 (AZ) and March 2008 (CA) the same 8-wheel drive forwarder (1996 Valmet 840, serial number 9146, Valmet Logging, Sweden) drove for an equivalent of one pass back and forth. The tyres were inflated to a pressure of 360 kPa and the wood-loaded forwarder weighted 17 (CA) to 23 Mg (AZ). The mean soil water content at the time of traffic varied between 0.27 (10–50 cm) and 0.34 g g^{-1} (0–10 cm) at AZ, and between 0.30 (10–50 cm) and 0.49 g g^{-1} (0–10 cm) at CA. The wheel tracks were adjacent to each other creating an equally trafficked 30 × 50 m surface area in each of the three blocks of each site. In the three blocks, land strips of same surface area and adjacent to the trafficked plots (T-treatment) remained undisturbed by heavy traffic and were considered as control plots (C-treatment).

Ground vegetation cover was estimated on both sites during the growing season as described in Goutal et al. (2011a). In 2009, the normalized cover percentages of *Anemone nemorosa* L., *Glyceria striata* Lam., *Juncus* sp. and *Rubus fruticosus* L. were 19, 7, 3 and 71% respectively for the C-treatment and 5, 44, 43and 8% for the T-treatment at AZ. At CA, the normalized cover percentages of *Carex palescens* L., *Hypericum pulchrum* L., *Juncus* sp., and *Rubus* sp. (*fruticosus* L. and *idaeus* L.) were 9, 4, 16 and 71% respectively for the C-treatment and 6, 1, 68 and 26% for the T-treatment (Brêthes A., unpublished data).

The entire AZ and CA sites areas were planted in autumn 2007 and autumn 2008 respectively with 1600 plants ha^{-1} of sessile oak (*Quercus petraea* L.).

Soil atmosphere composition

Soil atmosphere was sampled monthly during three years using gas collectors inserted vertically into the soil in February 2008 (AZ) and by the End of March 2008 (CA). Soil atmosphere monitoring procedure has already been described in Goutal et al. (2011a). In brief, each gas collector consisted in a soil air equilibration chamber buried at the depth studied and connected by a stainless-steel pipe to a sampling port (closed by a silicon rubber septum) at the soil surface. The depths investigated were 10, 25, 35, 50 and 65 cm in the C-treatment and 5 to 10 cm shallower at each depth in the T-treatment according to the mean depth of the rut where the gas collectors were inserted. The depths of all the monitoring sensors (see also further) with reference to the soil surface differed between treatments to allow investigations of the same soil layer. The interval between collectors was approximately 0.5 m. In both sites we sampled three permanently established collectors per depth and treatment (one in each block), except for the first depth in the T-treatment (five replicates) and for the AZ site where five more replicates per treatment were inserted at 30 (C-treatment) and 25 (T-treatment) cm depth near the soil climate sensors. From March 2008 (AZ) or May 2008 (CA) to May 2011, once a month and within two days we collected soil air samples in a 15 ml vacutainer during morning on both sites. In this study, we only considered the May 2008–May 2011 period for the AZ site (see Goutal et al. 2011a for the March 2008–May 2008 period) and the May 2008–March 2011 period for the CA site to enclose in each case three years following traffic (2d to 4th at AZ and 1st to 3d at CA). The composition of soil air samples (N_2, O_2, CO_2, N_2O and CH_4) was determined at the laboratory (at 20 °C) using a gas chromatograph (CP 4900, Varian, France) within a maximum of two to three days after field sampling. The soil air sample was simultaneously injected into two columns (poraPLOT Q and Molecular Sieve 5 Å) with a helium carrier and was measured by a thermal conductivity detector (TCD). Although this gas chromatograph has a detection threshold of 8 ppm concerning CH_4 concentration, only very small amount of CH_4 could be observed and the results for this gas were inconsistent. Consequently we did not analyze the effect of treatment and soil recovery on soil air CH_4 contents. By comparing the gas concentration measured by the GC upon calibration with the actual value given for the different gas standards used for calibration, we calculated a relative error of maximum 4, 2, and 5 % of the actual value, for [N_2O], [O_2], and [CO_2] respectively.

Soil climatic conditions

Rainfall, air and soil temperature, and soil water content were recorded every two hours on two data loggers (DL2e Data Logger, Delta-T devices Ltd, England for rainfall and soil temperature, and Trase “B.E.” by Soil Moisture, Sols Mesures, France for soil water content) located approximately in the centre of each site. The climate recorded on both sites during the study period showed that the CA site experimented three periods of drought; August 2009, July 2010 and May 2011 (Fig.1, also validated by the soil moisture measurements) whereas we observed only two periods of drought (June 2008 and May 2011) at AZ.

Time domain reflectometry (TDR) probes and temperature sensors (five replicates per depth, treatment and site) were inserted into the soil at three depths (15, 30 and 60 cm in the C-treatment; 10, 25 and 55 cm in the T-treatment). The soil dielectric constant values measured by the TDR probes were converted to volumetric water content using calibration curves fitted for each soil layer and site (Goutal et al., 2011a; appendix 1).

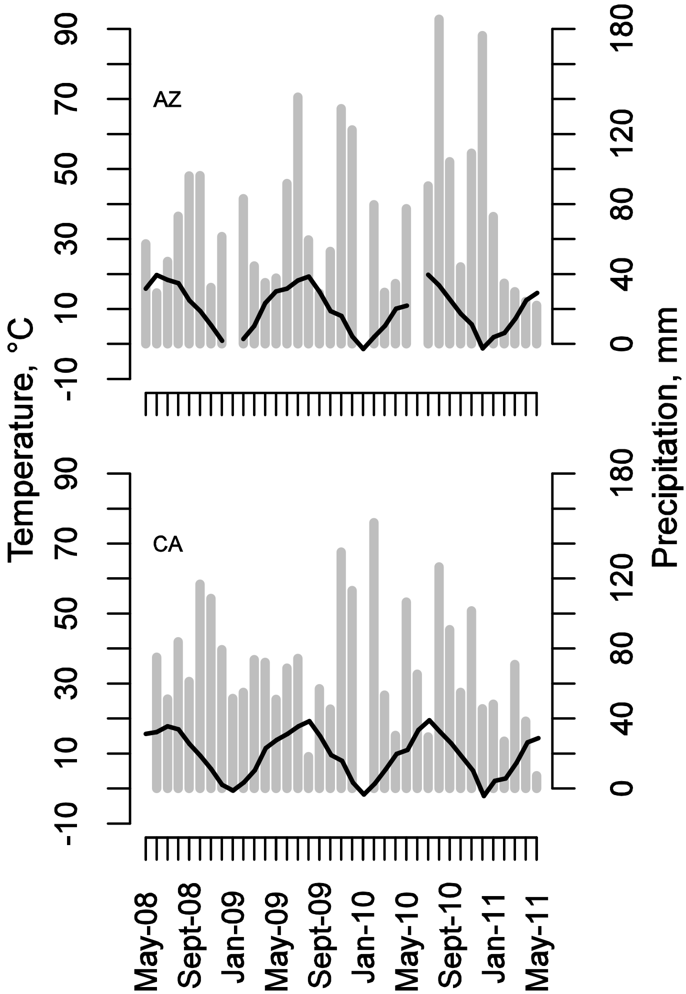

Fig. 1 Monthly air temperature (line) and precipitation (grey bars) during the soil air monitoring period for the sites of Azerailles (AZ) and Clermont en Argonne (CA); precipitation data were missing for the months of January 2009 (AZ), January 2010 (AZ and CA) and June 2010 (AZ) due to technical problems, especially due to a lack of energy supply to the data loggers (solar panels)

Calculation of soil air-filled porosity

The proportion of soil volume occupied by the gas phase (air-filled porosity, ε_A) is an important parameter in gas diffusion models (Moldrup et al. 2004). Therefore we calculated ε_A using the TDR measurements:

$$\varepsilon_A = 1 - \frac{BD}{\rho_S} - WC_v$$

With ε_A, soil air-filled porosity (m^3 soil air m^{-3} soil); BD, soil bulk density (g cm^{-3}); ρ_S, density of solid (g cm^{-3}); WC_V, soil volumetric water content (m^3 m^{-3}).

The bulk density of the CA and AZ soils varied with the water content and the relationship between BD and WC was influenced by the treatment, therefore we used the soil shrinkage curves presented in Goutal et al. (2011b) to calculate the mean of bulk density per 5 % increment in soil volumetric water content. For each daily averaged TDR measurement we could then calculate the mean bulk density of the corresponding volumetric water content increment. At AZ, soil bulk density ranged from 1.23 (at a water potential of -10 cmH_2O) to 1.38 g cm^{-3} (air-dry) and from 1.52 (-10 cmH_2O) to 1.56 g cm^{-3} (air-dry) at a depth of 15 cm in the C- and T-treatments respectively. At CA, it ranged from 1.35 (-10 cmH_2O) to 1.37 g cm^{-3} (air-dry) and from 1.47 (-10 cmH_2O) to 1.48 g cm^{-3} (air-dry) in the C- and T-treatments respectively. The density of the solid was measured on dried and ground (200 μm) soil samples (one per block × depth, only in the C-treatment) using pycnometers (1305 by Micromeritics, USA). For the AZ site, ρ_S averaged 2.62 (standard deviation (SD): 1×10^{-3}), 2.66 (SD: 2×10^{-3}), and 2.69 (SD: 2×10^{-3}) for the 10–20, 30–40 and 50–60 cm soil layers, respectively. For the CA site, ρ_S averaged 2.62 (SD: 2×10^{-3}), 2.65 (SD: 2×10^{-3}), and 2.66 (SD: 2×10^{-3}) for the 10–20, 30–40 and 50–60 cm depths, respectively. Goutal et al. (2011b) observed a significant bulk density recovery during the three (CA) to four (AZ) years after heavy traffic, but only in the surface soil layer. We neglected the recovery of soil bulk density with time as we considered only the depths of 15, 30 and 60 cm. The shrinkage curves were only measured at 15 cm depths in both treatments. To calculate the BD for each WC increment at 30 and 60 cm depths, we only added the corresponding depth effect on BD assuming that the change in shrinkage curve with depth could be neglected and therefore that the depth effect on BD was not dependent on WC. It was probably not true as clay content increased with depth for both soils, yet as clay content did increase the same way in both treatments we considered ε_A as a value allowing for comparison of gas diffusion conditions between treatments.

Statistical analysis

In each site to test the effect of treatment and its evolution in time we chose linear mixed model as they are adapted to longitudinal data and to data sets with missing values (gas collectors filled with water occasionally). We observed an impact of heavy traffic varying with the depth considered, therefore to assess the impact of compaction and recovery time on soil air composition we worked on each soil layer separately:

$$y = \mu + b \times Ti + c \times Tr + d \times Tr : Ti + (1|Co) + \varepsilon$$

With y, the soil gas content considered (CO_2, O_2, or N_2O, in % vol.); Ti, time since soil disturbance (in weeks); Tr, treatment applied (two levels; C or T); Co, soil gas collector (each gas collector is labeled with a unique number, e.g. the gas collector in the block I and the C-treatment at 10 cm depth has the number 1, the next and deeper one in the same block and treatment has the number 2...). The effect of the sampling location (gas collector) was considered as a random effect, i.e. inducing random variations around the fixed and reproducible effects. The residuals were assumed to be auto-correlated as a function of time within each sampling location. Therefore the variance-covariance structure of the residuals was modeled with an auto-correlation structure of lag 1 with a time covariate (Ti) within each gas collector (Co). Indeed we assumed that the closer in time the observations from the same gas collector the more they were correlated and that the observations from different gas collectors were uncorrelated.

