Soil carbon and nitrogen dynamics and greenhouse gas mitigation in intercrop agroecosystems in Balcarce, Argentina

by

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AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

Through appropriate soil and crop residue management, soil can function as a sink for carbon (C) and nitrogen (N) for the mitigation of greenhouse gases (GHG). No research has yet investigated the potential of intercrop agroecosystems to reduce emissions of GHG to the atmosphere. This research evaluates whether maize-soybean intercrop agroecosystems sequester more C and N and emit fewer GHG than maize and soybean sole crop agroecosystems. An experiment was conducted at Balcarce, Argentina using four treatments: a maize sole crop, a soybean sole crop, and two intercrops with either 1:2 or 2:3 rows of maize to soybean. The objectives were to quantify soil organic carbon (SOC) and soil total nitrogen (TN) at 0-10, 10-20, 20-40, 40-80 and 80-120 cm depths, rates of decomposition of maize and soybean crop residue after 312 days, crop residue C- and N-input at harvest, and emissions of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Significant decreases in SOC were observed with depth in all treatments after 40 cm, and significant decreases in TN were observed with depth in all treatments after 20 cm. Crop residue from maize had the greatest input of C and N to the soil, but the slowest rate of decomposition. Soybean biomass had the least input of C and N to the soil and the fastest rate of decomposition. The 1:2 and 2:3 intercrop agroecosystems had moderate crop residue inputs of C and N and intermediate rates of decomposition. No significant differences in GHG emissions were detected between treatments throughout the growing season. The major influences on GHG emissions were weather events, soil temperature and moisture, and crop residue input. Annual GHG emissions were determined; the CH_4 sink in the 1:2 intercrop and the soybean sole crop was significantly greater ($P < 0.05$) than the 2:3 intercrop and the maize sole crop. Emissions of CO_2 were inversely proportionate to N_2O, with the greatest C sink in the 1:2 intercrop.

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Chapter 1 – General Introduction

1.1 Climate Change

Important global issues of the 21st century include food insecurity, water scarcity and eutrophication, soil degradation and desertification, energy scarcity and biofuels, waste management, and climate change. There is a direct link between these global issues and the sustainable use of the world's finite resources (Lal, 2007). Climate change refers to the effect of human activities on the climate. Gases such as carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are emitted to the atmosphere, and absorb infrared radiation from the earth's surface, thus keeping it warmer than it would otherwise be (Houghton, 2005). Currently, increases in atmospheric concentrations of these gases are causing changes in precipitation and shifts in the natural ranges of plants and animals. The global average temperature is approximately 0.8 °C above its pre-industrial level (IPCC, 2000). A recent measurement of the change in heat content of the oceans to a 3 km depth from 1955 to 1998 confirms that the oceans are warming (Houghton, 2005).

Ice cores, like other evidence of climate conditions in the distant past (e.g. tree rings and corals) show that rising atmospheric CO_2 levels are associated with rising global temperatures (IPCC, 2000). An even closer correlation is found with CH_4 concentration (Houghton, 2005). Human activities, primarily the burning of fossil fuels, and secondly changing land-use, have increased the concentration of CO_2, CH_4, N_2O and other heat-trapping gases in the atmosphere. There is an international scientific consensus that most of the change in climate observed is attributable to human activities (Arctic Climate Impact Assessment, 2004). Temperature changes are projected to create shifts in atmospheric and oceanic circulation patterns, an accelerated rate of sea-level rise, and wider variations in precipitation, leading to wide-ranging consequences including impacts on coastal communities, animal and plant species, and water and land resources (Arctic Climate Impact Assessment, 2004).

1.2 Greenhouse Gases

Globally, CO_2 accounts for about 60% of gases contributing to global warming potential (GWP). Other gases contributing to the current global warming trend are CH_4 (15%), and N_2O (6%). These gases are long-lived in the atmosphere and increases in their concentrations are expected to lead to a general rise in global temperature (FAO, 2001; IPCC, 2001). Carbon dioxide, CH_4, N_2O, water vapour, ozone (O_3) and chlorofluorocarbons (CFC) are collectively known as greenhouse gases (GHG). The presence of GHG is a natural and critical part of Earth's atmosphere. These gases influence radiation balance by permitting the short wave radiation to enter the Earth's atmosphere and capturing a fraction of the long wave radiation emitted back by the Earth (Seguin et al., 2006). This is known as the greenhouse effect because the glass in a greenhouse similarly absorbs infrared radiation, an effect first recognized by the French scientist Jean-Baptiste Fourier in 1827 (Houghton, 2005). The balance between the incoming solar radiation and Earth's outgoing thermal radiation holds a certain amount of heat at the earth's surface, keeping temperatures warm enough to sustain life. However, the continued increase in atmospheric GHG has disturbed this balance. The enhanced greenhouse effect arises because the increase of GHG in the atmosphere absorb the Earth's emitted thermal radiation and act as an insulating layer (Houghton, 2005). The imbalance in radiative forcing may be expressed as the change in net irradiance at the tropopause (Ramaswamy, 2001). The current radiative forcing of these gases is 1.46 W m^{-2} for CO_2, (GWP = 1), 0.5 W m^{-2} for CH_4 (GWP = 21), and 0.15 W m^{-2} for N_2O (GWP = 310) (IPCC, 2000).

The concentration of several GHG have changed drastically since the industrial revolution because of fossil fuel combustion, cement manufacturing, land use change, and the associated agricultural practices, such as plowing, residue burning, use of fertilizers, and manure. The concentration of CO_2 has increased by 35% from 280 ppm in 1750 to 377 ppm in 2004 and is presently increasing at the rate of 1.8 ‰ or 0.47 % per year (Follett et al., 2005; Lal et al., 2007). Principal sources of CO_2 emissions are fossil fuel combustion (7 Giga ton per year) and deforestation (0.6 – 2.5 Giga ton per year) (IPCC, 2000). Since 1958, about 55% of the CO_2 emitted by fossil fuel combustion has been stored in the atmosphere (Lal et al., 2007).

The concentration of CH_4, responsible for 20% of the radiative forcing of the Earth, has increased by 155% from about 700 ppb in 1750 to 1783 ppb in 2004 and is presently increasing at the rate of 5 to 13 ppb per year (Lal, et al., 2007). Although CH_4 has about 21 times the GWP of CO_2, and agriculture may be responsible for about 27% of total CH_4 emission, very few studies have involved field crop management impacts on CH_4 (Jawson et al., 2005). Tillage intensity can affect a range of biochemical properties but there is considerable uncertainty with regard to the impact of soil management practices on CH_4 uptake in agroecosystems (Omonode et al., 2007). Principal anthropogenic sources of CH_4 include fossil fuel exploitation, rice paddy cultivation, ruminant animals, residue burning, and landfills.

Nitrogen (N) is ubiquitous in the environment. It is required for the survival and development of all living organisms. It is one of the most important nutrients, essential for protein synthesis in plants, animals, and microorganisms (Follett et al., 2005). The atmospheric concentration of N_2O, responsible for about 6% of the radiative forcing of the Earth, has increased by 18% from about 270 ppb around 1750 to 318.6 ppb in 2004 and is presently increasing at the rate of 0.8 ppb per year (Lal et al., 2007). Anthropogenic sources of N_2O include fertilizer use, residue burning, and industrial processes. Agricultural practices account for most of the increase of N_2O emissions in the atmosphere during the past century (Lal et al., 2007; Schlesinger, 2000).

Because many countries have set targets for reduction of a suite of GHG, including CO_2, CH_4, and N_2O, evaluating the potential of C- and N-sequestration programs requires consideration of the consequent change in emissions of other GHG. For example, Abu-Khader (2006) suggests that the removal of CO_2 may cause an increase in emissions of other GHG such as N_2O. Li et al., (2005) showed that microbial production of N_2O is tightly linked to C availability; therefore C sequestration strategies affect the production of N_2O. The positive correlation between C and N_2O flux arises from the coupled biogeochemical cycles of C and N (Li et al., 2005). Thus markets emerging for the trading of GHG emissions credits will require full accounting of all gases in order to be effective. Different GHG can be compared on a common basis by converting emissions values of non-CO_2 GHG into CO_2-equivalents using GWP (Li et al., 2005).

