THE POTENTIAL OF CONTROLLED TRAFFIC FARMING TO MITIGATE GREENHOUSE GAS EMISSIONS AND ENHANCE CARBON SEQUESTRATION IN ARABLE LAND: A CRITICAL REVIEW

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ABSTRACT. *The drive toward adoption of conservation agriculture to reduce costs and increase production sustainably causes concern due to the potentially negative effects of increased soil compaction. Soil compaction reduces aeration, water infiltration, and saturated hydraulic conductivity and increases the risk of waterlogging. Controlled traffic farming (CTF) is a system in which: (1) all machinery has the same or modular working and track width so that field traffic can be confined to the least possible area of permanent traffic lanes, (2) all machinery is capable of precise guidance along those permanent traffic lanes, and (3) the layout of the permanent traffic lanes is designed to optimize surface drainage and logistics. Without CTF, varying equipment operating and track widths translate into random traffic patterns, which can cover up to 85% of the cultivated field area each time a crop is produced. Nitrous oxide (N_2O) is the greatest contributor to agriculture's greenhouse gas (GHG) emissions from cropping, and research suggests that its production increases significantly under conditions of high (>60%) water-filled porosity when nitrate (mainly from fertilizer N) and carbon (usually from crop residues) are available. Self-amelioration of soils affected by compaction occurs slowly from the surface downward; however, the rate of amelioration decreases with increase in depth. Consequently, all soils in non-CTF systems in mechanized agriculture are prone to some degree of compaction, which compromises water infiltration, increases the frequency and duration of waterlogged conditions, reduces gaseous exchange between soil and the atmosphere, inhibits root penetration and exploitation of nutrients and water in the subsoil, and enhances N_2O emissions. Adoption of CTF increases soil porosity in the range of 5% to 70%, water infiltration by a factor of 4, and saturated hydraulic conductivity by a factor of 2. The greater cropping opportunity and enhanced crop growth for given fertilizer and rainfall inputs offered by CTF, coupled with no-tillage, provide potential for enhanced soil carbon sequestration. Reduced need and intensity of tillage, where compaction is avoided, also helps protect soil organic matter in stable aggregates, which may otherwise be exposed and oxidized. There is both circumstantial and direct evidence to suggest that improved soil structural conditions and aeration offered by CTF can reduce N_2O emissions by 20% to 50% compared with non-CTF. It is not compaction per se that increases the risk of N_2O emissions but rather the increased risk of waterlogging and increase in water-filled pore space. There may be an elevated risk of GHG emissions from the relatively small area of permanent traffic lanes (typically <20% of total cultivated area) if these are not managed appropriately. Quantification of the benefits of compaction avoidance in terms of GHG emissions may be possible through the use of well-developed models.*

Keywords. *Carbon dioxide (CO_2), Carbon sequestration, Fertilizer use efficiency, Nitrous oxide (N_2O), No-tillage, Precision agriculture, Rainfall use efficiency, Random traffic, Soil compaction.*

Greenhouse gases (GHG) are those that absorb light in the infrared region and reduce transparency to thermal radiation from the Earth's surface (Snyder et al., 2007). Considerable research has been devoted to assessing the effects of different land management practices on the net emissions of trace gases from soils that have the potential to impact global warming. Agriculture is regarded as a sector emitting three GHG, namely, carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), but with the potential to act as a sink for CO_2 (Smith, 2004). The carbon (C) sink capacity of soils is equivalent to the C deficit generated through historic land-use and management practices (Lal, 2007, 2013a). As much as 70% of the total GHG emissions from cropping are associated with nitrogen (N) fertilizer (IPCC, 2007), which is a combination of CO_2 and N_2O from its manufacture and

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distribution, and direct and indirect emissions of N_2O from its use on crops. CO_2 is considered the reference gas and is assigned a global warming potential (GWP) of 1, while N_2O and CH_4 have potentials of 296 and 23, respectively, on an arbitrary 100-year time scale (IPCC, 2007). Nitrogen oxides derived from photochemical reactions of N_2O are the largest contributors to ozone (O_3) destruction in the stratosphere (Ravishankara et al., 2009), although O_3 depletion by N_2O is lessened by CO_2 thermal effects (by about 20%) in the middle stratosphere (Portmann et al., 2012, provide this value based on the IPCC A1B/WMO A1 scenario over the time period 1900-2100).

Globally, approximately 65% of all N_2O emissions emitted from the biosphere arise from soils, including emissions stimulated from fertilized and manure-treated agricultural soils (Mosier, 1994; IPCC, 2001; Rees et al., 2013). In 2006, N_2O emissions linked to the use of synthetic N fertilizers were estimated at 605 Tg CO_2e, which represented about 30% of total N_2O emissions from agricultural soils (IFA, 2009). Total emissions from fertilizer use represent approximately 1.5% of global GHG emissions (IFA, 2009). The amount of N_2O in the atmosphere has been estimated to increase at a rate of 0.2% to 0.3% per year (Granli and Bøckman, 1994; IPCC, 2001; Snyder and Fixen, 2012), and it is generally accepted that N_2O contributes to about 8% of total GHG emissions (Rees and Ball, 2010; Syakila and Kroeze, 2011). While some land management practices have the potential to increase, reduce, or mitigate GHG emissions (e.g., Di and Cameron, 2002; Vergé et al., 2007; Eagle and Olander, 2012; Rees et al., 2013; Scheer et al., 2013), the relationships and net outcomes are often complex and can lead to the opposite effect of that anticipated. Most studies (e.g., FAO, 2014) have tended to apply a broad-brush approach, categorizing land management into forestry, livestock, or cropping, for example. Although a few studies (e.g., Adviento-Borbe et al., 2007) have attempted to make direct links between emissions and specific management practices, including fertilizer and manure management (e.g., Smith et al., 1997; Smith et al., 2000; Snyder et al., 2009; Millar et al., 2010; Smith et al., 2012; Norton, 2014), an increasing number now consider the effects of tillage system (e.g., Skiba et al., 2002; Rochette, 2008; Soane et al., 2012) and the direct effect of machinery-induced soil compaction (e.g., Ball et al., 1999; Sitaula et al., 2000; Ruser et al., 2006; Beare et al., 2009; Vermeulen et al., 2007; Tullberg, 2010; Gasso et al., 2013). There is also recognition of compaction as a constituent of soil physical degradation that accelerates erosion processes (Reed, 1983; Tullberg et al., 2001; Rickson, 2014) and loss of soil organic C (Lal, 2004a). There appears to be a positive relationship between N_2O emissions and the combined effects of soil bulk density and clay content, but bulk density has a relatively larger influence on regulating fluxes than clay content (Ball, 2013). Ball (2013) indicated that such a relationship, i.e., $\log(N_2O \text{ emission}) = -1.97 + (2.57 \times \rho_b) + (0.015 \times \text{clay, \%})$, did not hold true for certain periods of the year when emissions were lower and likely to be affected by the previous crop in the rotation. Other research also suggests that the influence of compaction on N_2O emissions appears to be higher in clay soils compared with sandy soils (Mosquera-Losada et al., 2007; Snyder et al., 2009). The work reported in this article focuses on the influence of traffic-induced soil compaction on GHG emissions. Readers are referred to several reviews dealing with the effects of soil compaction by grazing livestock on N_2O emissions from grasslands (e.g., Oenema et al., 1997; Anger et al., 2003; Luo et al., 2010a).

CONTROLLED TRAFFIC FARMING

The Australian Controlled Traffic Farming Association, Inc. (http://actfa.net/) defines controlled traffic farming (CTF) as a system in which:

1. All machinery has the same or modular working and track width so that field traffic can be confined to the least possible area of permanent traffic lanes,
2. All machinery is capable of precise guidance along those permanent traffic lanes, and
3. The layout of the permanent traffic lanes is designed to optimize surface drainage and logistics.

Without CTF, varying equipment operating and track widths translate into random traffic patterns, which can cover up to 85% of the cultivated field area each time a crop is produced (Kroulík et al., 2009; Tullberg, 2010). Alternative CTF systems to the single (common) track width have also been developed, and many of these are more readily adoptable within European farming systems (Chamen, 2006). For those situations, a tier approach is established that encourages growers to progressively reduce the area of the field subjected to traffic through improvements in the design of their CTF system. For example, the tier system developed by CTF Europe Ltd. (http://www.controlledtrafficfarming.com/) includes the following (as % of tracked area): 30% to 40% (tier 1), 20% to 30% (tier 2), 10% to 20% (tier 3), and ≤10% (tier 4). Tier 4 may only be achievable with the use of gantry systems (Chamen et al., 1992a). In non-CTF systems, the area subjected to traffic often exceeds 40% of the cultivated field area. Seasonal controlled traffic systems (sCTF) are designed to confine most field operations (usually with the exception of harvesting) to (semi) permanent traffic lanes. These systems represent a technical solution for the vegetable industry, for example, where incompatibilities between harvesting equipment are common (McPhee et al., 2015). In addition to being practical, CTF has fundamental advantages in maintaining all aspects of "good" soil structure with lower inputs of energy and time compared with conventional traffic systems (Chamen, 2011. In the context of this study, we define "conventional traffic systems" as those that exhibit "random" traffic patterns with agricultural vehicles). Self-amelioration of soils affected by compaction occurs slowly from the surface downward; however, the rate of amelioration decreases with increase in soil depth (Dexter, 1991; Chinn and Pillai, 2008; McHugh et al., 2009). Traffic impacts are persistent (Alakukku, 1996; Radford et al., 2007), meaning that all soils in non-CTF systems are affected by some degree of compaction at depths greater than 100 mm (Ansorge and Godwin, 2007;

Antille et al., 2013a), which reduces water infiltration and hydraulic conductivity, increases the risk of waterlogging, and creates favorable conditions for enhanced N_2O emissions.

CTF in Australia represents a profitable technological innovation for arable land-use (Kingwell and Fuchsbichler, 2011), and similar observations are made by Chamen et al. (2015) for the United Kingdom. Despite these benefits, CTF has been adopted by less than 40% of Australian grain farms, and global adoption of this technology appears to be small (Tullberg et al., 2007; Chamen, 2014). While CTF is an engineering solution to some of the unwanted effects of soil compaction (Smith et al., 2013), this is a limited objective. CTF transforms a problem of random traffic-induced soil compaction into an advantage of improved trafficability and timeliness, which have additional agronomic and environmental benefits. Compacted permanent traffic lanes also provide a safe runoff management system by limiting the concentration of runoff (Li et al., 2007). CTF eliminates an important motivation for tillage, and traffic-induced variability of soil-crop conditions is a significant hindrance to management of no-tillage (NT) cropping. In modern mechanized crop production, controlled traffic might reasonably be seen as a prerequisite for eliminating tillage, and NT is an essential component of CTF for Australian grain farming, together with increased cropping intensity (Yule and Radford, 2003). If compaction effects (direct and indirect) are a serious issue for cropping, then CTF is clearly a substantial enhancement of conservation agriculture (CA) (FAO, 2001). Thus, one of the main benefits associated with the adoption of CTF is increased productivity; however, this article focuses on the sustainability aspects in management of arable land.