As we also assumed that the impact of heavy traffic on soil atmosphere composition would depend on soil climate conditions we also tested for each of the three depths equipped with temperature and moisture sensors the following model:

$$y = \mu + e \times \varepsilon_A + f \times T_S + b' \times Ti + c' \times Tr + \gamma + \varepsilon$$

With ε_A, soil air-filled porosity (m^3 m^{-3}); T_s, soil temperature (°C); γ, interactions between the different predictor of y. As soil temperature and moisture sensors were only inserted in one block, we worked on the mean per depth and treatment of soil temperature and water content and y was the averaged soil gas content of the three blocks for the depth, treatment and gas considered. Therefore no random effect was accounted for and the variance-covariance structure of the residuals was modeled with an auto-correlation structure of lag 1 with a time covariate (Ti) within each treatment (Tr).

The linear mixed models were fit using the Maximum Likelihood (ML) procedure of the R software. Selection of the best model was based on the distribution of the residuals, the Aikake information criterion (AIC), Bayesian information criterion (BIC) and the likelihood ratio Chi-2 test (LRT) for nested models.

Finally the strengths of the relationships between the different paired soil gases concentrations within each treatment were assessed using the Kendall rank correlation coefficient. In the following sections, the term significant refers to a significance at a probability α-level of 0.05.

Results

Soil atmosphere composition evolution following to heavy traffic

The impact of heavy traffic on soil air CO_2 and O_2 concentrations ([CO_2] and [O_2] respectively) depended on soil depth, season and year after compaction (Fig. 2). We observed a slight or absent impact in the surface layers of both sites. The difference in [CO_2] between the T- and C-treatment became more visible below 40 or 20 cm depth at AZ or CA, respectively.

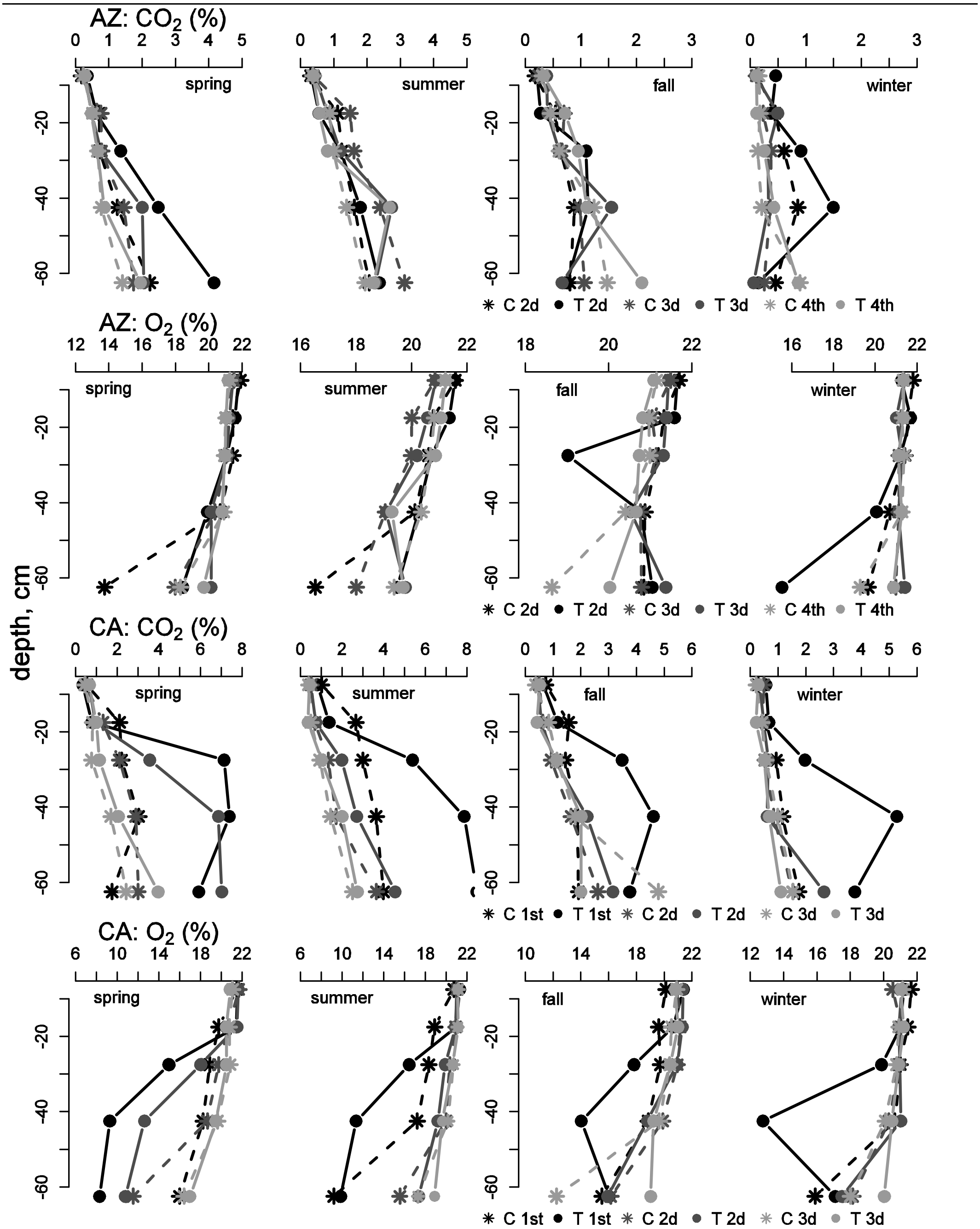

Fig. 2 Averaged soil air CO_2 and O_2 contents by depth, treatment, season and number of year after compaction (3 replicates × about 4 sampling times per season, depth, treatment and number of year after compaction): AZ, Azerailles; CA, Clermont en Argonne; C, control treatment; T, trafficked treatment; 1st, 2d, 3d, 4th, first, second, third and fourth year after compaction

Table 1 Selected mixed linear models fitted to explain soil CO_2 ([CO_2]) and O_2 ([O_2]) contents (in % vol.) of the different soil layers investigated at Azerailles (AZ) and Clermont en Argonne (CA) as a function of the effects of treatment (Tr, two levels: control, C and trafficked, T) and time since treatment application (Ti, in number of weeks); selection of the significant effects was based on the AIC, BIC and p-value of the likelihood ratio Chi-2 test (LRT) for nested models after analysis of the distribution of the residuals

Site and soil layer	Soil gas	μ[a]	Tr	Ti	Tr :Ti	AIC	BIC	SD Co	RSD	Phi (~Ti|Co)
AZ 5 and 10 cm	[CO_2]	0.29 (0.02)	n.s.	n.s.	n.s.	-22	-7	0.04	0.23	0.61
	[O_2]	21.8 (0.1)	n.s.	-0.0031 (0.0008)	n.s.	459	478	$2×10^{-5}$	0.54	0.56
AZ 15 and 20 cm	[CO_2]	0.62 (0.07)	n.s.	n.s.	n.s.	236	249	0.09	0.48	0.85
	[O_2]	21.1 (0.07)	n.s.	n.s.	n.s.	371	384	0.09	0.61	0.77
AZ 25 and 30 cm	[CO_2]	1.4 (0.1)	n.s.	-0.0045 (0.0009)	n.s.	960	981	0.26	0.64	0.76
	[O_2]	20.88 (0.09)	n.s.	n.s.	n.s.	1491	1508	0.16	1.15	0.81
AZ 40 and 45 cm	[CO_2]	2.2 (0.3)	n.s.	-0.006 (0.002)	n.s.	554	570	0.19	1.05	0.82
	[O_2]	20.4 (0.1)	n.s.	n.s.	n.s.	581	594	0.15	1.11	0.80
AZ 60 and 65 cm	[CO_2]	1.7 (0.2)	n.s.	n.s.	n.s.	600	613	0.24	1.39	0.64
	[O_2]	17.6 (0.9)	n.s.	0.012 (0.005)	n.s.	836	852	1.01	2.76	0.69
CA 5 and 10 cm	[CO_2]	0.77 (0.08)	n.s.	-0.0029 (0.0006)	n.s.	305	323	0.15	0.40	0.53
	[O_2]	21.1 (0.1)	n.s.	n.s.	n.s.	757	772	0.24	0.90	0.56
CA 15 and 20 cm	[CO_2]	1.7 (0.2)	n.s.	-0.009 (0.001)	n.s.	413	430	0.34	0.68	0.80
	[O_2]	19.7 (0.3)	1.3 (0.5)	0.009 (0.003)	-0.01 (0.004)	554	577	0.26	0.97	0.81
CA 25 and 30 cm	[CO_2]	2.4 (0.8)	3 (1)	-0.013 (0.006)	-0.026 (0.009)	702	725	0.75	1.62	0.88
	[O_2]	19.1 (0.8)	-3 (1)	0.014 (0.006)	0.021 (0.009)	822	845	0.85	1.94	0.80
CA 40 and 45 cm	[CO_2]	3.1 (0.6)	4.7 (0.9)	-0.013 (0.006)	-0.04 (0.009)	739	762	0.42	1.77	0.82
	[O_2]	18.1 (0.8)	-8 (1)	0.015 (0.008)	0.06 (0.01)	883	905	$2×10^{-4}$	2.49	0.78
CA 60 and 65 cm	[CO_2]	2.7 (0.7)	5 (1)	0.0005 (0.007)	-0.04 (0.01)	844	867	0.37	2.22	0.81
	[O_2]	12 (1)	-2 (2)	0.03 (0.01)	0.04 (0.02)	1120	1143	$4×10^{-4}$	4.40	0.77

[a] only the significant coefficient (at a α-level of 0.05 probability) are given with their standard errors in parentheses; n.s., non significant; μ, model intercept ([CO_2] or [O_2] of the control treatment immediately after compaction); AIC, Aikake information criterion; BIC, Bayesian information criterion ; SD Co, standard deviation of the random effect (gas collector; Co); RSD, residual standard deviation; Phi AR(~Ti|Co) value of the lag 1 autocorrelation of the residuals as a function of time within each gas collector

At AZ during the fourth year after compaction differences in [CO_2] between the T- and C-treatment were slight but still appeared at several seasons and depths (e.g. summer 45 cm or fall 30 and 65 cm). We also observed a regular decrease in [CO_2] at 65 cm in the T-treatment in comparison to the C-treatment or in comparison to the upper soil layers at AZ. At CA the [CO_2] was strongly increased in T-treatment in comparison to the C-treatment, independently of the season considered. This strong impact of heavy traffic on [CO_2] was present until summer of the second year after compaction coinciding with the first period of drought experimented at CA (Fig. 1).

At AZ, no significant time by treatment interaction and no significant effect of treatment on [CO_2] could be detected whatever the depth (Table 1); only a negative effect of time on [CO_2] was significant at 30 and 45 cm depth in both treatments. The effect of treatment on [CO_2] was significant at CA but only below 20 cm depth. At CA, a significant decrease in [CO_2] with time was observed in the C-treatment at every soil layers except in the deepest one where [CO_2] increased significantly at a very slow rate. In the soil layers where heavy traffic had a significant effect (below 20 cm), [CO_2] decreased faster as a function of time since compaction than in the C-treatment at CA (Table 1). Therefore below 20 cm depth at CA soil compaction increased significantly [CO_2] yet this impact was decreasing throughout the three years following heavy traffic which can also be seen from Fig. 2.

The impact of heavy traffic on [O_2] showed opposite trends in comparison to [CO_2] (Fig. 2). Intermediate decrease in [O_2] owing to compaction often corresponded to increase in [CO_2]. Yet some decreases in [O_2] due to soil compaction were not associated with increases in [CO_2], especially in the deepest soil layer during spring and summer (e.g. the C-treatment had a lower [O_2] whereas it had lower or equal [CO_2] than the T-treatment during spring or summer at 60 cm depth). In both sites and in each treatment, when [O_2] decreased, [CO_2] increased most of the time (Table 2). The correlation coefficients between soil [O_2] and soil [CO_2] were significantly negative all depths confounded and were the highest in comparison to all other paired soil gases contents. At AZ the coefficient of correlation was slightly higher in the C- than in the T-treatment.