1.3 Land-use Change

Terrestrial ecosystems are important components in the biogeochemical cycles of carbon (C) and N that create sources and sinks of CO_2, CH_4, and N_2O and thereby influence emissions of GHG. Land-use change is an alteration in the way land is used, including conversion of forests and grassland to pasture or cropland. Terrestrial ecosystem processes, particularly C and N cycles, may be modified by changes in climate and by land-use and land-cover change (IPCC, 2000). Many areas are altered for food and wood production, human habitation and recreation. These changes affect the amount of C and N stored in vegetation and soil by changing aeration, water and temperature dynamics, aggregation, as well as the quantity of crop residues returned to soil (Grandy and Robertson, 2007; IPCC, 2000).

The vulnerability of macroaggregates to destruction following tillage intensification and substantial shifts of C from physically protected slow pools into active pools following aggregate destruction demonstrates the need to protect stabilized SOM from tillage (Grandy and Robertson, 2007). Under undisturbed natural conditions, the soil C and N pool is in equilibrium, and the input of C and N (litter fall, root residue, C brought in by run-on, dust) is balanced by output (erosion, decomposition, and leaching) (Lal, 2004). Conversion of natural to agricultural ecosystems, however, typically reduces the amount of input and increases the magnitude of output. The reduction in input is caused by a decline in crop residue production and reduction in the fraction returned to the soil. The increase in output is attributed to increase in oxidation of SOM because of change in soil moisture and temperature , and increase in losses caused by soil erosion and leaching. Losses of soil C and N occurs through respiration, oxidation, nitrification and denitrification, resulting in emissions of CO_2, CH_4, and N_2O to the atmosphere through NO_x, ammonia (NH_3) and nitrogen gas (N_2 (FAO, 2001). Many cultivated soils have lost 50% to 75% of their initial SOC pool (Lal, 2004). In general, 60% to 75% (about two thirds) of the C lost can be re-sequestered through adoption of recommended management practices (Lal et al., 2007).

Grassland ecosystems comprise approximately 30 – 50% of terrestrial land-use in temperate regions (Müller and Sherlock, 2004). Disruption of the plant-atmosphere-soil equilibrium and emission of GHG from soil-related processes began with the onset of settled agriculture (Lal et al., 2007). The net release of CH_4 is directly related to the cultivation of rice paddies and domestication of animals (Lal et al., 2007). During the last four decades, agricultural land gained almost 500 million hectares from other land uses, a change driven largely by increasing demands for food from a growing population. Every year during this period, an average 6 million hectares of forestland and 7 million hectares of other land were converted to agriculture, a change occurring largely in the developing world (Smith et al., 2007). This trend is projected to continue into the future. In Argentina, agriculture has become increasingly dominant over mixed crop and animal systems, as large areas of permanent pastures are being converted to cropland. Soil C losses due to cultivation and tillage of virgin soils have been reported up to 55% after only 5 years of cultivation (Noellemeyer et al., 2008). Land-use change in the region from permanent pasture to arable agriculture has greatly affected SOC stocks and other physical and biological soil properties, such as decreases in structural porosity and infiltration rate and increased bulk density(Noellemeyer et al., 2008).

1.4 Sustainable Agriculture and Food Security

The rate of increase in global crop yields are projected to decrease, especially in developing countries where natural resources are under great stress because of soil degradation that may be exacerbated by projected climate change (Lal, 2006). Over the last three decades, world food production has grown faster than its human population. As a result, prices for many major crops declined when adjusted for inflation (Easterling and Apps, 2005). In developing countries, a lack of access to food, political instability and inadequate resources have lead to an estimated 1.3 billion malnourished people (Easterling and Apps, 2005). In order to reconcile an ample food supply with farmer income and reduced impacts to the environment, sound agricultural practices will need to be adopted (FAO, 2001). This will require increases in the productivity of existing land through restoration of degraded soils and improvement in soil quality. Farmers who have sufficient access to capital and technologies may be able to adapt their agricultural systems to these projected changes, including changes in the types of crop and livestock produced. Considerable costs could be involved in this process, such as investments in the dissemination of information, and gaining experience with different crops or irrigation (Easterling and Apps, 2005). Nations with large resource endowments (such as developed countries) will fare better in adapting to climate change than those with poor resources (such as developing countries). This could increase disparities in income between developed and developing countries.

Over the next decades, global food demand is expected to increase. The challenge is how to maintain long-term sustainability, given the current degradation in the natural resource base. Sustainable resource use is of particular importance in developing countries from where approximately 80% of the increase in food demand is expected to come (Pretty et al., 2002). For example, the percentage of people depending primarily on natural resources for their livelihoods (soil, rangeland, forestry products, fish) is higher in Africa than anywhere else (Dar and Twomlow, 2006). In developing countries, there are an estimated 800 million people lacking adequate access to food, of which 31% are in east and south-east Asia, 31% in South Asia, 25% in Sub-Saharan Africa, 8% in Latin America and the Caribbean, and 5% in North Africa (Pretty et al., 2002). It is therefore necessary to find ways to enhance productivity and production without draining the stock of natural capital. Forests, rangeland and farming systems must be conserved and protected to ensure sustainable agricultural growth (Dar and Twomlow, 2006). Traditional rural production systems are generally sustainable under conditions of low population pressure and lack of market integration, when system productivity is geared towards subsistence. These systems remain in sustainable equilibrium until changes such as population growth or external economic pressures occur at a rate faster than can be sustained without resource degradation (Dar and Twomlow, 2006). These changes can cause an intensification of agriculture into marginal lands, where the risk of crop failure, environmental degradation, erosion and loss of biodiversity increases due to inappropriate management practices that can exhaust the soils of nutrients and organic matter. Unfortunately, incentives to pursue environmentally sustainable practices are commonly lower than incentives to simply extract natural resources (Dar and Twomlow, 2006). Sustainable agricultural systems produce food and other market goods, but also contribute to a range of valued public goods, such as clean water, wildlife, C and N sequestration in soils, flood protection and groundwater recharge (Pretty et al., 2002). However, farmers may not adopt otherwise unprofitable agricultural mitigation practices in the absence of policies or incentives (Smith et al., 2007). For example, reductions in emissions of CH_4 and N_2O can be converted to CO_2-equivalents, and traded. Trading C credits can provide another income stream for farmers, and provide the much needed incentives to invest in soil improvements (Lal, 2008).

The world population is projected to reach over 9 billion by 2050 and 10 billion by 2100 (Follett et al., 2005; Lal, 2008). All of this increase is projected to occur in developing countries (Lal, 2008). These are also the regions where soil resources are limited, fragile, and prone to degradation by the projected climate change and the increase in pressure from the projected population increase (Lal, 2008). Agriculture, implemented properly, is an important solution to the issue of achieving global food security but also of improving the environment.

Global agricultural vulnerability is assessed by the anticipated effects of climate change on food prices. Recent advances in modeling of vegetation response suggest that transient adverse effects associated with ecosystems responding to climate change will dominate over the next century (Easterling and Apps, 2005). Agriculture occupies a larger portion of global land area (about 38%) than any other human activity (Betts, 2006). Because of its scale and intensity, agriculture emits a large quantity of greenhouse gases into the atmosphere (Lal, 2006). It presently accounts for about 25% of the CO_2, 50% of the CH_4 and 70% of the N_2O released globally via human sources. Based on evidence of modeling studies, a global temperature rise of greater than 2.5 °C is likely to stress food security in many developing countries (Easterling and Apps, 2005). Lal et al. (2007) speculated that climate change may enhance mineralization in soils with a positive feedback. It is estimated that an increase in temperature would cause a shift of vegetation and a decrease in the soil organic carbon (SOC) pool which would adversely affect soil quality. Because farmlands are intensively managed, farmers can control the amounts of SOC and N to some extent (Hutchinson et al., 2006).