It is envisaged that establishment of such traffic management systems coupled with CA and with the aid of precision agriculture can significantly increase both the productivity and sustainability of arable farming. Progressive deterioration of the soil resource has resulted in a yield plateau despite continuing genetic and varietal improvements (Bolton and Crute, 2011). Nonetheless, there is a need to increase global food production to meet a projected 50% increase in food demand by 2030 (Bruinsma, 2009). It is also known that increased soil C sequestration improves soil quality and activates land recuperative processes, which in turn advances food security (Lal, 2006, 2015). Meeting global food demand is possible mainly through increased crop yields, and to a lesser extent through expansion of agricultural area (West et al., 2010). However, concerns have been raised over the potential increase in global GHG emissions (~25% relative to 2005 levels) associated with this required increase in the production of food crops (Schulte et al., 2011). The bulk of this increase in GHG emissions would occur in developing countries, including Latin America (USEPA, 2006). The extent of increase in emissions will depend on how the required increment in production is achieved, whether through higher yields or expansion of agriculture into new areas, or both (Schulte et al., 2011). Increments in GHG emissions associated with improved crop yields have been historically lower than those corresponding to changes in land use (Burney et al., 2010). In this scenario, CTF has an important role to play by:

1. Promoting increased crop yields with relatively lower GHG emissions compared with conventional traffic systems (Li et al., 2007; Vermeulen and Mosquera, 2009; Tullberg et al., 2011; Smith et al., 2014),
2. Potentially reducing the need for expansion of agriculture into historically non-agricultural land, and
3. Increasing efficiency and productivity in regions such as South America where CTF, coupled with well-established soil conservation practices (e.g., NT in Argentina and Brazil; Derpsch et al., 2010), could have a synergistic effect on productivity and sustainability. This is an important consideration because of the relatively high GHG emissions observed in developing countries (Vergé et al., 2007; Ogle et al., 2013) and thus potential for emissions reductions.

The first outcome outlined above is possible owing to a combination of overall improvement in soil conditions (McHugh et al., 2009; McPhee et al., 2015) leading to enhanced fertilizer use efficiency (FUE) and nutrient uptake (Galambošová et al., 2014). Improving FUE through CTF is an important agronomic and environmental consideration since much of the effort in reducing N_2O emissions is centered on reducing N application rates. While acknowledging the relationship that exists between fertilizer N rate and N_2O emissions (Millar et al., 2010), such reductions could compromise meeting the future demand for food crops, restrict CO_2 capture in crop biomass, and affect regeneration and maintenance of soil organic matter (Ladha et al., 2011; Snyder and Fixen, 2012). This is one of the reasons why the assessment of GHG emissions must be based on the unit of output (van Groenigen et al., 2010). The second outcome outlined above may be justified provided that the first outcome is true and economic or other pressures do not stimulate that shift in land-use.

The potential benefits of coupling CTF and permanent NT is supported by studies conducted in temperate as well as subtropical environments (e.g., Wang et al., 2011; Mangalassery et al., 2014), which showed a significantly higher (up to 30%) net GWP in conventional tillage compared with NT. It is also envisaged that the trend observed in the past few decades toward the use and development of larger, more powerful, agricultural machinery will continue (Kutzbach, 2000). The main drawback is the associated increase in machinery weight, which increases the risk of soil damage due to compaction (Raper, 2005). This issue is particularly important for cropping situations, such as cotton (*Gossypium hirsutum* L.) based systems in Australia, which are relatively intensive in terms of N fertilizer and water inputs (Rochester, 2011) and where recent developments in harvesting technology resulted in: (1) significant increases in axle loads, and (2) "ideal" CTF systems being difficult to implement due to incompatibilities between tire configurations, track width of pickers, and crop row spacing (Braunack and Johnston, 2014). These conditions suggest increased potential for GHG emissions from those cropping systems and may also be observed in sugarcane (*Saccharum officinarum* L.) and sugarbeet (*Beta vulgaris* L.), which are intensively managed and subjected to similar mechanization constraints (Arvidsson, 2001; Tzilivakis et al., 2005a, 2005b; Braunack and McGarry, 2006; Renouf et al., 2008; Denmead et al., 2010).

Aim

The aim of the work reported in this article is to elicit sufficient fundamental information about the mechanisms involved in carbon (C) and nitrogen (N) dynamics to be able to envisage the likely impact of confining all traffic-induced soil compaction to the least possible area of permanent traffic lanes, a system known as controlled traffic farming (CTF). It is hypothesized that NT coupled with controlled traffic and with the aid of precision agriculture technologies have the potential to increase C sequestration and mitigate N_2O emissions in arable land. Adoption of such traffic and soil management practices may help to overcome the challenge of increasing SOC pools (Lal, 2007), not only in temperate soils but also in subtropical and tropical farming systems. The information compiled in this article should help to stimulate a shift toward increased uptake of CTF.

Soil Carbon and Carbon Dioxide

Carbon (C) is the underlying source of CO_2. However, the relative quantities of its gaseous and physical forms are constantly changing. If mitigation of CO_2 loss is the requirement, the role of agriculture must be to maximize mechanisms that sequester C as soil organic matter (SOM) and minimize mechanisms that oxidize it into the atmosphere. Most C is lost from soil through the respiration of organisms that break it down. When oxygen (O_2) is present, this is done by aerobic organisms, and their respiration releases CO_2. However, where O_2 is in short supply, anaerobic organisms are involved, and their respiration releases methane (CH_4) (Serrano-Silva et al., 2014). Respiration in soil is enhanced when water-stable aggregates (WSA) are broken open to expose new SOM to the microbes, particularly in warm and moist conditions (Six et al., 2000a). The soil C pool has been characterized as follows (Smith, 2004; Smith and Conen, 2004; Smith and Falloon, 2005):

1. Soils contain 1550 petagrams (Pg) of organic C worldwide to 1 m depth, which is about three times the amount held in vegetation (560 Pg) and twice the amount held in the atmosphere (800 Pg). Stockmann et al. (2013) suggest that soils contain approximately 2350 Pg of organic C globally. Soils are also estimated to be the largest biospheric source of C lost to the atmosphere in Europe and elsewhere each year. Soils have lost between 40 and 90 Pg of C through cultivation and disturbance in the past 160 years (Lal, 2004b).
2. Estimates of the potential for additional C sequestration in soils vary widely. However, the most recent estimates suggest a biological potential of 0.9 ±0.3 Pg of C per year. This potential includes a wide range of options, including conversion to grassland and woodland. Realistically, only 20% of this potential is likely to be achievable.
3. The potential for C accumulation under NT agriculture is around 350 kg ha^{-1} of C per year (West and Post, 2002). The annual fluxes of CO_2 from the atmosphere to land and vice versa (respiration and fire) are each of the order of 60 Pg of C per year. Sequestration of C in SOM has a role to play; however, the accumulation is finite. Considering different C emission trajectories, the worst of which predicts a four-fold increase compared with the present, C sequestration in soil could only contribute between 2% and 5% toward reducing the C emission gap.

Several studies (Smith, 2004; Smith and Conen, 2004; Smith and Falloon, 2005) indicate that all measures will play an important role if the drastic reductions in GHG emissions needed are going to be achieved. Specifically for Australia, the potential for C sequestration in arable land appears to be technically and economically limited (Lam et al., 2013). Practices such as reduced tillage or NT, conversion to pasture, and residue retention would only increase C stock in the range of 7% to 13% in the topsoil (depth range: 0 to 100 mm) (Lam et al., 2013). Based on Luo et al. (2010b) and Sanderman and Baldock (2010), Lam et al. (2013) argue that such a C accumulation in the upper layer of soil may not be sustained in the long term due to its vulnerability to environmental and management strains. There is a need to determine whether specific practices can sustain soil C stock increases in the longer term in order to motivate land managers to adopt improved techniques that may enable permanent (>100 years) storage of C in soil (Lam et al., 2013). The requirement for improved management practices indicated in the studies reviewed (Smith, 2004; Smith and Conen, 2004; Smith and Falloon, 2005) may also bring about other benefits in addition to climate change mitigation potential. For example, Lal (2006) estimated that an additional 24 to 40 million Mg per year of food grains may be produced if SOC pools in soils of developing countries are increased at a rate of 1 Mg ha^{-1}. Komatsuzaki and Ohta (2007) concluded that increasing SOC is not controversial; it improves soil and water quality and overall fertility as well as biological cycles (Lal, 2004a), but care must be exercised to avoid greater emissions of non-CO_2 greenhouse gases.

Field Practices that Enhance Carbon Sequestration

Although measuring emissions provides a dynamic of the C cycle for a particular period in time, the actual accumulation of C in soil, as a result of specific management practices or cropping, is of more relevance. Angers et al. (1997) observed that C storage (depth range: 0 to 600 mm) was unaffected by NT, chisel plowing, or moldboard plowing where these treatments did not affect crop production. Blanco-Canqui and Lal (2007), also measuring to 600 mm depth in the profile, highlighted the importance of assessing the dynamics of sequestration to sufficient depth to ensure that all changes in C are being monitored. In their study, NT was compared with plow tillage for 13 major land resource areas in the eastern U.S. Accounting for differences in bulk density, they concluded that over this depth profile, NT was no better than plow tillage in sequestering SOC, and this was in accord with 14 out of 16 other studies (e.g., Franzluebbers et al., 2012) assessing tilled and NT systems. This is at odds with their earlier publication (Blanco-Canqui and Lal, 2004) in which they state that "any cultivation practice that reduces the disruption of aggregates will enhance SOC sequestration." This is addressed to some extent by Blanco-Canqui and Lal (2008), who state that "SOC under NT may be more stable with less turnover time and less seasonal changes than that under plowing." However, it still does not explain the absence of reported differences between NT and plow tillage. Bronick and Lal (2005) also indicate that "the effectiveness of SOC in forming stable aggregates is related to its rate of decomposition" and that "practices that minimize soil disruption enhance aggregation and structural development." The likelihood of containment of SOC in more stable aggregates is reinforced by Denef et al. (2004), who observed that more than 90% of the extra SOC to 200 mm depth under NT compared with a moldboard plow-based system was contained within micro-aggregates. Therefore, it is apparent that it is the intensity and type of tillage, and the conditions under which it is employed, that are critical in terms of disrupting aggregates and exposing previously protected SOC. Following tillage, CO_2 and N_2O emissions from soil are influenced by aggregate size distribution and are significantly higher from large macro-aggregates than other fractions (Bandyopadhyay and Lal, 2014). While the study by Bandyopadhyay and Lal (2014) considers CA, the combined role of CTF and NT in potentially assisting to achieve that goal, i.e., sustained increase in soil C sequestration, is not discussed.