Table 2 Kendall rank correlation coefficients between soil air content in CO_2 ([CO_2] in % vol.) or O_2 ([O_2] in % vol.) and soil air-filled porosity (ε_A, m^3 m^{-3}) or soil temperature (T_s, °C), and between soil air content in the different gases measured at the Azerailles (AZ) and Clermont en Argonne (CA) sites (all depths confounded)

	AZ C-treatment		AZ T-treatment		CA C-treatment		CA T-treatment	
	[O_2]	[CO_2]	[O_2]	[CO_2]	[O_2]	[CO_2]	[O_2]	[CO_2]
ε_A	0.27***	-0.12*	0.17**	-0.11*	0.47***	-0.39***	0.36***	-0.30***
T_s	-0.28***	0.40***	-0.22***	0.31***	-0.13*	0.25***	-0.12*	0.24***
[CO_2]	-0.67***		-0.59***		-0.69***		-0.68***	
[N_2O]	0.12***	0.05	0.10***	0.08*	-0.05	-0.09*	-0.09**	0.23***

The results concerning the different effects of treatment and time on [O_2] differed from [CO_2] especially in the deepest soil layer (Table 1). The effect of treatment on [O_2] was not significant at AZ whatever the depth considered whereas at CA it was significant in the three deepest soil layers but also at 20 cm depth. Heavy traffic decreased [O_2] but only below 20 cm depth at CA, above it increased or had no effect. In both treatments at AZ, [O_2] increased significantly with time at 65 cm depth and decreased at 10 cm depth. In the C-treatment at CA, [O_2] increased significantly with time at every depths except 10 cm whereas in the T-treatment it did not change at 20 cm and increased faster than in the C-treatment below 20 cm. The intercepts (C-treatment immediately following traffic) of the model explaining [O_2] decreased with depth at both sites, the opposite was found for [CO_2] except at the deepest soil layer. At 60 cm depth the model intercept for [CO_2] decreased in comparison to the upper layers whereas the intercept for [O_2] still decreased. The model intercepts were higher for [CO_2] and lower for [O_2] at CA than at AZ (Table 1). The auto-correlation structure of the residuals as a function of time within each gas collector was significant for the two gases (CO_2 and O_2) studied and a very uneven part of the variation (between 3 and 20% for CO_2 and between less than 1 and 16% for O_2) was due to the random structure (Table 1).

At both sites, soil air N_2O concentrations ($[N_2O]$) increased in the T- in comparison to the C- treatment, but only below 30 cm depth (Fig. 3). In the upper soil layers it was the opposite. At AZ no N_2O could be detected in soil air since the end of the second year after compaction (detection threshold of 4 ppm), and $[N_2O]$ measured otherwise corresponded mainly to peaks giving therefore a very low seasonal mean. At CA, we still detected N_2O in soil air even if $[N_2O]$ was lower than for the two first years after compaction. At AZ, the Kendall correlation coefficients between $[O_2]$ or $[CO_2]$ on the one hand and $[N_2O]$ on the other hand were positives in both treatments even if they were small (Table 2). At CA, they were negatives except for the relationship between $[CO_2]$ and $[N_2O]$ in the T-treatment where the correlation coefficient was relatively high and positive. As N_2O appeared only occasionally and not throughout the entire study period, testing the significance of the effect of treatment and its evolution with time on soil air concentration in this gas would have had little sense.

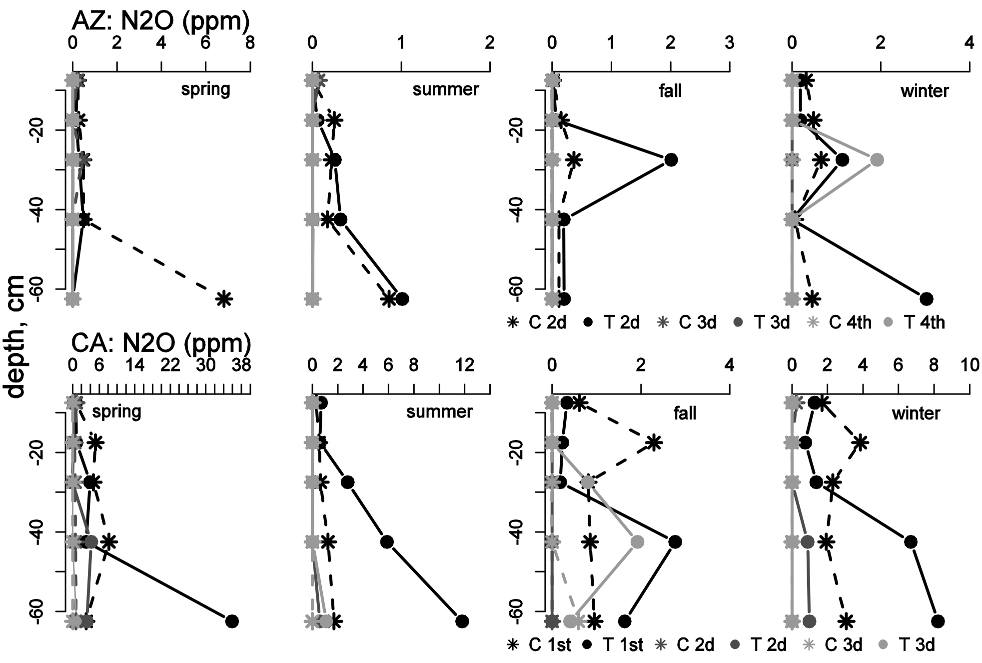

Fig. 3 Averaged soil air N_2O contents by depth, treatment, season and number of year after compaction (3 replicates × about 4 sampling times per season, depth, treatment and number of year after compaction): AZ, Azerailles; CA, Clermont en Argonne; C, control treatment; T, trafficked treatment; 1st, 2d, 3d, 4th, first, second, third and fourth year after compaction

Influence of soil air-filled porosity and temperature on the evolution of soil atmosphere composition

Soil temperature (T_s) was not affected by treatment at any of the depths investigated (p-values for the effect of treatment were always >0.10 for each depth and site studied, see also Fig. 4). On the other hand, soil air-filled porosity (ε_A) was significantly decreased at 15 and 30 cm (AZ), at 15 cm (CA) (Table 3, Fig. 4). At 60 cm depth, traffic decreased soil volumetric water content in both sites, yet as it also decreased porosity no significant effect on ε_A was detected at AZ and it even increased significantly ε_A at 60 cm at CA. For both variables (ε_A and T_s) the auto-correlation structure of the residuals as a function of time within each treatment was significant. Soil temperature and ε_A slightly decreased with the number of weeks since forest harvesting without interactions with the effect of treatment (Fig. 4). The effect of time was only significant for ε_A at AZ (Table 3) and for T_s at CA (p-values of 0.04, 0.07 and 0.10 at 15 cm, 30 cm and 60 cm depth, respectively whereas they were above 0.20 at AZ whatever the depth). At both sites, for the three soil layers investigated and in each treatment, ε_A and T_s were significantly and positively correlated (Kendall rank correlation coefficients of 0.56 and 0.51 at AZ 15 cm; 0.34 and 0.48 at CA 15 cm; 0.55 and 0.58 at AZ 30 cm; 0.48 and 0.52 at CA 30 cm; 0.51 and 0.57 at AZ 60 cm; 0.29 and 0.56 at CA 60 cm depths in the T- and C-treatment respectively).

Significant correlations were found between on one hand T_s or ε_A and on the other hand [CO_2] or [O_2] (Table 2). The relationship between [CO_2] or [O_2] and ε_A was more significant (lower p-value associated to the Kendall rank correlation coefficient) than with volumetric water content. At AZ the correlation coefficients on one hand T_s or ε_A and on the other hand [CO_2] or [O_2] in the C-treatment were stronger than in the T-treatment (Table 2). At CA the correlation coefficients with ε_A were stronger than at AZ. Those with T_s were smaller than at AZ. At AZ in the C-treatment a slightly more significant (p-value= 0.005 instead of 0.006) was found between [N_2O] and soil volumetric water content (-0.20) instead of ε_A. In the T-treatment of the AZ site, no correlation coefficient between [N_2O] and any other soil climatic parameter was significant. At CA it was in the T-treatment that a correlation coefficient between [N_2O] and a soil climatic parameter was significant (-0.14 between [N_2O] and ε_A).

Table 3 Selected mixed linear models fitted to explain soil air-filled porosity ($m^3\ m^{-3}$) of the different soil layers investigated at Azerailles (AZ) and Clermont en Argonne (CA) as a function of the effects of treatment (Tr, two levels: control, C and trafficked, T) and time since treatment application (Ti, in number of weeks); selection of the significant effects was based on the AIC, BIC and p-value of the likelihood ratio Chi-2 test (LRT) for nested models after analysis of the distribution of the residuals

	μ[a]	Tr	Ti	interaction	AIC	BIC	RSE	Phi (~Ti|Tr)
AZ 10 and 15 cm	0.26 (0.03)	-0.08 (0.01)	-4×10^{-4} (1×10^{-4})	n.s.	-281	-269	0.04	0.84
AZ 25 and 30 cm	0.18 (0.01)	-0.10 (0.01)	n.s.	n.s.	-311	-302	0.04	0.85
AZ 55 and 60 cm	0.08 (0.02)	n.s.	-3×10^{-4} (1×10^{-4})	n.s.	-275	-266	0.04	0.85
CA 10 and 15 cm	0.24 (0.01)	-0.08 (0.02)	n.s.	n.s.	-280	-271	0.04	0.89
CA 25 and 30 cm	0.15 (0.01)	-0.03 (0.02) n.s.	n.s.	n.s.	-287	-278	0.04	0.88
CA 55 and 60 cm	0.06 (0.005)	0.038 (0.007)	n.s.	n.s.	-375	-366	0.02	0.81

[a] only the significant coefficient (at a α-level of 0.05 probability) are given with their standard errors in parentheses; n.s., non significant; μ, model intercept (control treatment immediately after compaction); AIC, Aikake information criterion; BIC, Bayesian information criterion; RSE, residual standard error; Phi AR(~Ti|Tr), value of the lag 1 autocorrelation of the residuals as a function of Ti within Tr