1.5 Soil Organic Matter and Soil Quality

Soil organic matter (SOM), a long-lasting source of nutrients, is a matrix with a high cation exchange capacity (CEC), and is a vital part of soil structure. Because SOM has a very complex and heterogeneous composition, and is associated with mineral constituents in the soil, it provides a large storage capacity for nutrients and the removal of pollutants through complexes with metal ions (Berg and Laskowski, 2006; FAO, 2001). In addition, stabilized SOM, or humus, has a dominant role in the global carbon balance. The estimated amount of C stored in SOM substantially exceeds all C stored in living organisms (Berg and Laskowski, 2006).

Management of SOM and humus is essential to sustaining the productivity of the soil ecosystem. Soil organic matter provides plant nutrients and refers to the fraction of the soil that is composed of living organisms and once-living residues in various stages of decomposition. Humus is the end-product of SOM decomposition and is relatively stable because further decomposition occurs very slowly (Desjardins et al., 2005). In natural systems, a balance is reached between the amount of humus formation and the amount of humus decay. This balance also occurs in most agricultural soils, but often at a much lower level of soil humus (Sullivan, 2003). Humus contributes to soil that produces high-quality plants. The benefits of a topsoil rich in organic matter and humus include rapid decomposition of crop residues, granulation of soil into water-stable aggregates, decreased crusting and clodding, improved internal drainage, better water-infiltration, and increased water and nutrient holding capacity (Sullivan, 2003). Improvements in the physical structure of the soil reduce erosion and facilitate tillage, water storage capacity, and deeper, more prolific plant root systems (Sullivan, 2003). In general, the soil moisture content increases by 1 to 10 g for every 1 g increase in SOM content, which helps to maintain crop growth between periods of rainfall (Emerson, 1995). Enhancement and preservation of SOM enhances CEC, increasing the ability of the soil to retain nutrients, and thus microbial activity.

Soil organic matter is closely related to SOC because SOM constitutes the largest terrestrial reservoir of C (Abu-Khader, 2006; Blanco-Canqui and Lal, 2004; Bongen, 2003; IPCC, 1999). Soil structure and SOM help determine the extent of the SOC pool (Blanco-Canqui and Lal, 2004). Stable soil structure stores SOM and prevents rapid decomposition. Soil organic matter contains approximately 55% SOC and 45% other essential elements (Berg and Laskowski, 2006, Blanco-Canqui and Lal, 2004). Management practices to build up SOC must increase the input of organic matter to soil and decrease decomposition rates. In temperate regions, key strategies involve reducing bare fallow, increasing the use of perennial forages in crop rotations, retaining crop residues and reducing or eliminating tillage (Paustian et al., 1997). Practices that maintain SOM and increase soil productivity can improve global or regional food security (Lal, 2004b). Increased productivity may become more important as global food demands increase in coming decades (Smith et al., 2007).

Increasing the SOC and N pools of degraded soils would increase crop yields by increasing available water capacity, improving the supply of nutrients and enhancing the soil structure and other physical properties (Lal, 2006). There is a strong relationship between the SOC pool and soil fertility. Indeed, C sequestration is a beneficial strategy for farmers. With approximately 800 million food-insecure people, improving soil quality is essential to advancing food security. Increasing the SOC pool, which is a challenging task for soil scientists and land managers, can increase production of food grains (Lal, 2006). Increasing SOC, challenging as it may be, can help to meet current and projected food deficits in the developing world (Lal, 2007).

Soil organic matter represents a key indicator for soil quality, both for agricultural and environmental functions. Soil organic matter is the main determinant of biological activity; the diversity and extent of activity of soil fauna and microorganisms are directly related to SOM. Biological activity, and the SOM upon which it depends, have a major influence on the physical and chemical properties of soils. Aggregation and stability of the soil structure increase with SOM content. This in turn increases infiltration and available water capacity of the soil, as well as resistance against erosion by water and wind. Soil organic matter also improves the availability of essential plant nutrients (FAO, 2001).

1.6 Carbon and Nitrogen Sequestration

Sequestration of C and N in SOM and terrestrial biomass removes C and N from the atmosphere until the maximum capacity for the ecosystem is reached, which may take decades, centuries or millennia, depending on the management practices and the type of system (Smith et al., 2007). However, changes in land-use or in management may reverse the gains of C and N sequestration over a similar period of time. Sequestration is a rapidly and cheaply deployable GHG mitigation technique until a change in human use of resources or other, more permanent solutions become available (Smith et al., 2007).

The global C cycle has four pools: atmospheric, biotic, pedologic, and oceanic (FAO, 2001; Janzen, 2005). The atmosphere contains about 800 Pg C. Terrestrial biota contains about 500 Pg C, mostly in trees. Soils, in SOM, contain about 1500 to 2000 Pg C to a depth of 1 m, and even more deeper in the profile. The largest pool is the oceans, containing about 3900 Pg C, mostly at great depths and isolated from surface processes. These pools are dynamic, with C continually flowing within and among them (Janzen, 2005).

Carbon and N sinks constitute a part of the biogeochemical cycle where C and N may be stored for long periods of time (Bongen, 2003). Many reviews have been published on the importance of oceanic, geological and biological sinks, such as forests and agricultural lands, of GHG. The deep ocean is a prospective C sink because it is extremely vast and highly unsaturated in CO_2. The world's oceans are estimated to absorb 2.3 Pg atmospheric C per year and another major sink is in terrestrial ecosystems (IPCC, 2001). Underground geological storage is technically feasible, and CO_2 may be retained in reservoirs for millions of years. Soil is the largest terrestrial sink for C (Bongen, 2003; Abu-Khader, 2006). In the past few years, the focus of much research has been on intensively managed croplands, because implementation of new conservation and production practices can increase SOM and thus create sinks for C and N (Follett et al., 2005). For example, Regina et al. (2007) and Henckel et al. (2000) identify soil as the only known biological sink for CH_4. Soils are also the largest contributors of N_2O, with 6.0 Tg yr^{-1} from natural soils and 4.2 Tg yr^{-1} from agricultural soils (Chapuis-Lardy et al., 2007; FAO, 2001). The highest potential for SOC sequestration is in nutrient depleted and degraded soils of the developing countries (Lal, 2003). Agriculture is one of the sectors that could reduce the extent to which fossil fuel emissions need to be reduced by enhancing the C and N sink in soil (Lal, 2004; Desjardins et al., 2005; Kong et al., 2005).

World soils constitute the third largest global C pool (organic and inorganic to a 1 m depth), which is about 3.3 times the atmospheric pool and 4.5 times the biotic pool (IPCC, 1999). Globally, the C sink capacity of soils is about 1 Pg C/yr, which can annually offset 0.47 ppm of CO_2 in the atmosphere (Lal, 2007). Different C pools in the soil have different residence times, ranging from less than one year to hundreds of years depending on the biochemical composition (Table 1.1:).

Sequestration of C and N can be physical or chemical; physical refers to an encapsulation of organic matter fragments by clay particles or soil macro- or microaggregates and chemical refers to specific bonds of OM with other soil colloids or clays (FAO, 2001).

Table 1.1: Residence time of C in different pools (adapted from Houghton, 2005; IPCC, 2001).

Carbon Pool	Residence Time
Atmospheric CO_2	~ 4 years
Vegetation	
Non-woody	Months to years
Woody	Years to centuries
Soil	
Soluble C	Hours to months
Plant litter	Months to years
Organic matter	Years to decades
Humus	Decades to millennia
Fossil C	Many millenia

The ability to sequester C and N in soil depends on many factors, including climate, soil type, vegetation, and management practices. Climate influences C and N sequestration because the rate of residue decomposition is a function of temperature and moisture (Dornbush and Raich, 2006). Decomposition or mineralization is the process in which elements are converted from organic to inorganic forms. Mineralized elements are then available for plant uptake or microbial use (Colemean et al., 2004). The incorporation of these materials into the soil via soil organisms, and exudates from roots (leaching and fine root turnover) that are utilized by microbial populations, constitute the natural pathways of incorporating residue C and N into the soil (Lal, 2004). The decomposition rate increases with temperature, while it decreases with increasingly anaerobic conditions (Oelbermann et al., 2008). Indirect climate effects are mediated via vegetation or soil faunal activity (Blanco-Canqui and Lal, 2004).