A 28-year study of SOC sequestration in West Lafayette, Indiana, compared plowing to 250 mm depth with NT within continuous maize and maize-soybean rotations (Gál et al., 2007). NT accumulated 23 Mg ha^{-1} more SOC than plow-based tillage (depth range: 0 to 300 mm) but only 10 Mg ha^{-1} when the soil was sampled to a depth of 1 m. Taking account of differences in bulk density, this was further reduced to 8 Mg ha^{-1} on an equivalent mass basis. Unfortunately, although yield differences between the two rotations and tillage systems are quoted, the total biomass yields from the tillage contrasts over the 28 years of the study are not reported. There is an inference that the NT system produced less, but no absolute figure is given. These data would have been useful in either reinforcing the average 0.29 Mg ha^{-1} per year gain in SOC on the NT plots or detracting from it. Patiño-Zúñiga et al. (2009) investigated different tillage and residue management practices on soil characteristics and observed that non-tilled beds, with full retention compared with full removal of residues, increased SOC by greater than 10% over a period of six years. However, CO_2 emissions from these same beds were 1.2 times larger compared with beds where residues were removed and by a similar magnitude greater where they were tilled, but less so when a permanent raised bed planting system was used.

The sequestration potential of a whole raft of measures for the EU-15 countries is estimated to be 45 Tg of C per year (considering only constraints on land-use), but about 20% of this value is considered realistically achievable (Smith and Faloon, 2005). Although Smith and Faloon (2005) suggest options, which include reduced tillage and NT, improved crop rotations, soil application of organic amendments and extensification, among other measures, they do not include cover cropping. In contrast, Komatsuzaki and Ohta (2007) recognize cover crops as an essential tool for sustainable soil management, not only because of their potential to sequester C but as scavengers of residual soil N. Smith (2004) highlights that "soil C sequestration could meet at most about one-third of current yearly increase in atmospheric CO_2-C, but the duration of the effect would be limited, with significant impacts lasting between 20 and 50 years." However, CTF has the potential to absorb more CO_2 through increased cropping frequency or crop yield (e.g., by 10% to 30% for winter cereal crops under northern European conditions compared with non-CTF; Smith et al., 2014) and so provide more C-rich residue which, together with NT and reduced stimulation of oxidation, could increase SOM and possibly long-term C sequestration.

It appears that C dynamics are finely balanced, with specific circumstances leading to sequestration while others of little difference lead to C loss. The introduction of cultivation in its broadest sense means that soils lose SOC, but whether this is due to the introduction of cropping or the practices associated with it is less clear. Further delicacy of interpretation is associated with depth of measurement, with greater depths diluting what may appear to be a highly significant effect in the topsoil. Thus, tying down the particular effect of soil compaction is likely to require well-designed and carefully conducted experimentation.

Field Practices that Lessen Carbon Dioxide Emissions

Oxidation of SOM is enhanced by tillage, particularly in warm and moist environmental conditions. Therefore, crop production systems that reduce the need for tillage and its intensity will reduce the risk of oxidation of SOM. Linn and Doran (1984) measured 3.4 times higher CO_2 emissions from NT compared with plowed soils and concluded that this was largely due to 62% water-filled pore space (WFPS, eq. 1) that supported a relatively higher level of microbial activity compared with 44% WFPS on the adjacent plowed soils. In contrast, Beare et al. (2009) observed that a non-compacted sandy clay loam, having approximately equal parts of sand, silt, and clay, produced 2.3 times more CO_2 when continuously wet than its compacted counterpart. However, as the compacted soil dried from 46% to 14% (v v^{-1}), CO_2 production was about 55% higher than from the non-compacted soil, which had a significantly lower WFPS. WFPS is a significantly more important determinant of CO_2 production in non-compacted soil than in compacted soil (Beare et al., 2009).

$$\text{WFPS} = \frac{(\theta_g \times \rho_b)}{\eta} \tag{1}$$

where WFPS is water-filled pore space (%), θ_g is gravimetric water content (g g^{-1}), ρ_b is soil bulk density (g cm^{-3}), and η is total porosity (cm^3 cm^{-3}) (Linn and Doran, 1984). Total porosity of soil is determined from density properties (McKenzie et al., 2002; eq. 2):

$$\eta = 1 - \frac{\rho_b}{\rho_p} \tag{2}$$

where ρ_p is particle density (g cm^{-3}).

A close relationship exists between traffic-induced compaction and tillage, with greater traffic intensity leading to increased tillage intensity (Arndt and Rose, 1966). This leads to a cycle of compaction-tillage-re-compaction, which increases energy requirements of primary and secondary tillage by factors of up to 2.5 and 3.25, respectively, depending on the depth of the operation (Chamen et al., 1996). Similar results have been reported widely (e.g., Williford, 1980; Lamers et al., 1986; Chamen et al., 1992b; Chamen and Cavalli, 1994; Dickson and Ritchie, 1996; Godwin, 2012; Tullberg, 2010, 2014). Indeed, as much as 90% energy savings occur within production systems that avoid traffic-induced compaction on the cropped area. Lamers et al. (1986) and Spoor (2006) also reported energy savings of up to 48% due to lower rolling resistance on permanent traffic lanes compared with running over cultivated soils.

Anecdotal evidence from farmers converting to CTF and reduced tillage or NT suggests up to 50% savings in tractor diesel fuel for field operations. The average diesel fuel use on farms in the United Kingdom varies between approximately 90 L ha^{-1} for cereals production to 420 L ha^{-1} in the horticulture sector (DEFRA, 2009). For cereals, fuel use ranges from less than 50 L ha^{-1} to more than 150 L ha^{-1} (fig. 1). However, these farms may often include small areas of horticultural production or intensive livestock. More information can be gleaned from the area of cereals established using reduced tillage. Of the total combinable crops area in England amounting to 1.76M ha, 0.48M ha were established using reduced tillage (DEFRA, 2009). Where reduced tillage accounted for less than 50% of the farmed area, diesel fuel use averaged 94 L ha^{-1}, and this forms a useful basis from which the impact of any reduction in fuel use might be gauged. If 50% savings in fuel use (of 94 L ha^{-1}) were achieved by avoiding compaction across the combinable crops area in England, this would save 82×10^6 L per year, equivalent to 220 Mg CO_2-equivalent (CO_2e) or 126 kg ha^{-1} of CO_2e per year (if 1 L of diesel fuel = 2.672 kg CO_2e; Srivastava et al., 2006).

Savings in the horticulture sector would be more significant on a per hectare basis, although over a smaller but undefined area in the statistics provided by DEFRA (2009). These figures are only indicative values because of the diversity of the enterprises surveyed. More detailed information as well as an indication of the disparity can be gained from Chamen and Cope (1994), who tabulated the energy required for a reduced tillage system and for a system based on moldboard plowing. Each set of data was taken from four years of field results using tractor fuel consumption for cultivations but not including drilling. The reduced tillage system used 263 MJ ha^{-1}, compared with 564 MJ ha^{-1} required with the plow-based system. Thus, avoiding compaction in the plow-based system would save 282 MJ ha^{-1} (50% of 564 MJ ha^{-1}), which in terms of fuel usage equates to about 6 L ha^{-1} (based on 47.8 MJ L^{-1} energy cost of diesel fuel, including its production), a significantly lower, and probably more realistic, figure than the one based on national statistics by DEFRA (2009). In England, this would represent a saving of approximately 28 Mg of CO_2e per year. The reader is referred to work compiled by Holland (2004), which discusses potential energy savings achievable with conservation agriculture compared with conventional agriculture relevant to European farming systems.

In Australia, under conditions representative of grain cropping in southeastern Queensland, Tullberg (2000) observed that the traffic effect of wheels on the draft of tillage implements (chisel and sweep tines operated behind a 70 kW tractor in a Vertisol, 250 mm depth) increased total draft by 30% or greater compared with the same implement operated in non-trafficked soil. The same work also indicated that about 50% of a tractor's power output may be

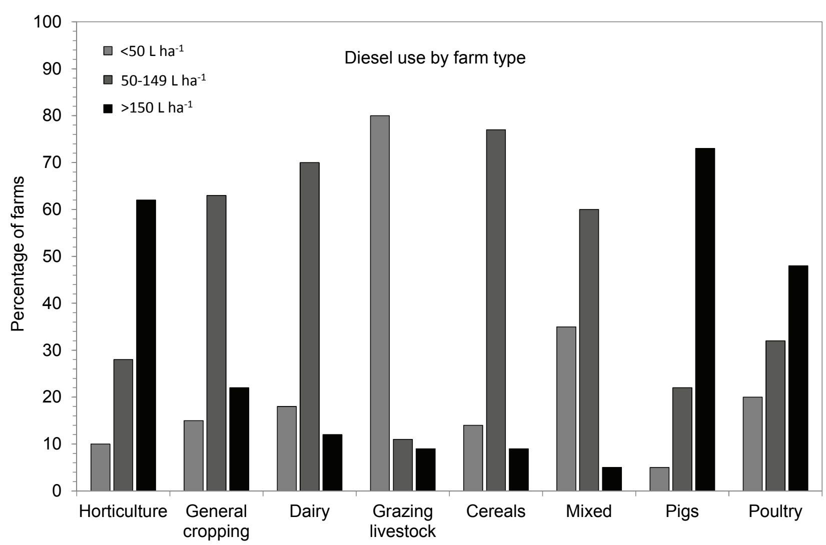

Figure 1. Diesel fuel use across different sectors by farm type in England (after DEFRA, 2009).

dissipated in the process of creating and disrupting its own wheel compaction, and Tullberg uses these observations to explain the potential reduction in tillage energy that occurs in CTF systems. As a reference, fuel usage estimates for chisel or sweep tillage operations are in the range of 7 to 12 L ha^{-1} (Tullberg, 2014). Reductions in tillage energy are primarily due to: (1) relatively lower soil specific resistance in the absence of traffic compaction, (2) tillage operations conducted at shallower depths when remediation of deep compaction is not required, and (3) reduced power loss in ground drive due to lower rolling resistance and reduced wheel slip (Tullberg, 2000; Spoor, 2006; Godwin, 2007).