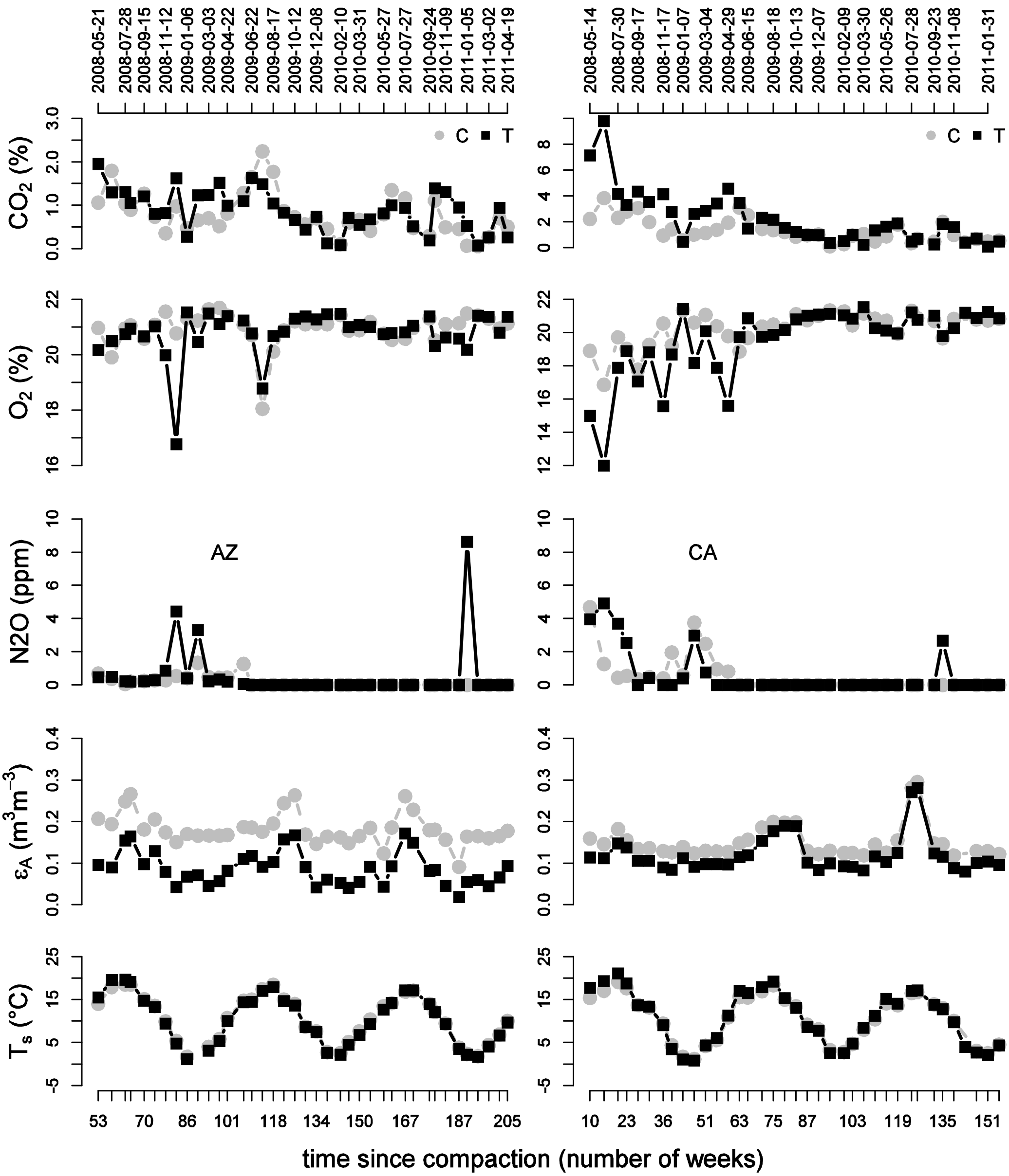

Fig. 4 Concurrent evolution of soil air CO_2, O_2, and N_2O contents (average of the measurements of the three blocks per site), soil air-filled porosity (ε_A) and soil temperature (T_s) at 30 cm in the control (C) treatment and 25 cm depth in the trafficked (T) treatment, in the Azerailles (AZ, left side) and Clermont en Argonne (CA, right side) sites, soil water content and temperature are measured only in one block (five replicates), bulk density (nine replicates) and soil air composition (three replicates) measured in the three blocks

Accounting for the influences of T_s and ε_A was significant at every depth and both sites when assessing the impact of soil compaction and time since heavy traffic on [CO_2] (Table 4). When ε_A increased, [CO_2] decreased in the same way in both treatments (the treatment by ε_A interaction was not significant) for each soil layer and site considered. When T_s increased, [CO_2] increased and no interaction with treatment was found except at 15 cm depth at CA. By accounting for differences in T_s and ε_A between weeks and treatments, the effect of treatment on [CO_2] became significant at 15 and 30 cm depths at AZ. Heavy traffic significantly decreased [CO_2] if we considered same T_s and ε_A in both treatments and no recovery could be detected (the interaction between the effects of treatment and time was not significant) (Table 4, Fig. 5). At CA, by accounting for the influences of T_s and ε_A on [CO_2] the effect of treatment became non significant at 30 cm depth whereas it did not change the effect of treatment and time at 15 and 60 cm depths. The control exerted by T_s and ε_A on [CO_2] changed significantly with time and this may have masked the effect of treatment at 30 cm depth. Besides, if we considered T_s and ε_A equal to their arithmetic mean both treatments confounded (Fig. 5), we observed that the trend in [CO_2] evolution in the T-treatment at 30 cm depth of the CA site was not well described. The quality of description improved when we took T_s and ε_A equal to their arithmetic mean per treatment (Fig. 6). Even the non-significant change in ε_A caused by heavy traffic at 30 cm depth caused drastic changes in [CO_2] evolution as ε_A had a strong influence on [CO_2] in this soil layer and this effect varied significantly with time. By accounting for T_s and ε_A variations, a significant recovery was only detected at 60 cm depth of the CA site, and the recovery rate was faster in comparison to the analysis of the effects of treatment and time without covariates (Table 1). At both sites, if we considered T_s and ε_A equal to their arithmetic mean per treatment we obtained the similar results as when we analyzed the effects of soil compaction and time since this compaction without covariates (e.g. no effect of treatment at AZ but effect of time at 30 cm, see Fig. 6).

Table 4 Selected mixed linear models fitted to explain soil air CO_2 ([CO_2]) and O_2 ([O_2]) contents (in % vol.) of the different soil layers investigated at Azerailles (AZ) and Clermont en Argonne (CA) as a function of the effects of treatment (Tr, two levels: control, C and trafficked, T) and time since treatment application (Ti, in number of weeks) while accounting for soil climatic (temperature and air-filled porosity) influences; selection of the significant effects was based on the AIC, BIC and p-value of the likelihood ratio Chi-2 test (LRT) for nested models after analysis of the distribution of the residuals

Site and soil layer[a]	Soil gas	μ[b]	ε_A	T_s	Tr	Ti	interactions	AIC	BIC	RSE	Phi (~Ti |Tr)
AZ 10 and 15 cm	CO_2	0.8 (0.1)	-3.0 (0.6)	0.041 (0.005)	-0.33 (0.06)	n.s.	n.s.	-49	-35	0.16	n.s.
	O_2	21.3 (0.3)	5 (1)	-0.06 (0.01)	0.5 (0.1)	-0.003 (0.001)	n.s.	82	96	0.38	n.s.
AZ 25 and 30 cm	CO_2	1.6 (0.3)	-6 (1)	0.07 (0.01)	-0.5 (0.2)	-0.0037 (9×10^{-4})	n.s.	60	74	0.33	n.s.
	O_2	20.2 (0.4)	9 (3)	-0.08 (0.02)	0.65 (0.3)	n.s.	n.s.	164	176	0.67	n.s.
AZ 55 and 60 cm	CO_2	-2 (1)	-10 (4)	0.36 (0.08)	n.s.	0.016 (0.007)	T_s:Ti, -0.0013 (5×10^{-4})	197	212	0.93	n.s.
	O_2	20.0 (0.7)	n.s.	-0.11 (0.05)	1.2 (0.6)	n.s.	n.s.	310	319	2.23	n.s.
CA 10 and 15 cm	CO_2	1.8 (0.2)	-5 (1)	0.053 (0.009)	n.s.	-0.005 (9×10^{-4})	T_s:Tr, -0.03 (0.01)	26	44	0.28	-0.76
	O_2	18.8 (0.8)	11 (4)	0.06 (0.04)	0.8 (0.3)	n.s.	ε_A:T_s, -0.4 (0.2)	125	138	0.54	n.s.
CA 25 and 30 cm	CO_2	9 (1)	-70 (11)	0.33 (0.05)	n.s.	-0.06 (0.01)	T_s:Ti, -0.0023 (5×10^{-4}); ε_A:Ti, 0.54 (0.09)	178	199	0.76	0.74
	O_2	13 (1)	71 (10)	-0.32 (0.05)	n.s.	0.07 (0.01)	T_s:Ti, 0.002 (5×10^{-4}); ε_A:Ti, -0.56 (0.09)	213	229	0.95	n.s.
CA 55 and 60 cm	CO_2	1.9 (0.7)	-34 (10)	0.22 (0.04)	6.1 (0.8)	n.s.	Tr :Ti, -0.04 (0.008)	268	284	1.4	n.s.
	O_2	12 (2)	80 (20)	-0.33 (0.09)	-5 (2)	0.02 (0.01)	Tr :Ti, 0.03 (0.02)	380	396	2.96	n.s.

[a] at both sites, the first soil layer (10 and 15 cm) corresponds to 15 cm (respectively 10 cm) for the soil climatic sensors, mean of 5 replicates per treatment, and to 10 and 20 cm (respectively 5 and 15 cm) for the gas collectors, mean of 3 to 8 replicates per treatment, in the C-treatment (respectively T-treatment). The second soil layer corresponds to 30 cm (respectively 25 cm) for the soil climatic sensors and to 30 cm (respectively 25 cm) for the gas collectors in the C-treatment (respectively T-treatment). The third soil layer corresponds to 60 cm (respectively 55 cm) for the soil climatic sensors and to 65 cm (respectively 60 cm) for the gas collectors in the C-treatment (respectively T-treatment)

[b] linear mixed model coefficients, only the significant coefficient (at a α-level of 0.05 probability) are given with their standard errors in parentheses; n.s., non significant; μ,model intercept (control treatment at soil air-filled porosity and temperature of zero and immediately after compaction); ε_A, soil air-filled porosity (m^3 m^{-3}); T_s, soil temperature (°C); AIC, Aikake information criterion; BIC, Bayesian information criterion; RSE, residual standard error; Phi AR(~Ti|Tr), value of the lag 1 autocorrelation of the residuals as a function of Ti within Tr

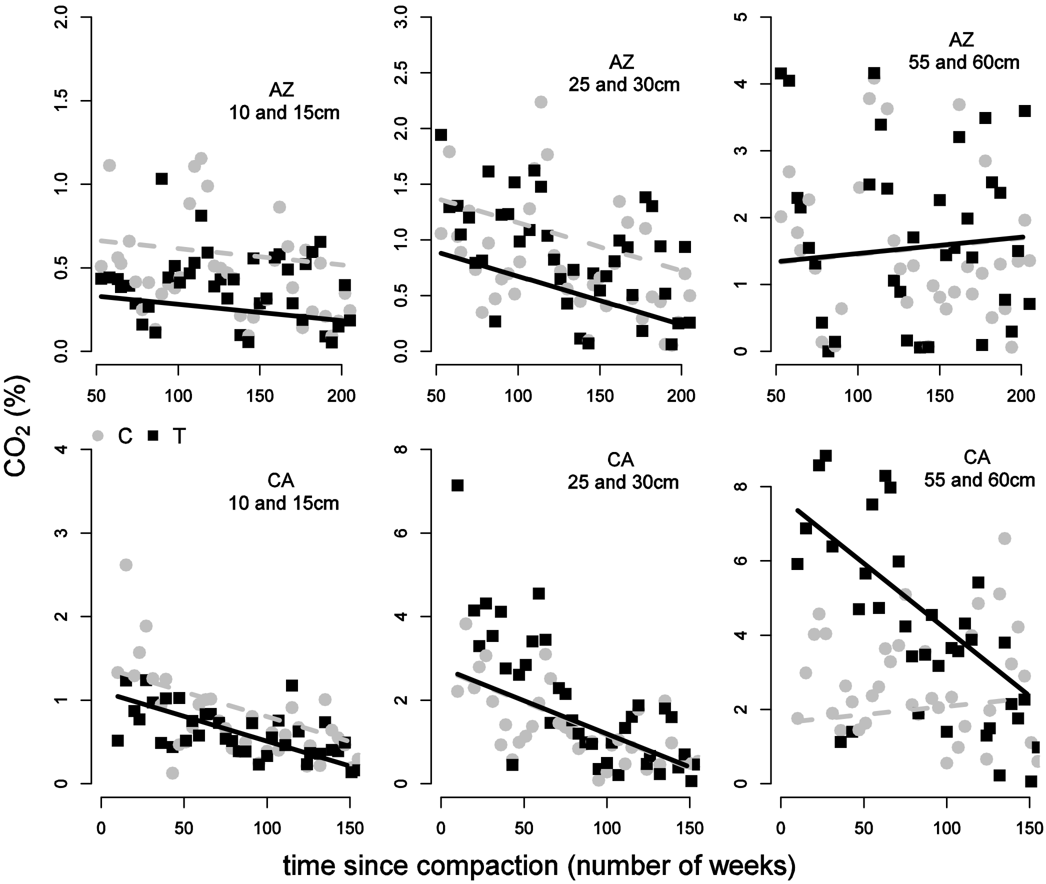

Fig. 5 Evolution of soil air CO_2 content (dots) at the Azerailles (AZ) and Clermont en Argonne (CA) sites, the lines correspond to the linear models explaining this evolution as a function of time following treatment (C, control; T, trafficked) application, treatment, soil air-filled porosity and soil temperature; each covariate being constant, equal to its arithmetic mean per site and depth (both treatments confounded)