Models of litter decomposition indicate that in both temperate and boreal systems, litter does not decay 100%, but to a certain limit, depending on litter type, and can be described by an asymptotic function (Berg and Laskowski, 2006). The limit value for decomposition therefore lies between 50 and 100% mass loss, indicating that between 0 – 50% of the litter remains as a recalcitrant part (Berg and Laskowski, 2006). This remaining litter may remain in soils for centuries or millennia without further degradation (Berg and Laskowski, 2006).

Carbon sequestration in croplands is controlled by a balance between the C in residues returned to soil and that released into the atmosphere (Bronick and Lal, 2005). In ecosystems where the organic matter is decomposed virtually as quickly as it is shed as litter, such as humid tropical environments, the net oxygen (O_2) production is close to zero. In such a case, not much more O_2 is produced during photosynthesis than is used in respiration, and thus not much more CO_2 is assimilated than is produced during organic matter oxidation (Berg and Laskowski, 2006). In temperate regions, crop residue input is greater than the rate of decomposition, allowing for the long-term accumulation of C in the soil (Berg and Laskowski, 2006). Cropping and tillage systems that promote the return of residues to the soil increase the SOC pool (Blanco-Canqui and Lal, 2004).

Soil type also affects C and N sequestration capacity. Finer textured soils generally have higher SOC contents than coarse textured soils, determined by clay and silt content (Ingram and Fernandes, 2001). In a study of temperate and tropical mineral soils to quantify the relationship between soil texture and SOC, Hassink (1997) reported that as the upper limit for the adsorption of organic inputs to clay and silt is reached, adding more organic material to the soil does not lead to increased C sequestration. A close relationship was noted between the proportion of soil particles smaller than 20 µm and SOC. Soil organic carbon associated with the smaller clay particles was better protected against decomposition (Hassink, 1997). In addition, soils with high clay and silt content facilitate the formation of micro- and macroaggregates which can further protect SOC (Blanco-Canqui and Lal, 2004). Urioste et al. (2006) reported that because of cultivation, accumulation of organic carbon and organic and inorganic phosphates was lowest in fine aggregates (<100 µm) of fine-textured soils and in coarse aggregates (100–2000 µm) of coarse textured soils.

The ability to sequester C and N in the soil depends on climate and soil type, but is essentially limited by the residue input to the system. Net primary productivity is the underlying control, and depends on vegetation type (Ingram and Fernandes, 2001). In a 15-year experiment in the Argentine Pampas, Conti et al. (1997) found that substrate type was the regulating factor in mineralization. The ratio of C/N of plant residues determines the effectiveness of organic material for improving soil structure and thus enhancing C sequestration. Organic residues with higher C/N ratio persist for a longer time and improve soil aggregation, whereas residues with low C/N ratio are rapidly decomposed and have reduced impact on soil structure improvement. For example, soybean (*Glycine max* L.) residues have lower C/N ratio and are more rapidly decomposed than maize (*Zea mays* L.) residues. Rapidly decomposing residues quickly enhance the formation of aggregates, but their action is transient, whereas slowly decomposing residues have a more gradual impact on aggregation, but their long-term effect on SOC sequestration is higher (Blanco-Canqui and Lal, 2004).

Prudently managed, world soils can help achieve food security and also mitigate climate change by absorbing atmospheric CO_2 and converting it into humus. Slowing the rate of increase of the concentration of GHG to the atmosphere is expected to require efforts from many sectors of the economy. Despite the enormous challenge of C and N sequestration, especially in regions of predominantly resource-poor farmers, it is truly a win-win strategy. While improving ecosystem services and ensuring sustainable use of soil resources and food security, SOC sequestration also mitigates global warming by offsetting fossil fuel emissions, improving water quality, and reducing non-point source pollution (Lal et al., 2007).

1.7 Crop Diversity and Complex Agroecosystems

The general focus of research on biodiversity has been at the ecosystem and global level (Mooney and Chaplin III, 1994). Recent concern focuses on the status of biodiversity at multiple levels of integration – from genes to landscapes – and how the diversity of organisms influences the ecosystem (Mooney and Chaplin III, 1994). Species within a community affect different ecosystem functions and global processes such as productivity, nutrient cycling, C and N fluxes to the atmosphere, and water availability. The use of increased diversity is a potential step towards sustainability in agriculture. Many of the problems associated with intensive, high input agriculture would benefit from increased diversity for greater resilience to environmental and productivity demands of the future (Jackson et al., 2007).

Increases in productivity of agricultural land in the past can be attributed to such practices as increased use of fertilizers, irrigation, and development of high-yield producing varieties of crops (Jackson et al., 2007). However, new solutions are necessary to produce more food without exploiting the natural resource base, upon which agriculture depends. Overall, the major trend in agriculture is one of reducing complexity (Vandermeer et al., 1998). However, adoption of biodiversity-based agriculture has been proposed as a means to improve sustainability without compromising environmental quality (Table 1.2).

Table 1.2: Farming practices that augment biodiversity in complex versus conventional agroecosystems (Adapted from Shiyomi and Koizumi, 2001; Vandermeer et al., 1998).

Complex Agroecosystems	Conventional Agroecosystems
Intercropping	Sole cropping
Agroforestry	Sole cropping, tree plantations
Alley cropping	Sole cropping
Minimum- or no-till	Conventional plowing
Rotation with legumes or other crops	Monoculture
Mulching	Bare soil
Hedgerows	Wild vegetation removal, open fields
Field margins	Large, open fields
Organic fertilizer	Chemical fertilizer
Mosaic landscape structure	Landscape simplification, woodland clearance

Intensive agriculture is considered to be the main force responsible for the fragmentation of grasslands and the decline in plant species. Effective conservation of resources requires the coexistence of species (Shiyomi and Koizumi, 2001). Mungai and Motavalli (2006) showed that alternative farming systems, such as complex agroecosystems, may help to reduce C and N losses from the system. Complex agroecosystem arrangements may include multiple cropping or sequential cropping. However, these are distinguished from intercropping because crops may be planted on a rotational basis, may occur during only a part of the growing season, or only cover a part of the planting area (Geno and Geno, 2001). Jurik and Van (2004) describe strip cropping, which is commonly used in the Midwestern states. This involves narrow strips of 4 or 5 rows of crops planted adjacent to each other. Maize and soybean are the major crops used in this agroecosystem, which is economically viable, and has soil conservation and aesthetic benefits (Jurik and Van, 2004). Geno and Geno (2001) define intercropping as the growing of two or more crops on the same field simultaneously. This is fundamentally different from other types of complex agroecosystems because crop intensification is both temporal and spatial; there is crop interaction during all or part of the season, and both crops must be managed at the same time. Complementarities between crops arise 1) spatially; differences in canopy and root dispersion, and 2) temporally; crops make their major demands on crops at different times (Geno and Geno, 2001; Willey, 1990).

Introducing a complex arrangement in an agroecosystem may help to decrease dependency on fertilizers. Research on the yield of maize and soybean across a range of geographical locations showed that intercropped maize yielded more than the corresponding sole crop at 0% applied N (Ahmed and Rao, 1982; Chui and Shibles, 1984). In sub-Saharan Africa, farmers are encouraged to enrich their soils by planting woody and herbaceous species simultaneously with the crops (Makumba et al., 2006). These species in agroforestry systems can increase nutrient content in the topsoil by enabling nutrient cycling from the subsoil, or from N_2 fixation (Makumba et al. 2006). Some research suggests that competition between trees and crops in a hedgerow cropping system may cause competition for resources (light, water, nutrients) and will reduce yields. However Makumba et al. (2006) found that a *Gliricidia sepium* − maize intercrop system developed at Makoka, Zomba, in Malawi was more successful than typical hedgerow intercropping.

1.7.1 Intercropping

Through the domestication of crops and animals, humankind has greatly reduced the natural level of biodiversity. Sole crop agricultural systems are a classic example of this. By planting mixtures of different crops, farmers can restore diversity to their agroecosystems, and realize many advantages over traditional sole crop systems (Sullivan, 2003). Intercropping, which is the planting of two or more crops on the same area of land at the same time, is one way farmers may increase the complexity of their agroecosystems (Sullivan, 2003). The advantage of intercropping results from the complementary use of growth resources – water, light, nutrients – over time and space, which may increase productivity per unit of land (Ahmed and Rao, 1982; Prasad and Brook, 2005; Sullivan, 2003). Biophysical reasons for intercropping include better utilization of environmental factors, greater yield stability in variable environments and soil conservation practices. Socio-economic reasons include a decrease of inputs, an increase of outputs and the consequent stabilization of household food supply (Tsubo et al., 2003).