Improved fertilizer use efficiency in CTF compared with non-CTF systems (Galambošová et al., 2014) has beneficial impacts on energy use efficiency. Energy balance calculations (Pimental, 2009) for typical wheat production systems in the U.S. suggest that N and P fertilizers with reported (mean) application rates of 70 and 34 kg ha^{-1} for N and P, respectively, represent about 30% (~5200 MJ ha^{-1}) of total energy input for the crop. Therefore, increased crop yield and fertilizer recovery will improve the MJ output-to-input ratio. In terms of the effects of soil structure on CO_2 emissions, there is a great deal of complexity driven by the dynamics of soil microbial populations, which react with soil moisture content and the WFPS created by different levels of compaction. However, in terms of energy inputs to soils, there is clear evidence of a significant reduction in emissions when controlled traffic creates large areas of non-compacted soil (reduced draft) together with firm traffic lanes delivering improved tractive efficiency and timeliness of field operations.

NITROUS OXIDE

Nitrous oxide (N_2O) is produced in soils by two processes: nitrification and denitrification (fig. 2) (Chadwick et al., 2011). Conversion of ammonium (NH_4^+) into nitrate (NO_3^-) through nitrification is a source of N_2O and produces NO_3^-, which is the source of N for denitrification, that is, the microbial reduction of NO_3^- to dinitrogen (N_2), where N_2O is a product of incomplete denitrification (Chadwick et al., 2011). In this process, NO_3^- can be used instead of oxygen (O_2) as a terminal acceptor for electrons from microbial respiration when the concentration of O_2 in soil declines to $\leq 3 \times 10^{-6}$ M (Greenwood, 1962). The amounts of N_2O given off are relatively small; however, microbial actions are highly variable over time and space. Nitrification occurs with WFPS up to about 55% (Groffman and Tiedje, 1991). Denitrifying bacteria thrive in anaerobic conditions and rapidly increase activity when WFPS reaches about 60% (Groffman and Tiedje, 1988). The relative rate of N_2O to N_2 production is high when the environmental conditions for complete denitrification are low (Granli and Bøckman, 1994). Therefore, N_2O is the prevalent reaction product under conditions marginal for complete denitrification (Granli and Bøckman, 1994). Denitrification processes leading to production of N_2O carried out by fungi under anaerobic conditions are discussed by Shoun et al. (2012). Under acidic conditions, NO_2^- can undergo chemodenitrification to form N_2O and N_2 (Bremner and Nelson, 1968; Macdonald et al., 2011). N_2 emissions do not represent a risk to the atmosphere; however, they are a direct loss of N to the air that is therefore not available to the crop. Soil conditions, such as pH, water content, temperature, and availability of easily decomposable SOM, NH_4^+, and NO_3^- are key determinants of how much N_2O a particular soil will produce (Russell, 1988). As will be noted in the following section, aeration of soils is one of the main driving forces for the type of processes involved, with low soil air leading to potentially significant production of N_2O.

EFFECT OF WATER-FILLED PORE SPACE

Studies conducted at Rothamsted Research (Harpenden, United Kingdom) by Bremner and Shaw (1958) showed that the rate of denitrification was affected by soil pH and temperature. However, the degree of water saturation had a significantly greater influence. Bremner and Shaw (1958) found that the critical moisture level for Rothamsted soils was at about 60% water holding capacity, above which denitrification increased rapidly despite these conditions being temporal and dependent on rainfall and plant growth characteristics. Noting these same temporal effects, Linn and Doran (1984) observed that soils under permanent NT at four locations had an average of 62% WFPS, whereas plowed soils at the same locations had 44% WFPS. This difference was reflected by a 9.4-fold increase in emissions of N_2O over a 24 h period on the NT soils, with similar contrasts reported by Aulakh et al. (1984). Dobbie et al. (1999) also concluded that WFPS was a dominant factor in N_2O emissions, finding that the largest fluxes occurred between 70% and 90% WFPS.

Soil compaction increases the volume of soil vulnerable to denitrification, especially at high soil water potential (Boone and Veen, 1994). Even short periods under anaerobic conditions can cause large N losses through denitrification and significantly affect the potential specific supply rate of NO_3^- (Boone and Veen, 1994). Ruser et al. (2006) investigated N_2O emissions from silty clay loam cores taken from a field under potato production. They determined the effect of tractor-induced compaction and soil moisture on N_2O fluxes and examined the influence of wetting and drying cycles on such emissions following application of a

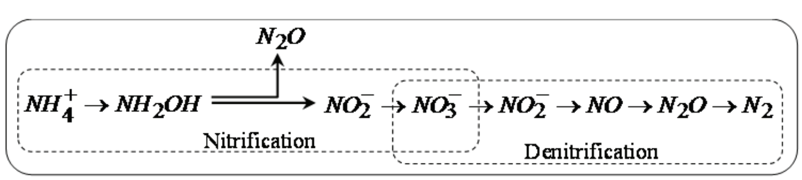

Figure 2. The processes involved in the release of nitrous oxide (N_2O) to the atmosphere (after Chadwick et al., 2011, with permission). A similar conceptual model, which includes respiration pathways, is given by Laudone et al. (2011).

NO_3^--based fertilizer. It was observed that N_2O emissions from the non-trafficked soil represented approximately 8% of the N applied as fertilizer, compared to 17.5% from the tractor-compacted soil in the interrow (Ruser et al., 2006). A summary of their experimental results is quoted in table 1.

The higher WFPS value used for the trafficked interrow reflected field measurements taken in this region, which were always higher than 60%, whereas in the other areas, WFPS in the region ranged between 40% and 70%. As can be seen in table 1, the actual field conditions are critical in terms of N_2O emissions. High N_2O emissions in ridges were associated with higher availability of organic C from crop rooting. High production of N_2O occurred in compacted areas at WFPS ≥70%, whereas production of CO_2 was not affected by soil moisture content (Ruser et al., 2006). For all treatments, the highest N_2O emission rates were recorded following a re-wetting phase. Burger et al. (2005) measured CO_2 and N_2O efflux after irrigation or simulated rainfall on two tomato fields in California's Central Valley. The highest CO_2 efflux occurred at about 60% WFPS, and elevated emissions of N_2O occurred at WFPS above 60%. Irrigation management was seen as a method of controlling the duration of elevated emissions, even when C and inorganic N availability were high. Beare et al. (2009), working in Canada on a loam soil cultivated to maize, measured N_2O emissions from compacted and non-compacted soils in three conditions: while maintained at field capacity, during a drying phase, and during a re-wetting phase. At field capacity, N_2O emissions were 67 times higher from the compacted soil than from the non-compacted soil. This contrast remained during drying, when emissions from the non-compacted soil decreased to 7% of their value during the wet phase and the compacted soil decreased to 4%. However, emissions from the compacted soil still represented 20 times that of the non-compacted soil. During the first day of the rewetting phase, N_2O production from the non-compacted and compacted soils increased to 13 and 259 μg N_2O-N kg^{-1} d^{-1}, respectively, but decreased rapidly over the next three days to the levels observed at field capacity. During the 18-day measurement period including wet, dry and re-wet phases, the cumulative production of N_2O-N from the non-compacted soil was 15 μg N_2O-N kg^{-1} of dry soil, compared with 296 μg N_2O-N kg^{-1} of dry soil from the compacted zone, whose density was 1.49 g cm^{-3} compared with 1.01 g cm^{-3} for the non-compacted soil. Khalil et al. (2002) observed that an increase in WFPS from 35% to 75% during a re-wetting phase of a loamy soil under maize increased N_2O emissions from approximately 600 to 10,000 μg N_2O-N m^{-2} per day. The increment in emissions was higher as chicken manure was applied but decreased as the soil dried, despite application of N at the point of peak emission. Within ten days, N_2O emissions had fallen to their initial value coincident with a WFPS of around 53%. Allen et al. (2009), working with sugarcane in subtropical soils in Queensland, Australia, reported positive correlations between N_2O emissions and soil temperature, WFPS, and soil mineral N content. Therefore, significant reductions in N_2O emissions could be achieved by avoiding waterlogging in the presence of high levels of soil mineral N (Allen et al., 2009), which agrees with observations made by Smith et al. (2012). This latter work also highlights the effect of timing of N application and source, and its relationship with WFPS and soil temperature in developing conditions leading to denitrification.

It is widely believed (e.g., Bremner and Shaw, 1958; Aulakh et al., 1984; Linn and Doran, 1984; Burger et al., 2005) that a critical WFPS of approximately 60% is the threshold above which N_2O emissions increase rapidly. Some researchers (e.g., Ball et al., 2008) have reported a close link between elevated WFPS and high N_2O emissions. Li et al. (2005b) plotted WFPS against denitrification (fig. 3), suggesting that if the relative WFPS on non-trafficked soils compared with trafficked soils could be determined, the potential of CTF to mitigate N_2O emissions would be more clearly understood. This relationship might be inferred from research assessing changes in porosity in trafficked soils compared with non-trafficked soils. Several studies (e.g., Canarache et al., 1984; Blackwell et al., 1985; Campbell et al., 1986b; McAfee et al., 1989; Wagger and Denton, 1989; Dickson and Campbell, 1990; Schäfer-Landefeld et al., 2004; Vero et al., 2013) have identified

Table 1. Nitrous oxide (N_2O) emissions from a silty clay loam soil under potato production as affected by nitrate-based fertilizer application, soil moisture content, and soil compaction (after Ruser et al., 2006, with permission). SBD is soil bulk density, and WFPS is water-filled pore space.

Area Sampled	SBD ($g\ cm^{-3}$)	WFPS (%)	Maximum N_2O Flux (μg N_2O-N m^{-2} h^{-1}): After N Fertilizer Application[a]	Maximum N_2O Flux: After Re-Wetting	Cumulative Emissions[b] (mg N_2O-N m^{-2})
Potato ridges	1.02	90	1426.3	2412.0	1677 a
	1.02	70	128.0	639.0	127 b
	1.02	60	10.9	75.7	9 c
	1.02	40	11.4	17.7	4 d
Non-trafficked interrow	1.24	90	1046.3	1578.7	1218 a
	1.24	70	29.3	41.9	20 b
	1.24	60	8.7	40.6	9 b
	1.24	40	5.8	35.2	9 b
Trafficked interrow	1.65	98	1768.1	4568.4	2620 a
	1.65	90	411.4	494.3	322 b
	1.65	70	45.4	65.7	28 c
	1.65	60	6.7	166.7	5 d

[a] Maximum emissions were typically observed between 1 and 3 days following N fertilizer application, depending on WFPS; N fertilizer was applied on day 0, the soil dried on day 43, and the soil was re-wetted on day 55 (Ruser, 2014).