Accounting for the influences of T_s and ε_A was significant at every depth and both sites when assessing the impact of soil compaction and time since heavy traffic on [O_2] (Table 4). When ε_A increased, [O_2] increased in the same way in both treatments for each soil layer and site considered. When T_s increased, [O_2] decreased and no interaction with treatment was found. By accounting for differences in T_s and ε_A between weeks and treatments, the effect of treatment on [O_2] became significant at every depth at AZ. Heavy traffic significantly increased [O_2] if we considered same T_s and ε_A in both treatments and no recovery could be detected. At CA heavy traffic significantly increased [O_2] at 15 cm and decreased it at 60 cm depths if we took the same value of T_s and ε_A in both treatments. The same observations as for [CO_2] were made at 30 cm depth of the CA site when analyzing the effect of treatment and time while accounting for T_s and ε_A variations.

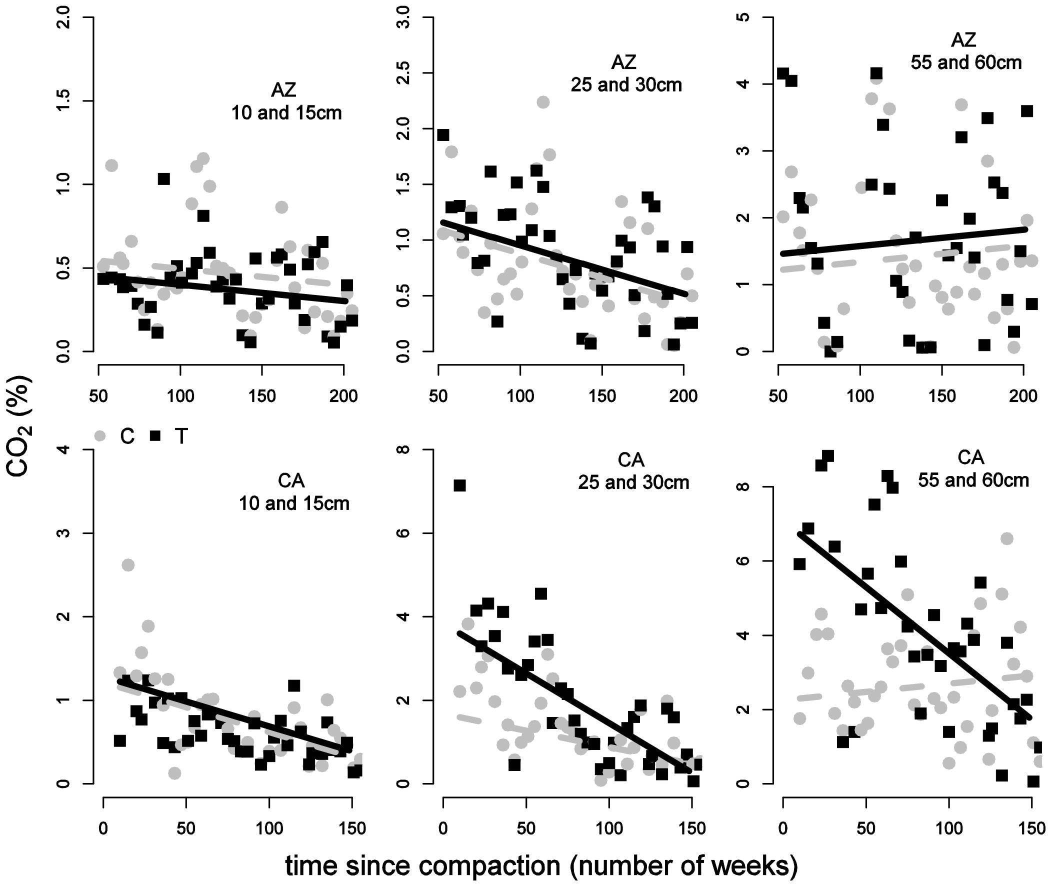

Fig. 6 Evolution of soil air CO_2 content (dots) at the Azerailles (AZ) and Clermont en Argonne (CA) sites, the lines correspond to the linear models explaining this evolution as a function of time following treatment (C, control; T, trafficked) application, treatment, soil air-filled porosity and soil temperature; each covariate being constant, equal to its arithmetic mean per site and depth and treatment

Discussion

CO_2 and O_2

Soil air composition indicated a decrease in [O_2] with increasing depth in the undisturbed soils of both sites (intercepts of Table 1), with the control plots at CA being more hypoxic than at AZ. It was probably caused by the lower ε_A values in the C-treatment at CA than at AZ (Fig. 4, Table 2). Most of the time in the control plots, ε_A values at CA must have been more limiting for gas transfer and must have exerted a stronger control on[CO_2] and [O_2] than at AZ. Indeed at AZ [CO_2] or [O_2] were mostly related to T_s (soil temperature) whereas at CA they were mostly linked to ε_A (soil air-filled porosity) variations.

A decrease in [O_2] mostly corresponded to an increase in [CO_2], yet occasionally a decrease in [O_2] could coincide with a constant or decrease in [CO_2] due to the high water solubility of CO_2 into the surrounding water filled pore space or into the underneath water table (Simojoki and Jaakkola 2000).

At CA below 20 cm depth, whatever the season, [CO_2] (respectively [O_2]) was strongly increased (respectively decreased) during the first one and a half year following the forwarder traffic. We assumed heavy traffic had created an impervious layer to gas that was independent from variations in ε_A (Fig. 4). This gas-impervious layer seems to have been partly removed by the first drought (August 2009). Indeed the impact of forwarder traffic on [CO_2] and [O_2] varied significantly with time since compaction. It may be explained by development of shrinking cracks (Weisskopf et al. 2010). Goutal et al. (2011b) found namely at CA a greater amplitude in soil water content variations in the T- than in the C-treatment explaining the fast bulk density recovery rate. Besides soil disturbance by heavy traffic was found to decrease the ability to withstand stresses caused by decreasing water content (Schäffer et al. 2008). Therefore the decrease in impact of traffic on [CO_2] or [O_2] may have resulted from the effect of traffic on soil ability to form cracks when drying and not from a real improvement in soil structure. Indeed, over the study period, ε_A was significantly reduced by heavy traffic and no regeneration towards undisturbed levels was observed (the treatment by time since compaction interaction was not significant). The impact of compaction on soil atmosphere composition was assessed at AZ only since the second year after constraint and the initial impact may have been as strong as at CA (Goutal et al. 2011a). Besides the first drought at AZ was observed in June 2008 at the beginning of our study period and since this time soil volumetric water content decreased strongly in the T-treatment and became lower than in the C-treatment at 15 cm depth during the following summer period (see Table 4, spring t2 vs. summer t2 in Goutal et al. 2011a). Therefore we assumed that heavy traffic had a strong initial impact on [CO_2] and [O_2] at both sites likely due to the very low ε_A and probable poor pore space connectivity (Richard et al. 2001) of the 0-20 cm trafficked soil layer. It led to hypoxic conditions in the underneath soil layers during the first one (AZ, Goutal et al. 2011a) to one and a half (CA, Figs. 2 and 4) year following the forwarder traffic at both sites. Consequently to this strong initial impact, compaction changed and decreased soil biological activity (Stępniewski and Stępniewska 2009; see also the impact of compaction on soil CO_2 effluxes at AZ, Goutal et al., 2011a). Indeed at both sites, during the second to at least the third (CA) or the fourth (AZ) year following traffic, disturbed soils had lower [CO_2] and higher [O_2] than the control as soon as the air-filled porosity was high and probably sufficient for non limiting gas transfers.

The first drought experimented on both sites led to cracks formation in the T-treatment and lowered the apparent impact on soil atmosphere composition by a probable increase in gas diffusion coefficient at the same ε_A value. Indeed the relationships between [CO_2] and [O_2] on one hand and soil climatic parameters on the other hand were significantly changed with time at 30 cm depth of the CA site. Consequently to soil cracks formation, ε_A and T_s exerted a lesser control on [CO_2] and [O_2]. This could probably be explained by an increased gas diffusion coefficient of the soil layer into consideration but also of the upper soil layers in comparison to the situation before soil cracks formation while considering the same ε_A. At 60 cm depth, on the contrary to the AZ site, at CA the impact of the very low ε_A of the upper trafficked soil layers upon underneath soil layers disappeared only two years after compaction, i.e. half a year after the first drought. This was perhaps due to the very low gas diffusion conditions in the undisturbed soil of the CA site, even a slight decrease in ε_A due to compaction may have more drastic consequences on gas mobility throughout soil profile than at AZ. Besides, at CA, ε_A was higher in deepest soil layer of the T- than the C-treatment. Even though treatment was found to significantly increase [CO_2] and decrease [O_2], indicating a strong control of the gas diffusion conditions of the upper trafficked layers upon the deep one. We assumed that cracks formation had no reached deep enough to allow for gas to escape (or to reach in the case of O_2) from the trafficked deepest soil layers until at least the end of the second year after compaction.

At AZ the significant decrease in ε_A at 15 and 30 cm depths should have led at least to decreased [O_2] (Weisskopf et al. 2010). Yet no significant effect of treatment could be observed neither on [O_2] nor on [CO_2] at AZ. We had to account for ε_A and T_s variations to detect a significantly negative (respectively positive) effect of heavy traffic on [CO_2] (respectively [O_2]) which was in accordance with the negative effect of heavy traffic on soil CO_2 production (respectively O_2 consumption) (Goutal et al. 2011a). Soil compaction decreased soil biological activity at AZ in comparison to undisturbed soil resulting in lower [CO_2] (respectively higher [O_2]) when gas diffusion was not deficient (dry soil conditions). Yet higher [CO_2] (respectively lower [O_2]) could also be observed as soon as water filled soil porosity (wet soil conditions). Therefore without taking into consideration soil climatic influences no effect of soil compaction could be detected on the soil air concentrations in these two gases. This can also be seen from Figs 5 and 6, if we took the averaged values of T_s and ε_A per treatment, very slight effect of compaction on [CO_2] could be observed at AZ. If we considered the same values of T_s and ε_A for both treatments (same gas production/consumption and transfer conditions), the effect of treatment was more obvious. Of course taking the same values of ε_A for both treatments was not realistic with regards to what happened in the field throughout the entire study period (significant decrease in ε_A caused by compaction). Yet it allowed separating the effect of heavy traffic into a decrease in soil biological activity and a decrease in soil gas diffusion. Both impacts resulted in a slight and not significant mean effect of treatment on [CO_2] and [O_2] if we did not consider soil climate conditions. This was probably exacerbated by the fact that high T_s (and high biological production of CO_2 / consumption of O_2) were mostly associated to high ε_A (high gas diffusion) in both treatments; therefore occasions for the T-treatment to differ from the C-treatment were not frequent.