The main reason for using an intercropping system is that by integrating crops, space and resources are used more efficiently (Ahmed and Rao, 1982; Boucher, 1986; Prasad and Brook, 2005; Sullivan, 2003; Tsubo et al., 2003). Tsubo et al. (2003) reported that radiation and water-use efficiency is greater in plants grown as intercrops than as sole crops. For example, cereal-legume agroecosystems utilize soil and water resources better than sole crop systems due to the combination of the tall cereal with an adventitious root system and the short legume with a deep tap root system (Prasad and Brook, 2005; Tsubo et al., 2003). Boucher (1986) suggested that intercropping is advantageous over sole cropping due to an efficient use of scarce resources, and is thus often used by small scale farmers with little access to input and poor soils. Both cereals and legumes are dietary staples and complementary crops. Canopy structures and root systems of cereal crops are distinct from those of legume crops. The formative rate is comparatively greater in cereal crops than in legume crops, such that competition for resources arises at different times. In addition, the leguminous component has the ability to fix atmospheric N, thus avoiding competition with the cereal for this nutrient (Prasad and Brook, 2005).

Residues of legumes are high in labile organic material, which increases soil aggregation and SOC concentration. This effect is often transient because the labile fraction is easily degraded (Blanco-Canqui and Lal, 2004). Thus it is important to integrate a leguminous crop with a recalcitrant, high C/N content crop such as maize. In theory, legume-based intercropping can reduce C and N losses from cultivated land (Austin et al., 2006). The incorporation of low C/N litter in SOM from leguminous crop residues has been shown to increase C and N retention in temperate agroecosystems (Drinkwater et al. 1998). At present, the global use of fertilizers is 78 million tons of N per year, although the use of N fertilizers by plants is quite inefficient; only approximately 50% of the applied N is assimilated (FAO, 2001). The net result is the loss of N from the soil-plant system via leaching, erosion, or gaseous emissions. Use of legumes in crop rotation can also appreciably reduce the requirements for N fertilizers for various cropping systems, thereby reducing net fossil fuel requirements and the C cost of manufacturing N fertilizers (Hutchinson et al., 2006; Zentner et al., 2001).

In agricultural systems, because vegetation is the primary influence on the characteristics of soil N, crop and soil management will have a profound effect on N dynamics (Burket and Dick, 1997). Organic C levels are closely correlated with the amount of plant C inputs, and in general, N-rich residues like legumes will cause a greater accumulation of N than non-N_2 fixing plants (Burket and Dick, 1997). Legumes return large amounts of crop residues to the soil, and can potentially increase SOC, thus increasing the likelihood for sequestering atmospheric C (Zentner et al., 2001).

Several studies found increases in soil C with the incorporation of legumes, which was attributed to the effect of legume residue on microbial communities (higher activity and soil microbial biomass (SMB) C), the production of polysaccharides, and aggregate stabilization (Grandy and Robertson, 2007; Dick, 1992). Drinkwater et al. (1998) showed that maize cropping systems which incorporated soybeans increased soil C content that was derived mostly from C_3 sources.

The key to sustainable agriculture lies in increased grain yield per unit area and a reduction in agricultural land expansion. As opposed to the typical rotational sole cropping systems used in developed countries in North America and Western Europe, intercropping systems used in developing countries in Africa, Asia and Latin America may help increase food productivity (Tsubo et al., 2003). Intercropping is of particular interest to smallholder farms, where there is greater need to make efficient use of limited space, or where supplies of external inputs are poor due to limited resources (Prasad and Brook, 2005). In Nepal, approximately 80% of the maize crop is grown in an intercropping system to make efficient use of the short rainy season (Prasad and Brook, 2005). In India, in order to utilize resources more efficiently, *Sorghum bicolor* (L.) − *Cajanus cajan* (L.) (pigeonpea) intercrops are combined as grain sorghum is a fast-growing, early-maturing crop and pigeon pea is a slower-growing, later-maturing crop (Willey, 1990). In the densely populated areas of southern Ethiopia, for example, maize-*Coffea arabica* (L.) (coffee) intercropping systems are the dominant agricultural production practices that have evolved in response to growing pressure on land and agro-climatic risks. These systems have been highly intensified over the years in response to high population pressure and the consequent shortage of arable land (Alene et al., 2006). Interest in the role of annual legumes in the small land holder cropping systems based on maize in southern Africa has increased during the last 15 years as efforts to develop and test sustainable soil fertility improvement options for these systems have expanded. Grain legumes are used both as a food source and for the improvement of soil fertility. The legumes derive most of their N needs from biological N-fixation, and produce a substantial amount of both grain and residue (Waddington et al., 2007).

Legume–maize intercropping enables farmers to produce nutritious foods from the legumes, while maintaining production of the maize staple each year. However, compared with sole crops in rotation, intercropped grain legumes are planted at low legume plant population densities in order to minimize resource competition, and so produce small amounts of dry matter per hectare, fix less N per ha and generally leave little N in the soil (Giller, 2001; Mafongoya et al., 2003). This means that there is less expectation of a significant contribution to soil fertility or of a subsequent improvement in maize yields with the intercrop. While there is often a large increase in maize yield in rotation where maize follows a sole crop grain legume, when intercropped the N and organic matter inputs from grain legumes, including soybean and *Apios americana* (L.) (groundnut), and the benefits to the grain yield of subsequent maize crops are reported to be much smaller in many smallholder farming situations (Waddington et al., 2007).

There are no long-term studies of the effect of grain legume-maize intercrops on crop productivity and its sustainability (Waddington et al., 2007). Additionally, large increases in the price of legumes would be required to make legume–maize intercrops more profitable than sole maize. This means it may be difficult to convince farmers to grow more legumes for slight benefits in crop system sustainability. Only the combination of increases in the price and demand for legume products, more productive varieties of the legumes, easier harvesting methods and cheaper maize prices, will improve the viability and adoption of legume-maize intercrop systems leading to more sustainable soil fertility management and productivity (Waddington et al., 2007). Furthermore, widespread implementation of these systems will require building of farmer knowledge and the awareness of soil fertility contributions and long-term benefits from legumes (Waddington et al., 2007).

1.8 Agroecosystem Management Practices in South America

Soils of developing countries are prone to degradation, and constitute 2 billion hectares of land area worldwide. Thus, soil management strategies must be identified in order to meet the food demands for an estimated additional 3.4 billion people within the next century and the likely change in food habits of the populations in emerging economies (Lal, 2007). At a world-wide scale, global climate change is affecting extraction and supply of goods and services from natural resources (Easterling and Apps, 2005). The greatest adverse impacts are likely to occur in areas where resource endowments are the poorest and the ability of farmers to respond and adapt is most limited (Easterling and Apps, 2005).

There are a number of differences in agricultural practices between North and South America, which stem mainly from the use of fertilizers and the intensity of land use change in North America. The capital-intensive agriculture in North America, which relies on large inputs of energy, machinery and fertilizers, results in large outputs of inorganic nutrients to adjacent aquatic and estuarine systems. However, the vast extent of land in both the tropical and temperate zones of South America often experienced a different land-use history, which includes slash-and-burn and conversion of forest or pasture ecosystems to low-input agriculture (Austin et al., 2006). In high-income countries such as Canada, governmental policies tend to subsidize production in order to protect the agricultural sector while in low-income developing countries, policies tend to tax and discourage production (Easterling and Apps, 2005).

The amount of land with degraded soil in South America is estimated at 138 million hectares (FAO, 2001). Erosion by water and by wind is quantitatively by far the most important degradation process. The main causative factors are deforestation, overgrazing and inappropriate land management. The SOM content is lower where degradation is more severe. The amount of C and N that can be sequestered through restoration of degraded land is considerable therefore, in areas where this is a technically and socio-economically viable option (FAO, 2001).