[b] Cumulative N_2O-N emissions were from a period of 58 days, and background emission rate at ≤60% WFPS was between 1 and 12 μg N_2O-N m^{-2} h^{-1}. Different letters indicate that values within sampled areas are statistically different at a 5% confidence level (Ruser et al., 2006).

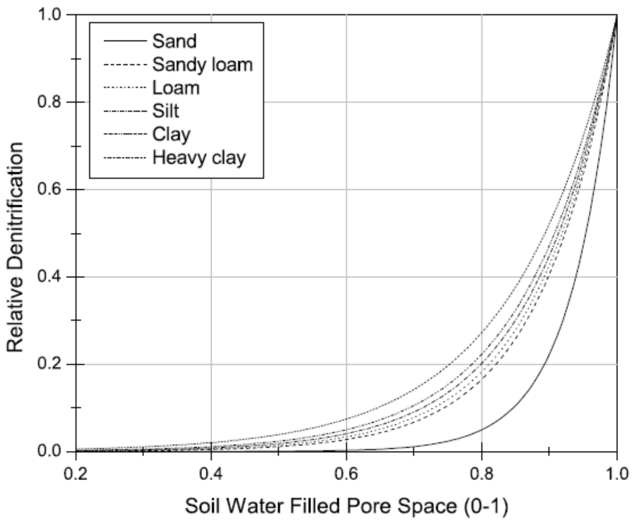

Figure 3. Effect of water-filled pore space (WFPS) on denitrification in differently textured soils (after Li et al., 2005b, with permission).

field traffic-related reductions in soil porosity in the range of 5% to 70%, with the greatest reductions occurring in the larger drainage pores.

EFFECT OF NITROGEN FERTILIZATION AND IMPACT ON FERTILIZER USE EFFICIENCY

The impact of soil compaction on nutrient uptake is significant (Alakukku and Elonen, 1995; Torbert and Reeves, 1995a, 1995b; Ishaq et al., 2003). This has financial implications for growers due to reduced fertilizer use efficiency (FUE) and potential crop yield, and therefore lower economic return from the fertilizer applied. Nutrients applied to soil that are not used by the crop are prone to environmental losses by processes linked to compaction, such as transport in overland flow or gaseous evolution. Soil compaction affects nutrient uptake through adverse effects on the following mechanisms (Lipiec and Stępniewski, 1995): (1) nutrient transport, absorption, and transformation in response to changes in soil aeration and hydraulic characteristics, which influence mass flow and consequently transport of NO_3^--N (Okajima and Taniyama, 1980; Wolkowski, 1990); (2) nutrient diffusion in the proximity of plant roots, which influences transport of P and K (Barraclough and Tinker, 1981; Barber, 1984); and (3) induced modification of root architecture due to increased soil strength, which influences root interception of nutrients (Barraclough and Weir, 1988; Atwell, 1993). The effect of compaction on diffusion processes of P and K may lead to further reductions in crop N uptake due to positive interactions that exist between N, P, and K (Azad et al., 1993; Aulakh and Malhi, 2005; Johnston and Milford, 2007; Antille et al., 2013b). Nitrogen deficits caused by reduced uptake of nutrients have an impact on potential crop yield and reduce water use efficiency (Sadras and Rodriguez, 2010). Therefore, soil conditions conducive to high rates of nutrient uptake by the crop following fertilizer application will enhance agronomic performance and reduce opportunities for environmental losses (Antille et al., 2015a). The "4Rs" principle of nutrient management, that is, optimized crop nutrition with the right product, right place, right rate, and right time, has been suggested as a key strategy to increase use efficiency and mitigate impacts on climate change associated with fertilizer use (Roberts, 2007; Norton, 2014). However, these practices alone have not always translated into significant improvements in the efficiency with which N inputs are recovered in crops (Conant et al., 2013) and therefore require implementation in conjunction with other practices leading to improved soil conditions to maximize capture of such inputs. For example, much attention has been paid to selecting the right rate, which is regarded as being the best single predictor of N_2O emissions for several crops (Millar et al., 2010; Grace, 2014) and fertilizer products (e.g., Harrison and Webb, 2001; Shoji et al., 2001; Snyder et al., 2009; Smith et al., 2012). In this regard, enhanced-efficiency fertilizers, such as slow-release and controlled-release formulations, appear to be a promising technology to reduce N_2O emissions (Shaviv, 2001; Akiyama et al., 2010). Despite this, published information shows contrasting results in terms of their effectiveness to reduce N_2O emissions, particularly in situations where emissions are episodic and promoted by rainfall events (Parkin and Hatfield, 2014; Hatfield and Venterea, 2014). In irrigated crops, this effect is associated with irrigation at times when mineral N levels in soil are high (Scheer et al., 2013). Such high N levels may occur after application of N fertilizer or following a drying phase in which soil mineral N accumulates due to N uptake being restricted by soil water availability (Antille et al., 2014). Compaction is reported to have a relatively greater effect on N_2O emissions compared with increasing N fertilization rates (Gregorich et al., 2014). This latter study showed that compaction increased N_2O emissions but also had an adverse effect on crop yield and N uptake. By contrast, increasing N fertilization in the range of 0 to 300 kg N ha^{-1} increased N_2O emissions but also led to higher yields and N uptake, respectively (Gregorich et al., 2014). In the absence of significant or widespread compaction, N saved in gaseous emissions enables increased crop N uptake and recovery. Based on the fact that well-designed CTF systems can confine compaction to less than 20% of the cropped area, it is fair to state that these systems have the potential to increase FUE (i.e., greater economic return from fertilizer applied to crops) due to reduced compaction-induced N_2O emissions. The conditions leading to improved FUE in CTF systems are: (1) enhanced internal and surface drainage, (2) facilitation of more efficient fertilizer application, that is, improved trafficability and timeliness to synchronize field application (supply) with crop requirement (demand), and (3) reduced draft (mainly for materials requiring soil incorporation or injection). Brentrup and Pallière (2008) recognized the importance of drainage in one of their recommendations to improve N use efficiency, including avoidance of N application to waterlogged soils and the need to maintain a good soil structure to this end. Matching nutrient supply with crop demand is highlighted as an essential agronomic practice to optimize use efficiency and yield (Fageria and Baligar, 2005).

The relationship between N application rate and direct N_2O emissions is often non-linear (e.g., exponential or hyperbolic) (Kim et al., 2013). The rate of increase in direct N_2O emissions is significant when the N uptake capacity by the crop is exceeded (Kim et al., 2013). The argument that N fertilization rate is the best indicator of N_2O emissions from most arable crops is supported by a strong (exponential) correlation between N_2O emissions and N application rate (Millar et al., 2010; Hoben et al., 2011). For this reason, N must be applied at the optimum economic rate to minimize N_2O emissions (Millar et al., 2010; Hoben et al., 2011), which also agrees with recommended agronomic practices (e.g., James and Godwin, 2003). We also suggest that N fertilization guidelines based on optimum economic N application rates be revised for crops established in CTF due to the enhanced N uptake and recovery (by up to 20%) that are expected when traffic compaction is avoided (Galambošová et al., 2014). The N uptake capacity of crops established in CTF may be reached or exceeded at higher N rates compared with non-CTF systems due to a higher yield-to-N response curve. Therefore, direct N_2O emissions will become significant at relatively higher N rates in the response curve in CTF compared with non-CTF systems.

Effect of Tillage and Traffic

Soil structure largely determines the nature of physical processes that occur within a soil (Dexter, 1988; Lal, 1991). The deterioration of soil structure is regarded as a form of soil degradation and has a stretch relationship with land-use and soil management practices (Pagliai et al., 2004; Bronick and Lal, 2005). Practices that minimize soil disturbance will protect the SOC pool, which will favor the formation of stable aggregates and therefore will assist soil structural development (Bronick and Lal, 2005). Random traffic with heavy equipment often creates a vicious cycle of compaction-tillage-re-compaction. Therefore, reduced need for tillage when compaction is avoided helps protect SOC in stable aggregates, which may otherwise be exposed and oxidized. Soils under plow tillage often exhibit less stable aggregates and less SOC compared with permanent NT (Filho et al., 2002). However, the effect of tillage on SOC depends on the intensity and timing of tillage operations (Studdert and Echeverría, 2000). SOC levels in the topsoil are also influenced by crop rotation and N fertilization, which affect the amount of biomass C that can be returned to the soil through crop residues (Díaz-Zorita et al., 2002; Fageria, 2012). For rainfed crops in low-rainfall areas (e.g., <500 mm per year), potential crop biomass and yield are mainly constrained by water availability, and crop residues are subjected to high decomposition rates, which respond to temperature regimes characteristic of such environments (Chan et al., 2003). These constraints explain the lack of significant differences in SOC stocks often observed between NT and conventional tillage in low-rainfall areas, particularly in lighter soils (Chan et al., 2003). In grain cropping, increased opportunity for establishment of double crops is recognized as a major benefit of CTF compared with non-CTF systems, which is due to improved soil conditions and water availability within the soil profile (Yule and Radford, 2003). In central Queensland, Australia, where rainfall is summer-dominated, the frequency of successful crops is approximately 0.7 crops per year (Harrison and Tisdell, 1997; Yule and Radford, 2003). This frequency is reported to be approximately 1 with NT and 1.2 or higher (double cropping) when NT is coupled with CTF (Harrison and Tisdell, 1997; Yule and Radford, 2003), which has implications for biomass C sequestration and residue return to the soil. Despite this, there appears to be a time lag for conservation practices to improve soil properties and enhance the agronomic and environmental performance of the system. This time lag is mainly dependent on soil type and management, as observed by Rhoton (2000) when changing from conventional tillage to NT.