N_2O

Nitrification and denitrification rates depend on biological parameters, soil water filled pore space, temperature and mineral nitrogen contents (Hénault et al. 2005). At both sites, T_s was not affected by heavy traffic, yet soil ε_A was significantly affected. In these conditions, biological activity and mineral nitrogen production may have been altered by heavy traffic. Therefore to explain the higher [N_2O] in soil surface layers of the C- than the T-treatment whereas the opposite was found in the underlying soil layers at both sites, we hypothesized that the mineralization of organic matter decreased in the T-treatment, which is in accordance with results for soil CO_2 efflux (Goutal et al. 2011a). Indeed Li et al. (2003) observed a decrease in nitrogen mineralization following forest harvesting and assumed it was related to poor aeration and limited physical access to organic nitrogen. This was confirmed by the changes in soil water content at both sites, in the T-treatment soil water content was both lower (AZ) or as low (CA) (decreased substrate diffusion) and higher (decreased soil aeration) (Schjønning et al. 2003) than the C-treatment. A decreased aerobic nitrogen mineralization (nitrification) in the T-treatment could have lead to a decreased [N_2O] in the surface layers as impaired gas diffusion conditions may not have prevailed (closeness to the external atmosphere). In the underlying soil layers an occasional accumulation of [N_2O] could be observed due to the more frequent deficient diffusion conditions in the T- than in the C-treatment even if N_2O production was lower due to deficient (limited) aerobic nitrate mineralization. Denitrification potential has been found to increase following soil compaction (Schnurr-Pütz et al. 2006), yet maybe if nitrate and ammonium production is also decreased by soil compaction it may lead to lower [N_2O]. Besides during the first year following compaction at CA very low [O_2] and very high [N_2O] could be observed in the T-treatment, perhaps leading to an exhaustion of soil mineral nitrogen (Vor et al. 2003; Weisskopf et al. 2010) which was further not enough provided by the decreased mineralization rates. This assumption was comforted by the presence of N_2O mainly during the two first years following clear cut; the immediate change in vegetation cover may have increased nitrogen mineralization due to changes in soil climate (e.g. T_s decreasing with time since harvesting in both sites and both treatments). Other causes for soil mineral nitrogen decrease following forwarder traffic could be lixiviation and/or increased plant absorption as we observe very different plant covers between the two treatments. Indeed Simojoki and Jaakkola (2000) observed a decrease in N_2O emissions in the presence of crop growth in comparison to bare soil. It was attributed mostly to uptake of mineral nitrogen by plants and secondly by water transpiration which increased ε_A in a soil containing relatively few nitrates.

Most of N_2O or nitrate production could have been lost to N_2 (Teepe et al. 2004). For example, the opposite behavior between the two sites observed concerning the relationship between [O_2] and [N_2O] or [CO_2] and [N_2O], could have been caused by the slightly more hypoxic conditions at CA than AZ. Yet the assumption that most of the N_2O produced was further reduced to N_2, would not explain why more [N_2O] was found in the deep soil layers of the T- than the C-treatment.

Whatever the causes of the differential response of [N_2O] to soil compaction, we could not conclude with regards to soil regeneration towards undisturbed levels of N_2O production and transfer. Indeed no or very few [N_2O] could be found in both treatments three years after disturbance. Besides the poor relationship strength with soil climatic parameters prevented any analysis of the respective influences of treatment on N_2O production and transfer.

Conclusion

Soil air O_2 and CO_2 contents were highly sensitive to soil structure degradation by heavy traffic in two forest in northeast France, especially during an initial phase lasting from immediately after compaction to the first soil drought. During this initial period, [CO_2] was increased (respectively decreased for [O_2]) independently from soil climatic conditions due to the probable strong decrease in gas diffusivity throughout the 0-20 cm soil layer in the T-treatment. While considering the same volume of soil occupied by the gas phase, soil gas diffusivity in this soil layer increased following the first drought experimented at both sites, probably due to cracks formation. Therefore the decrease in impact of heavy traffic on [CO_2] and [O_2] at CA was probably not the result of soil regeneration towards undisturbed levels of soil biological activity and gas transfers, but more the consequence of the decreased hydro-structural stability of compacted soils. Even if the cracks formed in the T-treatment increased soil porosity and pore connectivity, the improvement in soil structure was not sufficient to regenerate ε_A towards undisturbed levels throughout the entire study period. Following to the high initial effect of heavy traffic on [CO_2] and [O_2], only a small impact remained. This slight effect of soil disturbance on [CO_2] and [O_2] was attributed to compensatory phenomenon between the decreased production of CO_2/ consumption of O_2 on the one hand and the decreased gas diffusion on the other hand. Therefore the impact of treatment (increase or decrease) strongly depended on soil climatic conditions of the depth and sampling time considered. Only standardization with respects to soil physical characteristics could give rise to a significant effect of treatment.

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Appendix 1: calibration curves between soil dielectric constant measured by the TDR probes and soil volumetric water content

The relationship between soil dielectric constant measured by the TDR probes and soil water content was calibrated on three soil cores per site (one per depth). The undisturbed soil cores (approximately 7000 cm^3) were measured for mass and dielectric constant while the top of the sample was allowed to evaporate at the laboratory (20°C) until soil mass remained constant. Gravimetric water contents were calculated on an oven-dried basis (105°C, 48 h) and were converted into volumetric water content using the bulk density of the soil core. Different calibration curves according to Heathman et al. (2003) were fitted on the data set and the best one was selected according to the distribution of the residuals, its coefficient of determination and residual standard error:

$$WC_v = A \times K_a^{0.5} + B \qquad (1)$$

Where WC_v is soil volumetric water content ($m^3\ m^{-3}$), K_a is the apparent dielectric constant, A and B are two parameters significantly different per site and soil layer (Table 5).

Table 5 Selected calibration curves to convert the dielectric constant measured by the TDR probes into soil water content

	A[a]	B	RSE	d.f.	R^2
AZ 15 cm	0.1255 (4×10^{-4})	-0.202 (0.001)	0.002	77	0.99
AZ 30 cm	0.1187 (4×10^{-4})	-0.197 (0.001)	0.003	71	0.99
AZ 60 cm	0.1245 (9×10^{-4})	-0.185 (0.003)	0.005	72	0.99
CA 15 cm	0.1084 (5×10^{-4})	-0.158 (0.002)	0.002	49	0.99
CA 30 cm	0.1170 (4×10^{-4})	-0.194 (0.002)	0.002	52	0.99
CA 60 cm	0.127 (0.001)	-0.206 (0.005)	0.004	55	0.99

[a] A and B are the coefficients of equation (1) with their standard errors in parentheses; RSE, residual standard error; d.f., degrees of freedom; R^2, coefficient of determination.

ANNEXE 1

Description sommaire des sols avant exploitation à Azerailles, AZ et à Clermont en Argonne, CA ; adapté d'après les observations d'A. Brêthes (non publié)

AZ

Humus de forme mésomull

A (0–10 cm) horizon organo-minéral, brun foncé (10 YR 3/2), limon faiblement argileux, structure grumeleuse

E (10–25 cm) horizon brun à brun ocre (10 YR 4/2 et 5/3), limon plus ou moins argileux, structure polyédrique mal développée

Bt (25–50 cm) horizon brun ocre (10 YR 5/4), limon argileux, structure polyédrique bien développée, hydromorphie variable

II Sg ou II Btg (50–80 cm) horizon brun ocre (7,5 YR 5/3), argile limoneuse, structure polyédrique à prismatique, hydromorphie variable mais plus forte que dans l'horizon supérieur

II C (80–100 cm) horizon brun ocre (7,5 YR 6/3) avec des plages gris-bleuté (5 GY 7/1) plus ou moins étendues, argile lourde, structure prismatique, hydromorphie variable.

CA

Humus de forme dysmull à moder

A (0–10 cm) horizon organo-minéral, brun gris foncé (10 YR 3-4/2), limon moyen, structure massive localement micro grumeleuse, transition ondulée

E (10–35 cm) horizon brun clair (10 YR 5/3), limon moyen, structure massive, localement quelques traces d'hydromorphie (plages décolorées, tâches ocres, concrétions ferro-manganiques)

Bt (35–45/60 cm) horizon brun (7,5 YR 5/3), limon argileux à argile limoneuse, structure polyédrique peu à bien développée, hydromorphie variable

II Btg (45/60–70/90 cm) horizon brun (7,5 YR 5/3), argile limoneuse à argile lourde, structure polyédrique à prismatique, hydromorphie variable mais plus forte que dans l'horizon supérieur

II C (>70/90 cm) horizon brun (7,5 YR 5/3), face des agrégats décolorées (7,5 YR 7/1) avec des plages ocres plus ou moins étendues, argile lourde, structure prismatique, hydromorphie variable.

Observations morphologiques des fosses prélevées en mai 2011 (4 ans après tassement à AZ et 3 ans après tassement à CA) pour les mesures de densité apparente

AZ témoin relatif (bande non perturbée de 10 × 50 m à côté de la bande circulée, photos : bloc 2).

0–5 cm : horizon organo-minéral, brun, très grumeleux et très friable (chaque agrégat est relativement résistant à la désagrégation mécanique mais peu de liants entre eux).

5–15 cm : horizon minéral brun, grumeleux mais les éléments structuraux sont plus fins que pour l'horizon 0-5cm.

15–25 cm : horizon minéral plus clair mais non dégradé par hydromorphie, légèrement plus argileux avec une porosité moins visible que l'horizon précédent et une structure polyédrique relativement fine.

25–50 cm : horizon minéral de couleur ocre avec des concrétions noires ferro-manganiques assez importantes, quelques tâches claires. Horizon beaucoup plus argileux que le précédent et plus frais.

AZ tassé centre (centre de la bande circulée de 30 × 50 m, photos : bloc 2).

L'horizon organo-minéral est envahi de racines de jonc et la terre entre les racines a une structure légèrement grumeleuse. De 10 à 40 cm de profondeur, on retrouve la même différenciation des horizons que dans le témoin, mis à part que la structure observée dans le témoin a disparu, que des tâches de matière organique et des traces d'hydromorphie (zones blanchies et concrétions ferro-manganiques) peuvent être observées localement.

5–10 cm
zone de sol dégradée par hydromorphie

15–30 cm
La structure est massive et la porosité n'est plus visible

CA témoin relatif (photos : bloc 3, profil de sol sec). Certaines fosses montrent des traces d'hydromorphie dès la surface, notamment dans le bloc 1 (plus plat). Cependant quand ces traces sont présentes dans le témoin relatif elles sont généralement limitées à une portion de fosse.

0–5cm : horizon organo-minéral, structure polyédrique fine (pas de porosité visible sur le profil mais la structure massive s'effrite assez facilement en petits agrégats polyédriques peu stables).

5–20cm : limon jaune clair sain, macrostructure massive fragile, horizon relativement homogène, pas ou peu de traces d'hydromorphie.

20–35cm : horizon plus argileux et plus jaune que le précédent.

35–55cm : apparition de traces d'hydromorphie.

CA tassé centre : seules les couches de sol superficielles (0-20 cm) sont montrées ici, les couches profondes étant globalement identique au témoin (variations selon la profondeur d'apparition de l'altérite argileuse de gaize). Les profils de sol étant secs, les couleurs sont atténuées. Les photos ont été choisies de manière à illustrer les principales différences morphologiques entre sol tassé et sol témoin.