In the past decade, Latin American farmers have found that implementing certain management practices, such as conservation tillage, can be highly beneficial for soils. After harvest, crop residues are left on the surface to protect against erosion, and seed is directly planted into a groove cut into the soil. Weeds are controlled with herbicides or cover crops. The fastest uptake of conservation tillage systems has been in Brazil, where there are now 15 million hectares under no-tillage. In neighbouring Argentina, there are more than 11 million hectares under no-till, up from less than 100,000 ha in 1990 (Fabrizzi et al., 2003; Pretty et al., 2002).

Argentina is considered one of the major agricultural regions of the world, with most of its activity centered in the Pampa region where the conversion of natural grasslands to agriculture has taken place. This extensive region of more than 50 million hectares ranges in precipitation from humid systems with 1200 mm per year in the east to semiarid systems with 600 mm per year in the northwest. Cereal grain agriculture began in the 1870s with rapid expansion until 1937. Mechanized agriculture grew in importance starting in the 1970s, but lagged behind other countries due to the access to technology and large-scale international capital (Viglizzo et al., 1997). Water erosion and decreasing SOM are current challenges in the region. Since the 1990s, the area under no-till has expanded, allowing farmers to increase soil quality (Fabrizzi et al., 2003).

In the Pampas, the major crops – by cultivated area – are: soybean, wheat (*Triticum aestivum* L.), maize, and sunflower (*Helianthus annuus* L.) (Austin et al., 2006). The expansion of legume is one of the principal pathways of reactive N entering terrestrial ecosystems (Vitousek et al. 1997). This N-loading is especially relevant in Argentina, where both the expansion of soybean cropping on marginal land and the intensification of soybean cropping on currently cultivated land have increased markedly since the 1990s (Austin et al., 2006).

Da Silva et al. (2004) provided data on soil C storage under different cultivated pasture systems compared to native grassland. They observed that soil C in pastures under ongoing degradation do not contribute significantly to the sequestration of atmospheric C and the reduction of GHG concentration in the atmosphere (Jimenez and Lal, 2006). However it has been suggested that with improved management, the grasslands of the Argentine Pampas is one of the global regions with a high potential to sequester C (Hutchinson et al., 2006).

It is estimated that agricultural intensification will accelerate in the Latin American region in the next 50 years (Tilman et al. 2002). It has yet to be seen what the consequences will be for N loading in particular, as Latin American countries with substantial agricultural production, such as Argentina, increase the use of fertilizers in an effort to meet food demand and maintain a competitive presence in the world agricultural market. Monitoring agricultural practices in Latin America will be critical in determining the human impact on C and N cycles, due to the importance of the agricultural sector in the economies of most of the region (Austin et al., 2006).

1.9 Trends in Global Agriculture and Implications for Future Research

The relationship between anthropogenic activity by increasing atmospheric GHG concentration and global climate change has been discussed for over a century. In the mid-1800s, John Tyndall stated that CO_2 effectively trapped heat (Johnson et al., 2007). Early work on the relationship between atmospheric CO_2 concentration and temperature provided background data on atmospheric CO_2 concentration, thereby improving the ability to subsequently document increases in CO_2 concentration. In the 1970s, the GWP of other trace gases (CH_4, N_2O) became recognized

(Houghton, 2005; Johnson et al., 2007). By the 1980s, the anthropogenic influence on global warming gained enough credibility to spark international political activity leading to the establishment of the Intergovernmental Panel on Climate Change (IPCC). Most recently, ice core data of historical atmospheric CO_2 concentration has demonstrated a relationship between CO_2 and global temperature (IPCC, 2000; Johnson et al., 2007). The IPCC (2007) recently confirmed this as scientific fact that anthropogenic activities influence of GHG emissions and results in global climate change (Johnson et al., 2007).

Until the mid-20th century, agronomic practices, such as residue burning, did not support the return of organic matter to the soil, and were not conducive to sustained crop production (Johnson et al., 2006). According to the FAO (2007), increase in land-use change is expected to continue due to decreasing returns from technological progress, and greater use of marginal land with lower productivity. Use of these marginal lands increases the risk of soil erosion and degradation, with uncertain consequences for CO_2 emissions (Lal, 2004b). As demands for food increase, annual emissions of GHGs from agriculture are likely to increase as a consequence of land-use change (Smith et al., 2007). If CH_4 emissions grow in direct proportion to increases in livestock numbers, then livestock-related CH_4 concentration is expected to increase (FAO, 2003). Combined CH_4 emissions from enteric fermentation and manure management will increase by 21% between 2005 and 2020 (IPCC, 2007). However, changes in feeding practices and manure management could mitigate this increase. Agricultural N_2O emissions are projected to increase by 35-60% up to 2030 from increased fertilizer use and increased animal manure production (FAO, 2003; Schlesinger, 2000). Mosier and Kroeze (2000) estimated that N_2O emissions will increase by about 50% by 2020 relative to 1990. Although NH_3 is not directly considered a GHG, its transformation in soil does contribute to GWP. Synthetic fertilizers and residue burning are processes responsible for NH_3

emissions. Atmospheric NH_3 can be taken up by soils and converted to ammonium (NH_4^+) which can then be oxidized to N_2O (Follett et al., 2005). Thus, research and development of technologies to mitigate N_2O emissions during all of the steps in the biogeochemical cycling of N is needed (Follett et al., 2005).

Mitigation of climate change is most likely to be effective by a mix of practices, technologies, and policies that address the problem on different scales of time and geography (Follett et al., 2005). Agricultural science is addressing issues of GHG emissions and soil C- and N-sequestration, but policies that support technology transfer would help to ensure successful implementation. Policies to address GHG emissions based on science and backed by economic justification are the best incentives for natural resource managers, industry, and farmers to mitigate climate change. Realistic goals for soil C- and N-sequestration and the conservation of natural resources such as soil are necessary (Follett et al., 2005). The future evolution of GHG emissions from agriculture is uncertain. Due to stable or declining deforestation rates, and increased adoption of conservation agriculture practices, these emissions are likely to decrease or remain at low levels (Easterling and Apps, 2001; FAO, 2001; Smith et al., 2007).

According to current projections, the global population will reach 9 billion by 2050, an increase of about 50% over current levels (Follett et al., 2005; Lal, 2008; Smith et al., 2007). Growing demand for meat may induce further changes in land use, often increasing CO_2 emissions, and increased demand for animal feeds. Larger herds of beef cattle will cause increased emissions of CH_4 and N_2O. Intensive production of beef, poultry, will lead to increases in manure with consequent increases in GHG emissions. This is particularly true in the developing regions of South and East Asia, and Latin America, as well as in North America. Changes in subsidies and regional patterns of supply-and-demand are causing an increase in international trade of agricultural products (Smith et al., 2007).

Mitigation potential in the agricultural sector is highly uncertain; simulations of GHG emissions associated with land use vary greatly between scenarios. However, agriculture is a significant contributor to GHG and it is certain that mitigation will not occur without action. Higher emissions are projected in the future if current trends are left unconstrained. For example, atmospheric CO_2 concentrations, expected to double within the next century, may cause a positive-feedback cycle, affecting agroecosystems through changes in plant growth rates, drought tolerance, and nitrogen demands. Similarly, an increase in N_2O may cause a positive-feedback on CO_2 evolution, because of the tightly coupled cycling of C and N. Other projections include longer growing seasons in colder regions due to increasing temperatures (Smith et al., 2007). However, increasing temperatures will also accelerate decomposition of SOM, releasing stored SOC into the atmosphere. Increasing temperatures will also affect the spread of pests and diseases, which will impact agricultural production and practices (Arctic Climate Impact Assessment, 2004).

Conservation programs need to focus on C and N management in agricultural systems and best agricultural land management practices (Johnson et al., 2007). Access to adequate food is a basic human right, thus sustainable agriculture and the sustainable management of soil resources must be implemented (Lal, 2008). World soils can be prudently managed to achieve global food security and can also mitigate climate change by absorbing atmospheric GHG and converting it to humus through the process of soil C and N sequestration (Lal, 2007). While the agricultural sector has the potential to reduce its environmental footprint and mitigate GHG emissions, efforts from all sectors of the economy including alternatives to fossil fuels and a reduction in energy demands are necessary (Johnson et al., 2007).