As a result of traffic-induced soil compaction, pore size distribution undergoes greater relative change than bulk density or total porosity (Vomocil and Flocker, 1961). Consequently, soils affected by compaction can develop local anoxic conditions within the profile at water contents near field capacity (Berisso et al., 2012). Similarly, the development of plow-pans in conventionally managed systems reduces soil porosity, especially vertically oriented pores, which restricts water and air flows (Pagliai et al., 2004) and may also lead to anoxic conditions at that depth. N_2O emissions from NT are reported to be higher than from conventional tillage only in poorly aerated soil conditions (Rochette, 2008). Similar conclusions were drawn by Ball et al. (2008) based on studies in field conditions. This latter work attributed increased and upward emissions of N_2O to increased WFPS, which also impaired downward movement of N_2O that would be more likely to be converted to N_2. CO_2 emissions were also increased in soils with high bulk density, but only at low (-1 kPa) water tension (Ball et al., 2008). The observations made by Rochette (2008) reinforced the fact that many of the benefits associated with NT may be offset by compaction caused by random traffic. In winter cereal crops in Scotland, Ball et al. (2008) found that field emissions of N_2O over a three-month period under NT established for four years were nearly four times those from similar density, conventionally managed, soil (plow-based). Emissions coincided with high rainfall events, even when these occurred 30 days after ammonium nitrate fertilizer had been applied. The elevated emissions under NT were associated with longer periods of high WFPS due in part to a lack of pore continuity. By contrast, Ussiri et al. (2009) measured significantly lower N_2O emissions from NT than from moldboard-plowed or chisel-cultivated soil. Annual fluxes of N_2O were equivalent to 1690, 1825, and 875 kg CO_2e ha^{-1} per year for tine-cultivated, plowed, and NT soils, respectively. These results were attributed to the long-term nature of the NT treatment (>40 years, compared with four years for the experiments of Ball et al., 2008) that had led to a lower soil bulk density than under the other treatments (Ussiri et al., 2009). It appears that NT systems have potential to mitigate N_2O emissions when practiced in the long term (e.g., >10 years) (Six et al., 2004). This requires that N management practices are optimized for the crop-soil system following conversion from conventional to NT. In the short term, because of relatively poor crop uptake of N and potentially higher WFPS, NT can result in increased GWP compared with conventional tillage. However, this may be reversed after ten or more years in humid climates and uncertainly in dry climates after a longer period of time (Six et al., 2004). The disruption of the pore network by field traffic, and in certain soils the time required for self-amelioration of the soil structure (Radford et al., 2007), may help explain the results encountered by Six et al. (2004).

It is clear that elevated emissions of N_2O are mainly associated with denitrification rather than nitrification in compacted soils, which appears to be the driving process influencing the N balance in those soils (Stepniewski et al., 1994; Liepec and Stepniewski, 1995). Equally, compaction affects soil aggregation and increases the likelihood and duration of elevated WFPS, and these elevated levels (>60%) are closely associated with elevated N_2O emissions (Bakken et al., 1987; Horn et al., 1994; Ball et al., 1999; Sitaula et al., 2000; Li et al., 2005b; Hansen, 2008).

Effect of Organic Residues

Although earlier research suggested that C sequestration is only marginally higher under conservation tillage compared with conventional tillage systems, several studies identified stratification, with relatively higher C levels near the surface where tillage was minimal or absent (e.g., Zhang et al., 2014). This stratification of organic matter is therefore important in terms of its influence on N_2O emissions. Li et al. (2005a) used a biogeochemical modeling technique to assess the impact of C sequestration on the N cycle and found that reduced tillage and enhanced crop residue incorporation increased C sequestration but also increased N_2O emissions. The net outcome of these techniques together with manure application therefore had the potential in CO_2e terms to offset between 75% and 310% of the C sequestered, depending on the scenario. Patiño-Zúñiga et al. (2009) compared permanent (non-trafficked) and conventionally tilled raised beds and tied ridges (to promote water retention) with different residue management methods. CO_2 and N_2O emissions over a 24 h period were found to be 1.2 and 2.3 times higher, respectively, in tilled compared with NT (non-trafficked) beds where residues were retained. In the low compaction scenario (residues retained in permanent raised beds), nitrate (NO_3^-) production was higher under tillage compared with NT. Higher N_2O losses in the tilled beds were attributed to the breakup of soil aggregates, which inhibited O_2 diffusion. In southeastern Scotland, N_2O emissions were found to increase temporarily after N fertilizer application, cultivation of bare soil, plowing-up of grassland, and soil incorporation of crop residues (Baggs et al., 2000). However, high emissions usually lasted for a short period of time (~2 weeks) and approximated to ambient levels within 4 to 5 weeks after the cultural practice was undertaken. The magnitude and pattern of emissions were correlated with the type of tillage operation and the C:N ratio of the residue (Baggs et al., 2000). Soil incorporation of residues with narrow C:N ratios (e.g., ≤20) would normally yield relatively higher N_2O emissions compared with materials having wider C:N ratios. There is considerable literature on this subject, whose analysis cannot be justified here. However, it is noted that SOM is the energy source for the microbial processes that drive N_2O fluxes; hence, its presence or absence will have a significant effect on outcomes. The stratification of SOM commonly observed with NT (Zhang et al., 2014) may be exacerbated by compaction. However, the combination of reduced compaction and increased soil biota, hence, biological activity, and root penetration into the soil reduces stratification effects (Pangnakorn et al., 2003).

Synthesis

Soil Carbon and Carbon Dioxide

There is still much uncertainty and apparent contradiction in the results of research on C sequestration related to specific soil management techniques. This is partly due to inconsistencies in experimental designs and methods of assessing SOC sequestration, but also to a lack of understanding of SOC sequestration concepts (Olson et al., 2014). Bellamy et al. (2005), who addressed these sampling issues in their research, indicated that the relationship between the rate of soil C loss and C content in England and Wales was observed across all types of land-use, therefore suggesting a link to climate change. Similarly, research results from cropped lands appear to be contradictory in terms of tillage effects. Some research claims increases in SOC with NT (e.g., Costa Junior et al., 2013; Six et al., 2000b); other results report no difference (e.g., Domínguez et al., 2009) even after extended periods and to a significant depth in the profile (e.g., Angers et al., 1997; Chan et al., 2011, except when a pasture-phase was included in the rotation); while other work suggests that rates of SOC losses in NT are significantly reduced compared with conventional tillage (e.g., Page et al., 2013). This uncertainty is extensively discussed in several reviews (e.g., Hutchinson et al., 2007; Olson, 2013; Lal, 2013b) and is confirmed by Powlson et al. (2011), who suggested that increases in SOC from reduced tillage and NT systems may be smaller than claimed. Therefore, the main benefits of NT may be associated with improved soil quality and adaptation of agriculture to climate change, but with limited potential for its mitigation (Powlson et al., 2014). However, the potential benefits of coupling NT with CTF to reduce GHG emissions and enhance C sequestration in arable land are not discussed in previous analyses and do not appear to have been investigated experimentally. West and Post (2002) estimated potential soil C sequestration rates for a range of crops in relation to reduced tillage intensity or increased rotation complexity and determined the duration of time during which C sequestration occurs. Their study suggests that C sequestration rates following a switch from conventional to NT are likely to peak within 5 to 10 years, and that SOC levels would reach a new equilibrium within 15 to 20 years, but unfortunately their study does not consider the actual effect of traffic compaction on C sequestration rates.

Also crucial to any of these results is the depth of sampling. If the total accumulation of C is the measure of importance, then the work of Gál et al. (2007) is probably definitive. Beare et al. (2009) make the connection between WFPS and CO_2 emissions, suggesting that the greater WFPS generally experienced with non-trafficked soils tends to generate more CO_2 when this soil is wet. This is expected owing to greater biological activity in non-compacted soils, compensated for by greater CO_2 absorption by plants (Pangnakorn et al., 2003). However, the work reported by Beare et al. (2009) was not conducted in the field and therefore did not take account of actual field conditions. There is sufficient evidence to state that non-trafficked soils drain more readily (e.g., Blackwell et al., 1985; Lamers et al., 1986; Campbell et al., 1986a; McAfee et al., 1989; Chamen et al., 1992b; Hamilton et al., 2003) and may therefore experience much shorter periods with elevated CO_2 emissions than compacted soils. Compacted soils in this condition also emit less CO_2, so the crucial question is: how long do these soils remain in these relative conditions in the field? This will be determined by many different factors, one of which is the soil pore structure and its connectivity with the drainage system, whether natural or otherwise. The reason for the variability in research results is, therefore, almost certainly associated with the very specific conditions under which the trials are conducted as well as the depth of sampling. As is often stated, more research is needed to understand the underlying mechanisms, processes, and microbiological activity involved before the effects of different soil structure can be understood and possibly predicted. What is known is that organic matter is sequestered as a result of photosynthesis, and its abundance is primarily driven by the climate and vegetation present, which in the case of agriculture is the crops being grown, their rotation, and management practices, including tillage and fertilization. Assuming these are decided by commercial realities, the only additional influence that a grower might have is the inclusion of cover crops. These have been shown to increase the SOC pool. However, as with all organic matter, changes occur over a relatively long period, and they may only be invoked by the appropriate species composition and cropping cycle (Lal, 2004a; Franzluebbers, 2005; Sainju et al., 2007; Bavin et al., 2009). If soil compaction can be avoided, there is a greater opportunity for establishing cover crops because of the greater ease of creating seedbeds, which will also mean quicker and more reliable establishment (Chamen, 2009). Another indirect effect is quoted by Powlson et al. (2011) where water conservation under NT combined with surface mulching allows an extra crop to be grown, which is a direct addition of C from the atmosphere to the soil. Similarly, extra cropping or "opportunity" cropping is commonly practiced in Australia when a change to CTF (and long-term NT) improves water infiltration during intense rainfall events (Li et al., 2009). In terms of C sequestration in arable land, increased frequency of successful crops (approximately ≤0.7 with traditional tillage and random traffic, 1.0 to 1.1 with NT and random traffic, and ≥1.2 crops a year with NT and CTF) and yield (≥15%), and hence biomass, appears to be a distinctive advantage of CTF over non-CTF systems for relatively low rainfall environments, as discussed earlier for central Queensland conditions in Australia (Harrison and Tisdell, 1997; Yule and Radford, 2003; Li et al., 2007, 2008). The studies by Campbell et al. (2005) in Canada comparing tilled and NT soils suggest a positive relationship between the rate of change of SOC and cropping frequency, and therefore may support the above observations.

Satisfactory fertilization levels to support high crop biomass, and subsequent return of crop residues to soil, is mentioned by Campbell et al. (2001) as a key factor influencing changes in SOC. The need for continuous improvement in crop yield has been highlighted as a key strategy to mitigate the contribution of agriculture to global GHG emissions (Burney et al., 2010) and therefore reinforce the potential of CTF to advance crop production within a sustainable framework.

In terms of C oxidation from soil, the primary factors involved are aeration, temperature, water content, and the microbes that feed on the SOM. As aeration in the absence of machinery compaction is likely to be enhanced and more consistent than under traditional management (Blackwell et al., 1985; Wagger and Denton, 1989; Alakukku, 1996; McAfee et al., 1989), oxidation of SOM may be greater. However, the process may be offset by greater root deposition and crop residues from increased crop biomass (Håkansson, 2005; Qingjie et al., 2009; Reintam et al., 2009) and by less soil manipulation that exposes otherwise protected SOM. Overall, the only presently known outcome will be due to differences in crop biomass; with the extra cropping opportunities offered by controlled traffic, this is likely to result in soil C sequestration. However, this requires that crop residues are not removed (at least to a large extent) to ensure a high residence time and stability of the C sequestered in soil so that there is no immediate re-emission as a result of management practices (Lal, 2007). For Australian soils, Parton et al. (1996) suggested that in order to achieve positive changes in soil C (>0 g m^{-2} per year), total C inputs must be greater than approximately 300 g m^{-2} per year, whereas for Swedish soils, a positive balance may be achieved with 50% of that C input. Therefore, the effects of climate and soil type on C sequestration are significant and confirm the need to ensure that crop residues are returned to the soil.