Bloc 2, 0–20 cm

la coloration du profil de sol est plus hétérogène. La limite entre l'horizon organo-minéral et l'horizon minéral sous-jacent est très irrégulière, l'horizon organo-minéral pouvant, par endroits, descendre assez profondément dans l'horizon minéral.

Bloc 3, 0–10 cm

la structure est très massive et les traces d'hydromorphie sont plus intenses et fréquentes (zones blanchies et concrétions ferro-manganiques)

Bloc 1, 0–10 cm

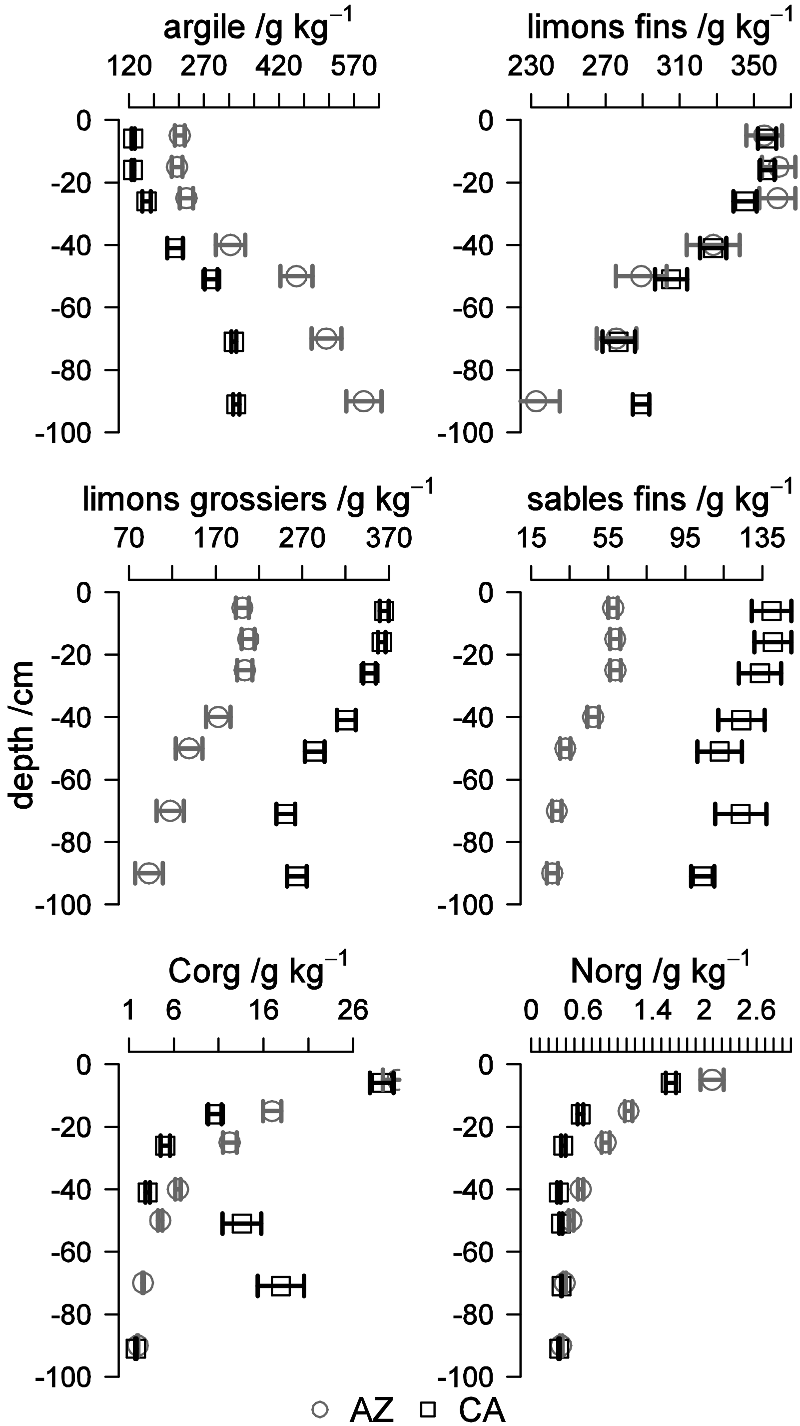
Présence de zones réduites (gris-bleu)

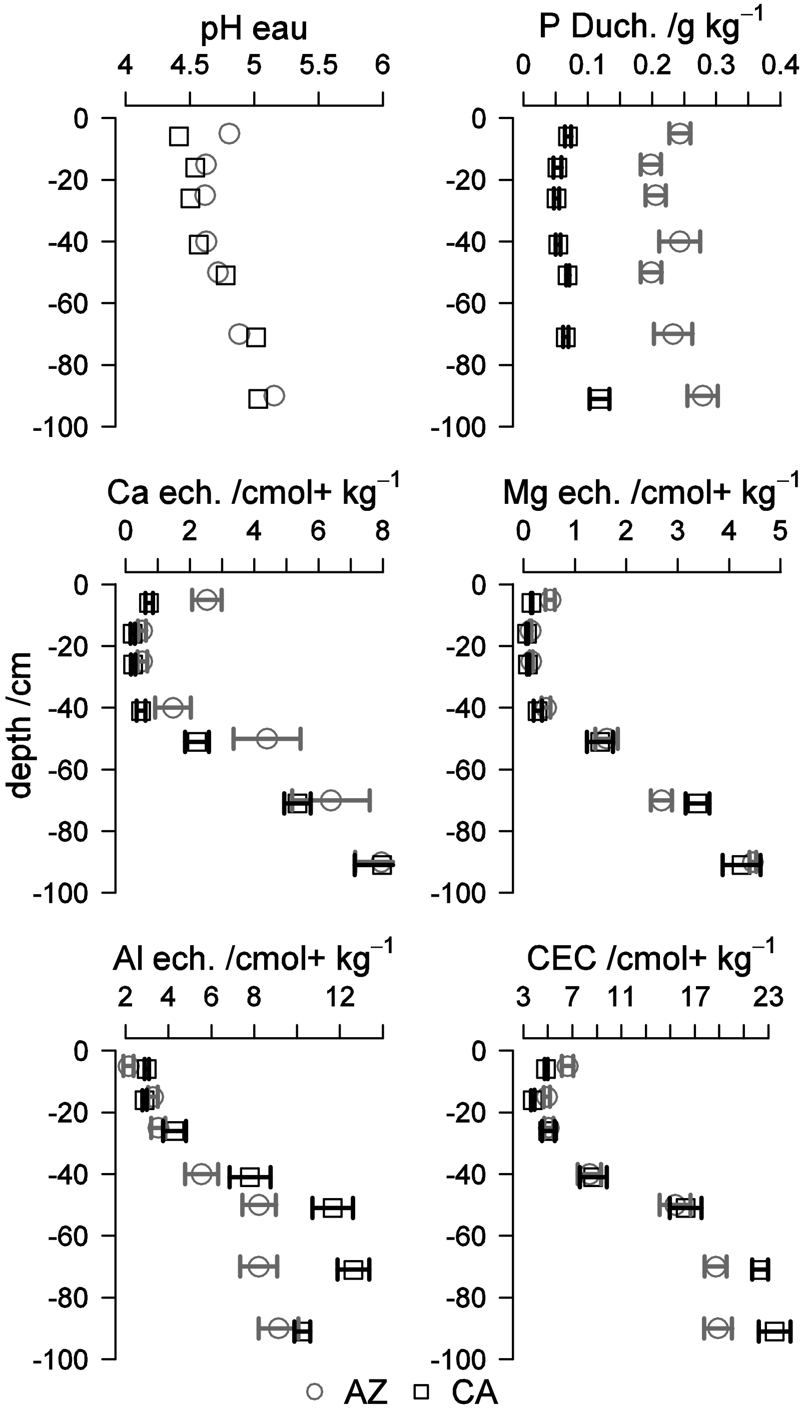
Bloc 1, 10–20 cm

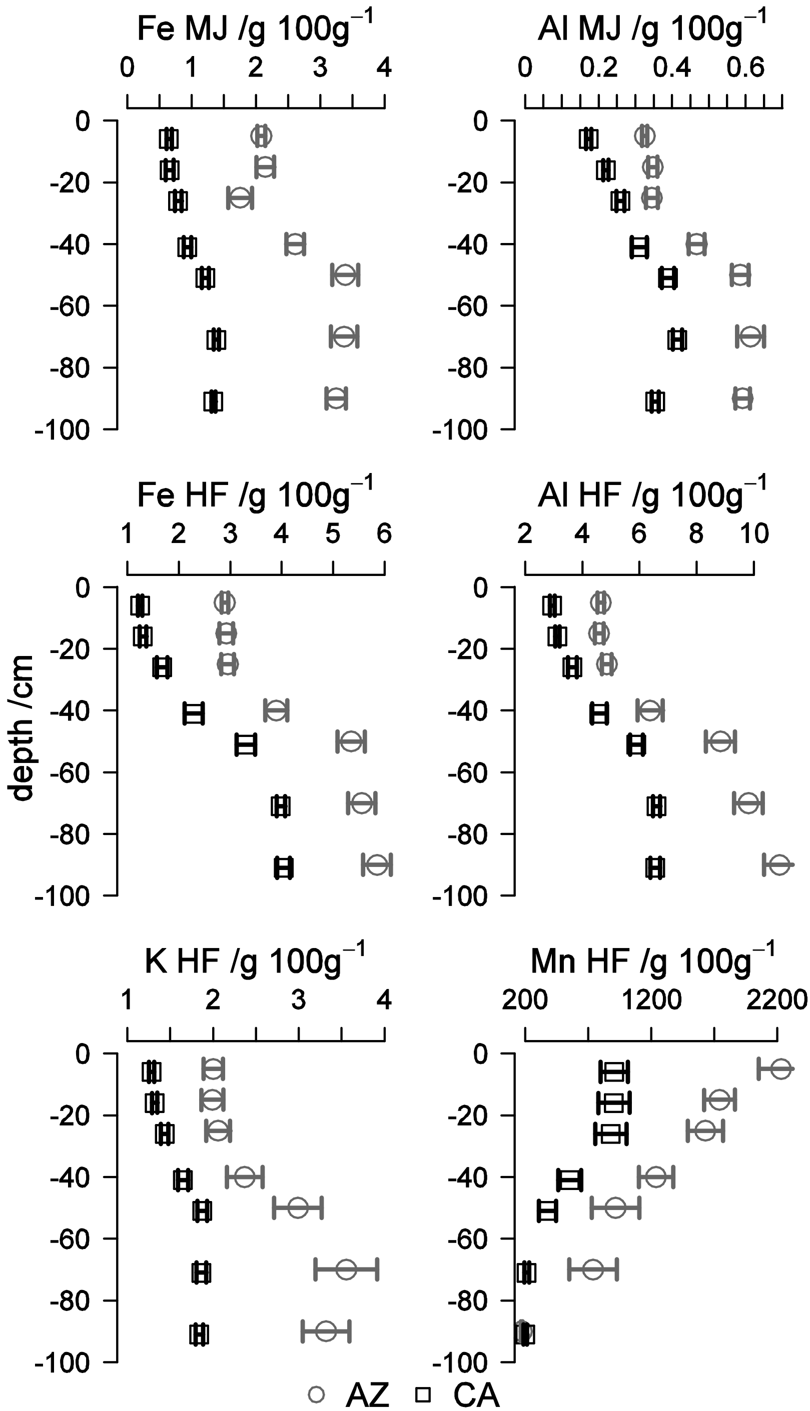
Dégradation hydromorphe très nette

ANNEXE 2

Analyses chimiques des sols réalisées avant exploitation des sites d'Azerailles (AZ) et de Clermont en Argonne (CA). Les graphiques représentent la moyenne et l'erreur standard par profondeur et par site pour chaque élément. Mesures réalisées sur 18 (AZ) et 13 (CA) fosses réparties sur l'ensemble de la surface des deux sites. Pour ne pas gêner les prélèvements ultérieurs, les fosses ont été positionnées aux extrémités des futurs placeaux (voir les plans en fin d'annexe 1). Les analyses ont été réalisées par le laboratoire d'Arras. Les éléments échangeables et la CEC (Capacité d'échange cationique) ont été extraits au chlorure de cobaltihexammine, le fer et aluminium libres (Fe MJ et Al MJ) ont été extraits selon la méthode de Mehra & Jackson et les éléments totaux (Fe HF, Al HF, Mn HF et K HF) ont été mis en solution par réaction avec des acides fluorhydrique et perchlorique (pour les méthodes d'analyse : http://www5.lille.inra.fr/las).