1.10 Research Objectives

Croplands may be either a sink or a source for atmospheric C and N (Easterling and Apps, 2005; Gardiner and Miller, 2004; Miltner et al., 2005; Rastogi et al., 2002; Sauerbeck, 2001). The SOM sequestration is controlled by the balance between residues returned to the soil and GHG released from decomposition of residue. Cropping and tillage systems that promote the return of residues to the soil have the potential to sequester C and N by increasing aggregation and the SOC pool (Smith, 2004).

The significance of this research is to address the depletion of SOC and contribution to GHG in the atmosphere from land-use change. Sole cropping lowers the amount of organic matter returned to the soil, which depletes soil nutrients (Jarecki and Lal, 2003). Implementing sustainable agricultural practices, such as intercropping, may help to maintain the amount of SOM, and lead to long-term sequestration of C and N in the soil. This relationship has not yet been fully quantified in sole crop systems, and not at all in intercrop agroecosystems. Results from this work will help improve models to optimize agroecosystem design. Also, this study will contribute valuable information on how to meet food production needs through increased soil fertility.

General Objectives

1) To evaluate if intercrop agroecosystems sequester C and N in order to develop strategies for the long-term mitigation of atmospheric C and N.

2) To compare inputs and outputs of C and N between intercrop and sole crop agroecosystems.

Specific Objectives

1) To quantify GHG flux rates from intercropping and sole cropping agroecosystems

2) To quantify C and N inputs to the soil from crop residues in intercropping and sole cropping agroecosystems

3) To quantify soil C and N stocks in intercropping and sole cropping agroecosystems
4) To evaluate the decomposition of crop residues and quantify losses of C and N during decomposition in intercropping and sole cropping agroecosystems.

Hypotheses

The intercrop agroecosystems will have greater inputs and sequester more C and N in the soil than the sole crops. The GHG emissions will be lower in the intercrop agroecosystems than the sole crops.

Thesis Outline

This thesis is structured in 5 chapters, progressing from the general introduction which outlines broad concepts including global climate change, food insecurity, and land-use change, to later chapters which quantify and discuss in detail C and N exchanges between the soil and atmosphere.

Chapter 1 introduced the reader to the literature available which ultimately describes the potential of intercrop agroecosystems in Latin America to sequester C and N in soil.

Chapter 2 provides a description of the study site, including the study design, history of the region, and soil and climatic characteristics.

Chapter 3 examines 1) specific soil biophysical characteristics sampled at the site, 2) decomposition rates of residue in the field, and 3) C- and N-input from above- and belowground maize and soybean residue.

Chapter 4 examines emissions of CO_2, CH_4, and N_2O from the soils of intercrop and sole crop systems.

Chapter 5 provides an in-depth final consideration of the results, linking GHG emissions to soil and plant C and N, as well as providing recommendations for future research.

Chapter 2 – Study Site

2.1 Introduction

The research station (37°45’55’’S, 58°18’11’’W) is located 15 km NW of the city of Balcarce, Argentina, in the rolling Pampas. The agricultural research station is managed by the Instituto Nacional de Tecnología Agropecuaria (INTA). The site is located 130 m above sea level, and has an average annual precipitation of 860 mm (Andrade, 1995; FAO, 2008). The area surrounding Balcarce is characterized by low average temperatures during the growing season, with mean annual temperature of 13.9 °C, a frost-free period of about 150 days and a mean of 2409 hours of sunshine per year (Andrade, 1995; FAO, 2008). The underlying geology is granite limestone, and historically was grassland, until conversion to cropland under conventional tillage (FAO, 2008). The soil is classified as a Mollisol (FAO) with an organic matter content of 5.6 g kg^{-1} (Andrade, 1995). Currently, the region is considered one of the most suitable areas for grain production in the world, with nearly 50% of area devoted to agriculture in the semiarid and humid portions of the region with deep and well-drained soils (Steinbach and Alvarez, 2006). The flooding pampas is mostly devoted to cattle farming. These areas are characterized with shallow soil, frequent flooding, salinity, poor drainage, and water erosion. Mixed grain crop-cattle production systems have extended recently over most of the Pampas (Viglizzo et al., 2002).

The study design was a randomized complete block design (RCBD) with three replications. In each block the treatments were: 1) a maize sole crop; 2) a soybean sole crop; 3) an intercrop with a maize-soybean-soybean repeated arrangement (1:2 intercrop); and 4) an intercrop with a maize-maize-soybean-soybean-soybean repeated arrangement (2:3 intercrop) (Figure 2.1).

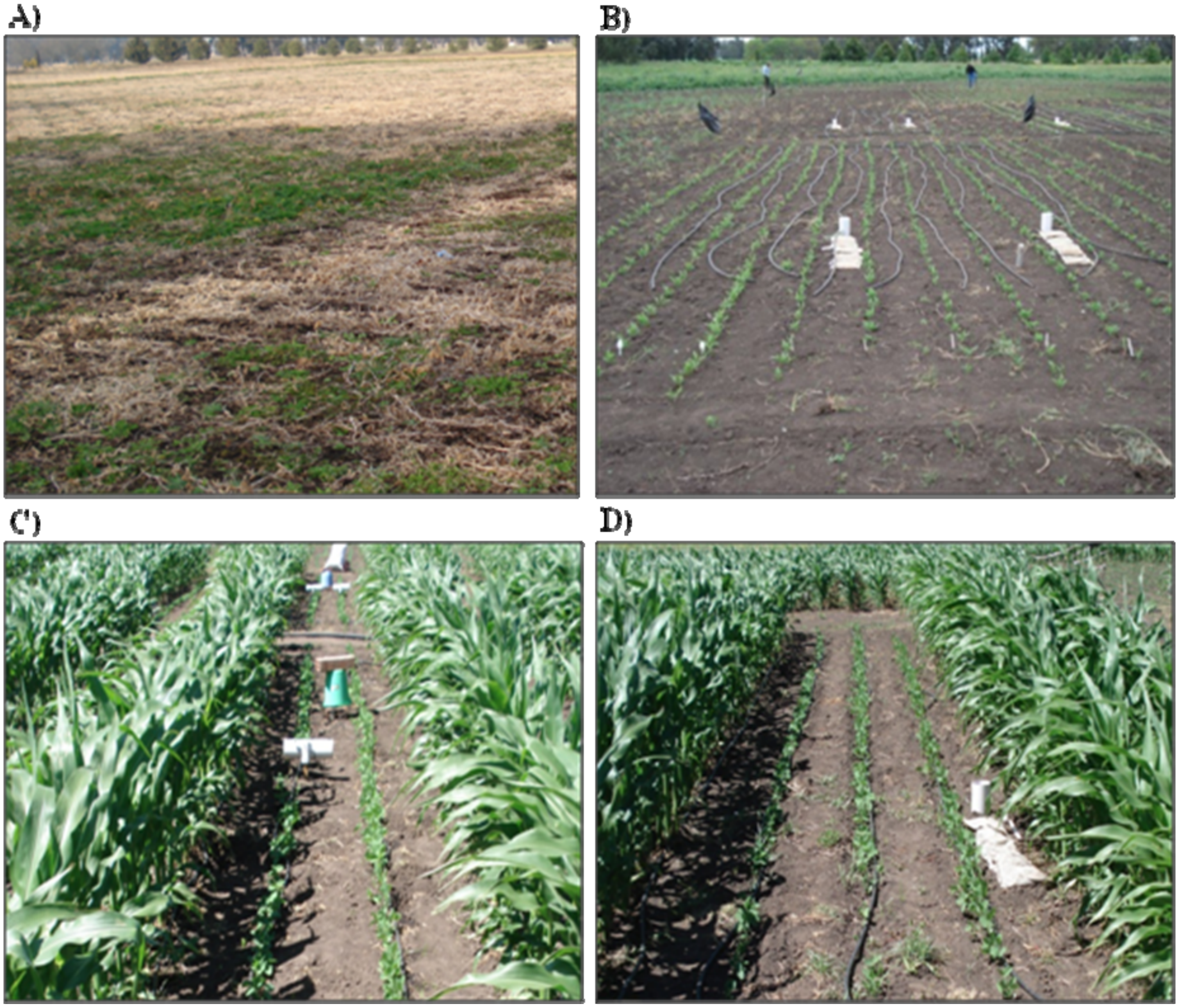

Figure 2.1: Field location at INTA-Balcarce, South-eastward-facing; A) in September, before plowing, B) in November after maize emergence, C) in December, a 1:2 intercrop, and D) a 2:3 intercrop.