NITROUS OXIDE

Most of the research consulted compared the effects of different degrees of machine-induced soil compaction rather than compaction or its complete absence. In contrast, work conducted by Vermeulen and Mosquera (2009) provided a direct comparison between non-trafficked and conventionally managed soils, which indicated a 20% to 50% reduction in N_2O emissions within a vegetable production system with seasonal controlled traffic, as well as a 5-fold to 20-fold increase in CH_4 uptake. Alakukku and Elonen (1995) determined significantly higher N uptake by crops in non-trafficked soils compared with trafficked soils, thus reducing the risk of emissions through wastage. Torbert and Reeves (1995a, 1995b) and Galambošová et al. (2014) drew similar conclusions and reported N recoveries that were 10% to 20% higher where traffic compaction was absent. These findings tend to question the rather simplified calculation of emissions provided by the Country Land and Business Association (United Kingdom) through their CALM (Carbon Accounting for Land Managers) calculator (CLA, 2011). CALM is a business activity-based calculator that enables estimation of the balance between annual GHG emissions and C sequestration associated with activities of land-based businesses. CALM calculations are based on yield potential alone, rather than differences in soil management regimes within a particular cropping rotation. What is certain from the research is that non-trafficked soils retain a pore structure and continuity that minimizes the risk of increased WFPS above a suggested critical value of 60% (Lamers et al., 1986; Sexstone et al., 1988; Chamen et al., 1992b; Dobbie et al., 1999; Li et al., 2005b; Berisso et al., 2012). It is also evident that non-trafficked soils have greater infiltration rates, which are often matched with improved hydraulic conductivity (Canarache et al., 1984; Meek et al., 1989; Strudley et al., 2008; Li et al., 2009; Rashid et al., 2015).

When considering the realization of "non-trafficked" soils within a CTF system, the impact of permanent wheel tracks needs to be considered. These typically represent 20% or less of the cropped area in well-designed systems and could lead to localized increased emissions of both N_2O and CH_4. However, the risk of high localized emissions can be significantly reduced by avoiding or reducing application of N fertilizer to traffic lanes and by minimizing the amount of organic matter from crops and residues added to soil in this area (Antille et al., 2015b). Avoiding application of fertilizer in this area is relatively simple with a liquid system based on individual nozzles and may be possible with solid materials applied with pneumatic applicators or incorporated either in the interrow or with the planter. However, it may be more problematic with solid fertilizers applied with twin-disk spreaders (Antille et al., 2015c). Split applications of N fertilizer, soil incorporation and site-specific placement, and careful selection of the N source based on soil conditions and soil type may also be an alternative to reduce N_2O emissions. However, split applications may be less effective with the use of urea and may result in increased N_2O emissions compared with single dressing (Venterea and Coulter, 2015). The reader is referred to several studies that deal specifically with the effect of N fertilizer source or placement on N_2O emissions (e.g., Granli and Bøckman, 1994; Harrison and Webb, 2001; Tenuta and Bauchamp, 2003; Liu et al., 2006; Snyder et al., 2009; Burger and Venterea, 2011; Smith et al., 2012; Gao et al., 2015). Sown wheel tracks will naturally have less crop biomass as a result of reduced growth; however, the most effective approach is to minimize the tracked area of the CTF system.

Modeling of Nitrous Oxide Emissions

Several models have been developed to predict N_2O emissions from soil, and these are useful in isolating the main factors involved. A compilation and in-depth analysis of available models are given by Chen et al. (2008), and the reader is referred to their work for further details. Chen et al. (2008) categorized available models into three types: (1) laboratory-scale, (2) process-based field-scale, and (3) regional/global-level models. The main challenge is the scaling-up of relatively robust field-scale models to enable their application at larger-scale situations to: (1) produce accurate inventories of N_2O emissions, and (2) assess mitigation practices (Chen et al., 2008). All models have a WFPS or aeration factor associated with them along with N content, temperature, and often soil pH. Frolking et al. (1998) used four models, all of which relied on soil water status for N_2O emission simulations, and three relied on WFPS in particular. Soils used to test the models and their WFPS are listed in table 2. Frolking et al. (1998) concluded that accurate simulation of soil moisture is a key requirement for reliable simulation of N_2O emissions, but the models tested were unable to predict large pulses of N_2O flux that occurred during freeze-thaw cycles. These fluxes can dominate the annual flux on some soils subjected to such processes (Frolking et al., 1998).

Li et al. (2005b) compared three modeling approaches for simulating N_2O emissions from loam-textured arable soils. Of these, the gas module of the Water and Nitrogen Management Model (WNMM) (Li, 2002) provided the most reliable results compared with measured emissions. Denitrification was simulated as a function of soil NO_3^- content, soil WFPS, and SOC. In the case of the WNMM gas module, Li et al. (2005b) differentiate between denitrification under saturated and unsaturated soil conditions. From Xu et al. (1998), they quote maximum N_2O emissions as a constant 1% of denitrification under saturated conditions and a maximum of 50% under non-saturated conditions. Thus, the percentage of N_2O emissions decreases with increasing WFPS above the point at which denitrification begins. Consequently, although denitrification may increase with increase in WFPS, the proportion of N_2O in relation to N_2 flux is assumed to decrease. However, whether N_2O emissions peak and then fall during this process is unclear and may vary from soil to soil (Y. Li was queried on this point, but no reply was received. B. C. Ball, when presented with these data, questioned the fall in proportion of N_2O in relation to N_2 flux). Some suggestion of this is noticed in the data of Jantalia et al. (2008), where emissions peaked at WFPS in the range of 40% to 60%. However, this is contra-indicated in figure 3, which shows that the rate of denitrification increases in all soils with increasing WFPS, and that it starts at lower levels in heavier soils. Gas diffusion rate is also a factor because it governs the O_2 supply for the respiring organisms involved in SOM decomposition. Poor diffusivity will promote anaerobic microbial activity more quickly than might occur otherwise, as discussed in other work (e.g., Lipiec and Stępniewski, 1995; Patiño-Zúñiga et al., 2009). Bouwman et al. (2002) used information from over 800 N_2O emission measurements to model factors regulating these emissions and concluded that the key parameters controlling emissions were N application rate for a particular fertilizer (also indicated by Millar et al., 2010), SOC content, and soil

Table 2. Water-filled pore space (WFPS) of different soils at field capacity (FC) used by Frolking et al. (1998) to test four emissions models and their mean WFPS. The simulated WFPS values are coarse averages obtained from the models over 700 days.

Location	Models Applied[a]	Soil Texture	WFPS at FC	Simulated WFPS
Colorado, U.S.	C, D, E, NSA	Sandy loam	0.27	0.20
Scotland	C, D, E	Clay loam	0.80	0.73
Germany A	C, D, E	Sandy loam	0.73	0.58
Germany D	C, D, E	Loam	0.83	0.66

[a] C = CENTURY, D = DNDC or Denitrification-Decomposition, E = ExpertN, and NSA = NASA CASA.

drainage. Total global annual emissions of N_2O from fertilized fields were calculated to be 2.8 Mg, representing 0.9% of total N applied. Conen et al. (2000) used an empirical model to predict N_2O emissions from agricultural soils using three parameters: topsoil mineral N, WFPS, and soil temperature. Soil mineral N was considered with two thresholds (<10 and >10 mg kg^{-1} of soil), and outputs ≥10 mg kg^{-1} were sorted into three N_2O flux ranges using a product of soil temperature (<5°C and >5°C) and WFPS, giving figures of <90%, 90% to 105%, and >105%. Results provided satisfactory predictions of seasonal fluxes, confirming the central role of soil water content in N_2O emissions. Del Grosso et al. (2000) applied a model to predict N_2O emissions but used O_2 availability as a function of soil physical properties that included WFPS. Similarly, predicted results showed a satisfactory correlation with measured rates from irrigated field soils.

An important component of process-based models is their ability to deal with the high spatiotemporal variability of N_2O emissions (Giltrap et al., 2009). The Denitrification-Decomposition (DNDC) model, in particular, has been expanded to include a wider range of ecosystems and land management effects on emissions of all three GHGs. Bessou et al. (2010) used the Nitrous Oxide Emissions Model (Hénault et al., 2005) to consider the relative impact of soil compaction. It must be stressed that these were relative effects in that the site was prepared by moldboard plowing and conventional tillage, and therefore subjected to random traffic, with an added overall compaction treatment, presumably right before final seedbed preparation. This tillage treatment raised the bulk density of the soil (Orthic Luvisol) from approximately 1.40 to 1.60 g cm^{-3}. The model used the availability of mineral N, soil water content, and temperature, with emissions based on the proportional amounts of N denitrified or nitrified using coefficients based on WFPS. Bessou et al. (2010) found that both observed and simulated emissions (based on the field data) of N_2O were significantly lower on non-compacted plots; however, the ratio of N_2O emitted over denitrified N was greater from the non-compacted soil. Therefore, soil compaction and climate conditions were the dominating influence on emissions and prevailed over the N fertilizer rate (Bessou et al., 2010). Laudone et al. (2011) developed a model of the void space for the simulation of denitrification and N_2O emission, which includes parameters relating to microporous and macroporous configuration. This model assumes that biochemical transformations relating to respiration and denitrification are conducted at “hot spots” within micropores, whereas macropores are responsible for conveying reaction products through the simulated network of soil pores (Laudone et al., 2011). This model can be used to estimate CO_2, N_2O, and N_2 emissions from a simulated soil with time, and assess the influence of soil compaction and soil water saturation on those emissions. A reasonably good agreement exists between predicted and observed data (Laudone et al., 2011). Due to the expected spatial variability in N_2O emissions, the reliability of flux measurements in field conditions increases with the number of chambers used within a given experimental site, which is more important than the number of headspace samplings per enclosure period (Chadwick et al., 2014). Sampling at one-day to four-day intervals can provide estimates of cumulative N_2O emissions with a precision of ±10% for more than 80% of the time (Parkin, 2008). If the sampling interval is increased to once in seven days, the probability of achieving a precision of 10% can drop to 20% (Chadwick et al., 2014). These are practical considerations that may be observed when attempting to correlate measured and predicted data.