argile /g kg^{-1}
120 270 420 570
limons fins /g kg^{-1}
230 270 310 350
limons grossiers /g kg^{-1}
70 170 270 370
sables fins /g kg^{-1}
15 55 95 135
Corg /g kg^{-1}
1 6 16 26
Norg /g kg^{-1}
0 0.6 1.4 2 2.6
depth /cm
0 -20 -40 -60 -80 -100
AZ CA

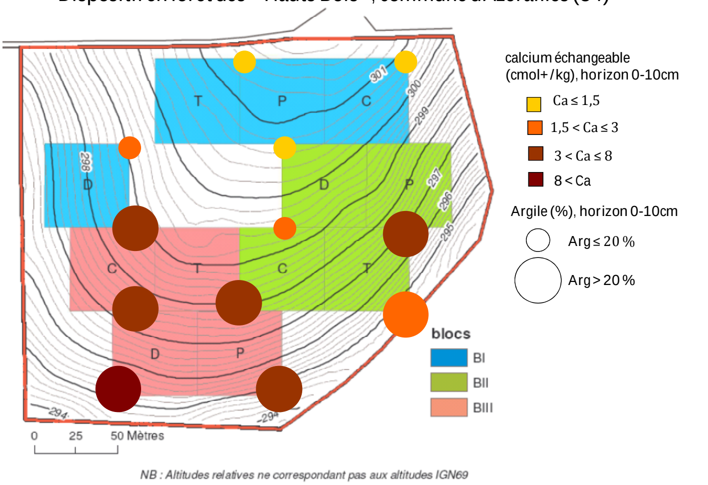

pH eau
4
4.5
5
5.5
6
P Duch. /g kg^{-1}
0
0.1
0.2
0.3
0.4
Ca ech. /cmol+ kg^{-1}
0
2
4
6
8
Mg ech. /cmol+ kg^{-1}
0
1
2
3
4
5
Al ech. /cmol+ kg^{-1}
2
4
6
8
12
CEC /cmol+ kg^{-1}
3
7
11
17
23
depth /cm
0
-20
-40
-60
-80
-100
AZ
CA



Fe MJ /g 100g^{-1}
Al MJ /g 100g^{-1}
Fe HF /g 100g^{-1}
Al HF /g 100g^{-1}
K HF /g 100g^{-1}
Mn HF /g 100g^{-1}
depth /cm
AZ
CA

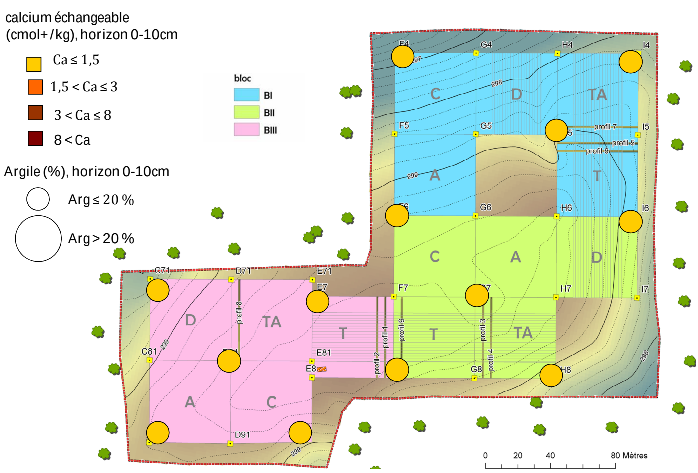
Variabilité spatiale de la teneur en argile et en calcium échangeable de l'horizon de surface (0-10 cm) des deux sites. Les numéros de bloc sont indiqués en chiffres romains (BI, BII et BIII). Au sein de chaque placeau (50×50 m), les lettres correspondent aux différents traitements ; C (control) pour le traitement témoin, T (trafficked) pour le traitement tassé, D pour le traitement tassé et décompacté par cover crop (non traité dans cette thèse), P pour le traitement tassé et décompacté par potet au profit du plant (non traité dans cette thèse), A (amended) pour le traitement amendé, TA (trafficked amended) pour le traitement tassé puis amendé.

Dispositif en forêt des « Hauts Bois », commune d'Azerailles (54)



Plan : C.Pasquier, 2007 (INRA Orléans)

Dispositif en forêt de « Grand Pays », commune de Clermont en Argonne (55)



Plan : C.Pasquier, 2008 (INRA Orléans)

ANNEXE 3

Préparation et traitements appliqués aux sites expérimentaux d'Azerailles (AZ) et de Clermont en Argonne (CA):


coupe rase et débardage par câble mât (AZ et CA)


récolte des rémanents à la main ou avec le cheval de fer (AZ et CA)


circulation en plein du porteur forestier (AZ et CA)


vue après tassement du sol (AZ)


vue après tassement du sol (CA)


Amendement calco-magnésien (CA)

Dispositif de suivi :

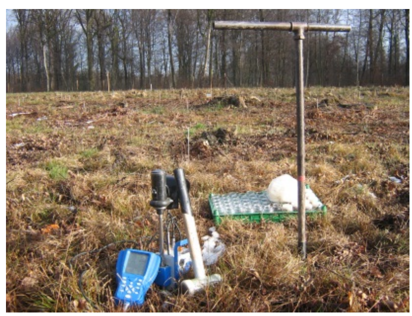
Relevé en continu de la température et de l'humidité du sol

Prélèvements mensuels de l'atmosphère du sol

Prélèvement annuel de densité apparente (CA, mai 2011)

Prélèvement annuel de densité apparente (CA, mai 2009)

Mesure de la courbe de retrait (laboratoire hépia, Genève, Suisse ; février 2011)



Mesure de la résistance à la pénétration des sols (PANDA®, novembre 2008)

Changes and recovery in soil physical and chemical characteristics of two forest soils following traffic by a full-loaded forwarder

Abstract

Soil compaction belongs to the major treats to soil quality with no exceptions of forest ecosystems where the frequency and intensity of loads application increase since several decades. The mechanisms and the duration of soil quality recovery following heavy traffic in forests remain poorly documented and their study requires multidisciplinary approaches.

The aim of this work was to evaluate the impact of forwarder traffic on the potential constraints to roots growth (aeration, water content, and penetrability) growing in two forest soils sensitive to compaction, and the evolution of these consequences in the short-term. This work is based on two experimental sites, set up in the northeast of France, with soils displaying similar morphologies (50 cm thick silt loam layer laying on a clayey layer) and having being loaded with the same forwarder under similar (wet) soil conditions. Soil climate (temperature and moisture), soil air composition, and soil bulk density and resistance to penetration were investigated continuously, monthly and yearly, respectively. Soil climate and air composition monitoring showed a strong initial decrease in aerobic conditions lasting one to one and a half year. The strong initial impact on soil aeration decreased concurrently with the first soil drought experimented at both sites, probably because of soil cracks formation in the disturbed soil. Yet heavy traffic still affected significantly soil air composition 3 to 4 years after compaction at both sites. To monitor changes in soil physical parameters, we had to standardize measures with regards to soil climatic conditions at the time of sampling. Three to four years after soil compaction, the difference in soil physical properties between treatments was still significant. However, changes in the impact of the forwarder traffic on soil physical characteristics could be stated in the surface layer (0–10 cm) of both sites. This beginning of soil restoration results at one of both sites in a difference between treatments that is no longer significant when soils are wet but that is still significant when the soils are dry. At the second site, the difference is still significant whatever soil moisture conditions but it has decreased since the start of the experiment. Consequently, this beginning of soil structure recovery is not accompanied by a disappearance of the hardsetting behaviour (decrease in hydrostructural stability) of the compacted soil at one site.

In this study changes in the consequences of the forwarder traffic were stated in the surface soil layers of both sites, these changes may be due to physical processes (wetting – drying, freezing – thawing). Nevertheless, the impact remains strong on roots growth (high resistance to penetration when dry, poor gas transfer when wet) and on stand resilience to external stresses (drought, storm).

Key words soil compaction, forest soil, natural recovery dynamic, soil atmosphere, bulk density, resistance to penetration.

Modifications et restauration de propriétés physiques et chimiques de deux sols forestiers soumis au passage d'un engin d'exploitation

Résumé

Les risques de dégradation physique des sols forestiers sous l'effet de contraintes mécaniques externes liées à la mécanisation des opérations forestières, augmentent considérablement. Les mécanismes et le temps nécessaires à la restauration non assistée de la qualité des sols forestiers tassés restent encore peu étudiés, et leur identification nécessite de coupler les approches physiques, chimiques et biologiques.

L'objectif de ce travail était d'étudier l'impact de la circulation d'un porteur forestier sur les conditions de l'enracinement (aération, régime hydrique et pénétrabilité) ainsi que son évolution à court terme. Ce travail s'appuie sur l'observation de deux sites expérimentaux mis en place dans le Nord Est de la France, concernant des sols de morphologie similaire (couche limono-argileuse de 50 cm d'épaisseur reposant sur un substrat argileux) et ayant subi des contraintes identiques. Des paramètres physiques (température et humidité du sol, densité apparente et résistance à la pénétration) et chimiques (composition de l'atmosphère du sol) ont été suivis pendant trois à quatre ans, à des fréquences allant d'un pas de temps quotidien à annuel.

Le suivi du climat du sol et de la composition de son atmosphère a mis en évidence une diminution forte des conditions aérobies pendant un à un an et demi après le passage du porteur. Cet effet initial sur l'aération du sol a diminué subitement dès l'apparition de la première période de sécheresse édaphique, probablement grâce à la formation de fissures dans l'horizon de surface du sol tassé. Cependant un effet significatif du traitement sur la composition de l'atmosphère du sol pouvait toujours être observé trois à quatre ans après tassement. Pour suivre l'évolution des propriétés physiques du sol après circulation du porteur, il a été nécessaire d'opérer une normalisation par rapport à l'humidité au moment du prélèvement. Trois ou quatre ans après la circulation du porteur, une différence toujours significative existe entre les propriétés physiques des sols témoins et celles des sols tassés. Cependant une évolution de l'impact du porteur peut être mise en évidence dans la couche de surface (0–10 cm) des deux sites. Ce début de restauration se traduit, sur un des deux sites, par une différence entre traitement qui n'est plus significative quand les sols sont humides mais qui l'est encore quand les sols sont secs. Sur le deuxième site, cette différence a diminué quelle que ce soit l'humidité du sol. Ainsi, le début de régénération de la structure du sol perturbé ne s'accompagne pas d'une disparition de son comportement de prise en masse lors de son desséchement sur un des deux sites.

Ce travail a permis de mettre en évidence une évolution des conséquences du porteur en surface du sol tassé qui serait liée à des processus physiques (gonflement – retrait, gel – dégel). Cependant, l'impact sur les conditions de l'enracinement (forte résistance à la pénétration quand les sols sont secs, faible aération quand ils sont humides) reste élevé de même que sur la résilience à long terme du peuplement.

Mots clés tassement sol, sol forestier, restauration naturelle, atmosphère du sol, densité apparente, résistance à la pénétration