Estimates of Nitrous Oxide Emission Reduction

Calculating the global potential for reduction in N_2O emissions that might be achieved in the absence of compaction is fraught with difficulty. Nevertheless, a conservative future estimate based on the research presented by Smith and Conen (2004) might be possible. They took data from ten different articles identifying differences in emissions between conventional and NT. Baseline emissions from conventional tillage on crop land averaged across the six countries from which research was examined suggest emissions of around 3 kg ha^{-1} of N_2O-N per crop season. If avoiding compaction by using CTF systems increased the porosity of 80% of the cropped area (assuming permanent traffic lanes occupy the remaining 20% of the area) by 10%, which would relate particularly to the larger drainage pores, the critical issue is how this would impact WFPS. The relationship between WFPS and denitrification is well documented, as well as the close relationship between denitrification and N_2O emissions (e.g., Sexstone et al., 1988; Li et al., 2005b; fig. 3). Using the definition of WFPS (eq. 1; Linn and Doran, 1984), it would make sense to conduct soil compaction studies employing simultaneous measurements of gravimetric water content and soil bulk density, or gravimetric and volumetric water content. Subsequently, these data could be used to identify differences in WFPS due to soil compaction and enable predictions of the temporal risk of N_2O emissions. Smith and Conen (2004), quoting from Smith et al. (1998), indicate that increased emissions of N_2O from NT in terms of CO_2-C equivalent were between 112 and 267 kg ha^{-1} of CO_2-C per year due to increased soil compaction. If it is assumed that switching to NT and compaction avoidance would largely avoid this increase, then the anticipated gain in C sequestration of 350 kg ha^{-1} of CO_2-C per year might be realized.

A review of environmental impacts of CTF conducted by Gasso et al. (2013) indicated the potential of these systems to reduce emissions by 21% to 45% for N_2O and by 372% to 2100% for CH_4, and reduce direct emissions from field operations by 23% compared with non-CTF. For N_2O, such reductions in emissions are of similar magnitude to those measured by Vermeulen and Mosquera (2009) for seasonally controlled traffic in vegetable crop production (range: 20% to 50%). Millar et al. (2010) suggest that N_2O emissions from row-crop agriculture may be reduced by up to 50% by using the N application rate in the lower range of the economic optimum, and indicate that N rate is the main factor influencing direct N_2O emissions. We therefore suggest that N recommendations based on the optimum economic rate be revised for crops established in CTF systems, which may differ from crops in non-CTF. As discussed earlier, this is due to enhanced FUE in CTF compared with non-CTF systems.

Experimental work conducted in southeastern Queensland, Australia, by Tullberg et al. (2011) on a Vertisol sown to winter wheat and fertilized with 80 kg ha^{-1} of N (anhydrous ammonia injected in the interrow) showed that mean N_2O emissions from simulated random wheeling were not significantly greater than those from permanent traffic lanes. However, both these emissions were significantly higher than emissions from permanent non-trafficked beds (fig. 4). Differences in CH_4 flux were significant on only one occasion when it was being absorbed by the permanent bed but emitted by wheeled treatments. In their work, total emissions measured over 42 days post-seeding were converted to CO_2e (GWP = 296 for N_2O, and GWP = 23 for CH_4), indicating total emissions of 57.8 (permanent non-trafficked beds), 325 (permanent traffic lanes), and 370 (simulated randomly wheeled-soil) kg CO_2e ha^{-1}, respectively. This indicates a 42-day post-seeding total CO_2e emission from their CTF grain production system of 90 kg ha^{-1}, that is, 39 kg ha^{-1} from the 12% permanent traffic lanes and 51 kg ha^{-1} from the 88% permanent bed. Such losses represent about 40% of the emissions of 214 kg ha^{-1} likely from a randomly trafficked soil where 50% of the cropped area is tracked. Targeted fertilizer placement should reduce emissions from traffic lane NO_3^--N concentrations. Site-specific N management (including deep placement, e.g., 100 to 150 mm; Liu et al., 2006), improved timing of application, and use of optimum (economic) rate of N should further reduce emissions and improve use efficiency.

This review confirms that soil compaction arising from random traffic patterns by agricultural vehicles has a universally negative outcome; including: (1) increased energy demand for ameliorative tillage, (2) adverse effects on fertilizer use efficiency and therefore on crop yield and potential biomass C returned to soil as residues, (3) increased loss of soil moisture and SOM, and (4) reduced water holding capacity, hydraulic conductivity, and infiltration. Non-organized traffic patterns enhance runoff, erosion, and nutrient transport to water courses, and impair internal drainage and gaseous exchange. The reduction in soil quality

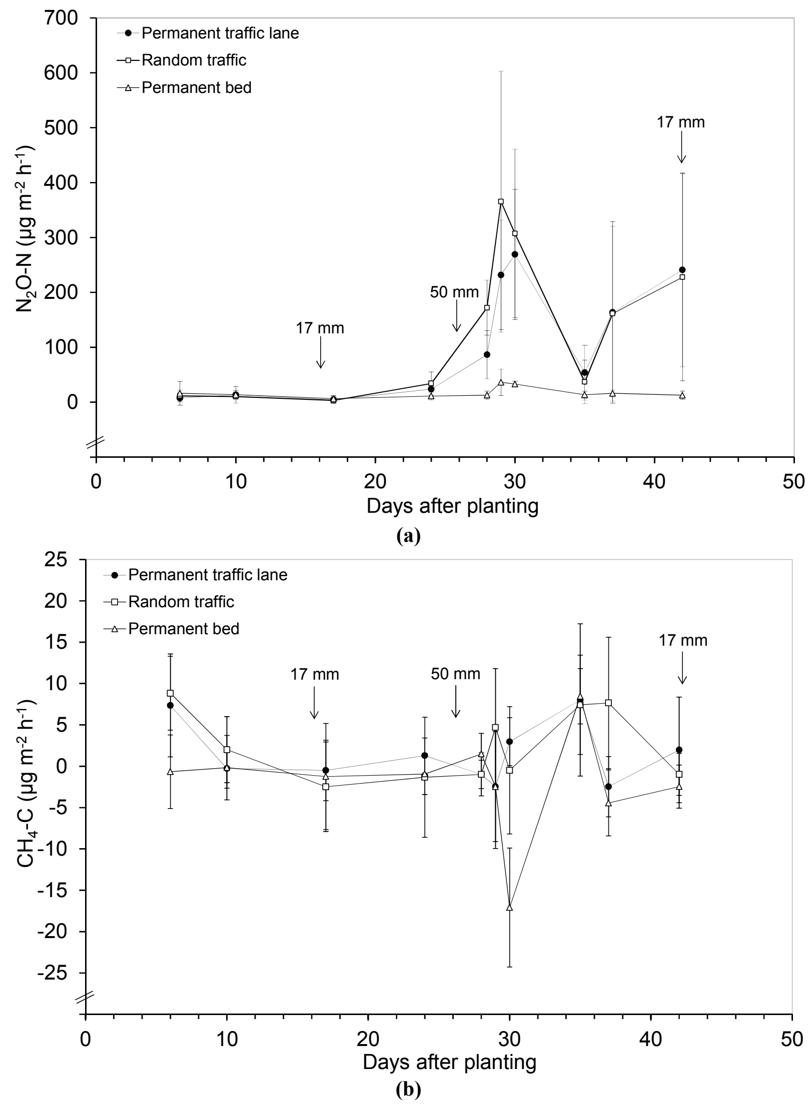

Figure 4. Effect of field traffic on (a) nitrous oxide and (b) methane emissions following establishment of a winter wheat crop in a Vertisol in southeastern Queensland, Australia. Arrows show rainfall events >10 mm. Error bars on mean data points (*n* = 4) denote ±1 standard deviation ($p < 0.05$) (after Tullberg et al., 2011, with permission).

and functions due to random traffic constrain crop yields, add significantly to the cost of crop production, and have adverse effects on the wider environment, including increased potential for GHG emissions. CTF offers an effective means to address these issues through compaction management, which is confirmed by current research in both the United Kingdom and Australia (Smith et al., 2014; Antille et al., 2015b; Chamen et al., 2015; McPhee et al., 2015).

CONCLUSIONS

The main conclusions derived from this review are:

1. An element of the present and anticipated increase in abundance of GHGs from cropped and fertilized soils is associated with reduced oxygen supply caused by traffic-induced soil compaction, which confirms that avoiding compaction is distinctly beneficial in terms of GHG emissions. The likelihood of waterlogging conditions increases with increase in soil compaction because it reduces the size, number, and connectivity of soil pores, which together assist good drainage. N_2O emissions due to denitrification increase significantly when WFPS rises above 60% in the presence of nitrate and organic matter.
2. Confining all load-bearing wheels to the least possible area of permanent traffic lanes should help assuage the considerable concern that reduced tillage and NT systems will increase CO_2e emissions, particularly with respect to N_2O. There may be an elevated risk of N_2O emissions from the relatively small area of permanent traffic lanes (typically <20% of total cultivated area) if these are not designed and managed appropriately, including placement of N fertilizer. Quantification of the benefits of reduced compaction in terms of N_2O emissions may be possible through the use of well-developed models, employed in conjunction with measured contrasts in WFPS between trafficked and non-trafficked soils.
3. NT per se may not be sufficient to maintain and restore soil structural conditions in the presence of random traffic and consequently may not offer opportunities for significant improvements in soil quality in the long term. As a result, agronomic productivity and use efficiency of inputs such as fertilizer and water may not be significantly enhanced. However, soil structural degradation from field traffic may be reversed with NT coupled with CTF and precision agriculture technologies (synergistic effect). These, in turn, can assist in enhancing productivity and soil C sequestration without encouraging release of other GHGs such as N_2O.
4. While acknowledging the inherently low opportunities for long-term C sequestration in arable lands subject to high temperatures and low rainfall, with consequently low biomass production, SOC stocks may be increased by coupling NT with CTF. The major impact of CTF in these circumstances is through its potential for greater cropping frequency (increased rainfall use efficiency), such as double-cropping or inter-cropping to mimic the effects of pasture, and improvement in biologically induced soil mixing by roots and biota. Other accumulations in soil C arise from an enhancement of protected SOM resulting from reduced soil manipulation. Therefore, joint adoption of such practices has the potential to mitigate soil aging, that is, the progressive decline in humified SOC pools. Greater cropping frequency requires careful selection of crops in the rotation and adequate fertilization practices (the "4Rs" principle) to sustain high yields and biomass production, and residue return to soil with high C:N ratio. An important consideration is to ensure that crop residues are not removed from the system.
5. The above conclusions confirm the hypothesis formulated prior to this study and therefore support changes in management practices involving increased adoption of NT and CTF. This approach is proposed as a technically viable, economically feasible, and environmentally sound option to improve soil quality, input use efficiency, and productivity in arable land.

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