The two intercrop arrangements were used for research on plant physiology to determine optimum plant density. Other research conducted at the same time as this study included yield and interception of photosynthetically active radiation.

Field work for this study took place from October 2007 to May 2008. Two maize hybrids were used in the experiment; a short- and a long-cycle hybrid. The long-cycle maize hybrid DK 682 was used in the maize sole crop treatments to reflect conventional farming systems in the region. The short-cycle maize hybrid DKC51-45AR2 was used in the intercrop treatments to facilitate the coordination of harvesting times with soybean. The field was plowed October 8th, 2007. Short- and long-cycle maize were planted on October 18th. Crops were fertilized with 35 kg ha^{-1} P before sowing, and maize was fertilized with 150 kg h^{-1} N. Soybean seeds were inoculated at planting on November 19th. Plots were over-sown and thinned by hand to the desired density. Maize final plant density for the 1:2 intercrop was 4.3 plants m^{-2} and for the 2:3 intercrop was 5.3 plants m^{-2}. Soybean plant density was 28 – 30 plants m^{-2} in all cases. Soil water to 1 m depth was kept over 50% of maximum available water by drip irrigation, to a total of 130 mm per replicate. Glyphosate herbicide was applied as needed. Short-cycle maize was harvested on March 25th, 2008 and long-cycle maize on April 15th. Soybean was harvested April 17th.

2.2 Historical Context

Sediment, pollen and biological indicators such as the density and abundance of mollusks have been used to reconstruct physical and hydrological changes in the paleontological history of the Pampas (Prieto et al., 2004). Between approximately 11 000 and 9000 years before present (ybp), shallow water without much circulation dominated the area. Fluctuations in climate are suggested to have initiated a reduction in water volume, interspersed with periods of flooding, resulting in a time when swamps and wetlands dominated from 9000 to 7000 ybp. A further shift to a subhumid-dry climate up to 3000 ybp (the late Holocene period) appears to be the main cause of the significant reduction of water bodies. Pollen records indicate that from 1790 to present, European settlement is the main cause of land-use change and disturbance of natural vegetation in the area (Prieto et al., 2004).

The largest worldwide land-use change involved conversion to agriculture at the expense of forests, grasslands and wetlands. The Pampas of Argentina were not an exception, both by conversion of grassland to cropland, and intensification of agriculture on land already cultivated (Viglizzo et al., 1997). The Pampas has a relatively short farming history, since humans have only colonized the region during the last century. The Pampas remained as native grassland until the beginning of the 20th century. Deforestation, over-grazing, intense cropping and unsustainable cultivation practices in association with extremely dry and windy conditions of the 1930s and 1940s caused a large dust-bowl episode that triggered severe dust storms, cattle mortality, crop failure, farmer bankruptcy and rural migration (Viglizzo and Frank, 2006). In the last half of the century, improved rainfall conditions favored the conversion of abandoned lands into grazing and cropland (Viglizzo and Frank, 2006). Currently, approximately 25% of uncultivated land persists, and it has been suggested that intensive cropping has surpassed ecological thresholds (Viglizzo and Frank, 2006). However, it has also been suggested that the Argentine Pampas is one of the global regions with a high potential to sequester C to help curb greenhouse gas emissions and thereby can contribute to mitigating global warming (Hutchinson et al., 2006).

2.3 Climate and Soil Characteristics

The Argentine Pampas is a wide plain of around 54 million hectares of fertile lands suitable for cattle and crop production. The relief is flat or slightly rolling, and grassland is the natural vegetation (Steinbach and Alvarez, 2006; Viglizzo and Frank, 2006). The region is not homogeneous, because soil quality varies and rainfall declines in general from north (200 mm/year) to southwest (1200 mm/year) (Steinbach and Alvarez, 2006). Rainfall regimes vary across time and space, causing drought and flood episodes that affect crop production (Viglizzo and Frank, 2006). The fragility of the soil to wind and water erosion is a major limiting factor in crop production in the region (Viglizzo et al., 1997). The climate is humid to sub-humid, with 80% of annual precipitation occurring in the spring-summer period (Fabrizzi et al., 2003).

Soils in the southern Pampas region, constituting approximately 13 million ha developed from eolic sediments of the quaternary period. The deep and well-drained soils of the region are moderately acidic, low in available phosphorus (P), and have a high SOC content, generally between 29 – 46 g kg^{-1}. Calcium carbonate content has been found to vary in the region between 14 – 64 g kg^{-1} (Prieto et al., 2004). Water erosion is a problem in areas between the sierras, or low mountains at 300 – 600 m above sea level. Increase in crop production of the last 20 years has resulted in decreased SOM and available P. However, introduction of the herbicide glyphosate, a genetically modified soybean strain, and the development of planters and drills have allowed the sustained increase of the area under no-till. The region is considered one of the most suitable areas for grain crop production in the world. Many cropland soils in the Pampas had been depleting SOC and degrading because of the sustained use of conventional tillage practices for more than a century of cultivation (Steinbach and Alvarez, 2006).

Chapter 3 – Soil Biophysical Characteristics, Crop Residue Input, and Decomposition

3.1 Introduction

Soil is one of the most important natural resources. A soil's quality depends on its capacity to function within the ecosystem and sustain biological productivity, to maintain environmental services, and promote plant growth and animal health (Kone et al., 2008). Some soil properties can be used as indicators of soil quality and ecosystem functioning, and include cation-exchange capacity, microbial communities, and levels of soil organic matter (SOM) (Fageria, 2002; Gregorich et al., 2001; Kone et al., 2008). Soil organic matter is the main source of C and nutrients for plants and microbes in soil.

The abundance of carbon (C) and nitrogen (N) is correlated with soil productivity, as adequate SOM is important in maintaining soil fertility (Kone et al., 2008). In order to meet increasing global food demands, increases in productivity of existing farm land through the restoration of marginal soils low in SOM is necessary (FAO, 2001). One way for farmers to do this is to use a complex agroecosystem design, such as intercropping. Some advantages of intercropping compared to traditional sole cropping include increased crop residue input and ground cover, reduced desiccation and erosion, and more efficient use of available nutrients and water (Geno and Geno, 2001). Intercropping has also been shown to provide a yield advantage compared to sole cropping (Alene et al., 2006). Population growth and the consequent decline in soil fertility and soil organic C (SOC) has led to greater land use intensification as well as crop diversification and the use of intercropping agroecosystems in developing countries (Ahmed et al., 1982).

The role of both plants and SOM are of paramount importance to increasing SOC and soil N. A critical first step in assessing the potential of a soil to mitigate atmospheric GHG is to evaluate initial C and N levels in soil, as well as the contribution of C and N from plant residue (Ordonez et al., 2008). Agricultural management practices affect the proportion of crop residues returned to the soil and influence C and N levels, which maintains or even increases crop production. In systems where access to external inputs of nutrients is limited, crop residues from legumes provide a readily available source of organic matter for soils (Kumar and Goh, 2003).

Soil organic C and N in croplands are directly linked to the return of fresh organic matter to the soil, which increases soil fertility and production. Carbon and N is stored in stable SOM, humus, where further decomposition occurs very slowly (Desjardins et al., 2005). Practices such as intercropping, the inclusion of legumes and cover crops, and the reduction in frequency of fallow periods increase C and N levels by increasing crop residue input (Blanco-Canqui and Lal, 2004; Gregorich et al., 2001; Janzen, 2005; Kong et al., 2005). These practices modify decomposition dynamics by changing soil aeration, water dynamics, and aggregation, as well as increasing the quantity of crop residues (Grandy and Robertson, 2007).

Plants are crucial to atmosphere−soil interactions because they control C and N fluxes to the atmosphere. The rate of SOM decomposition is a function of the quality of plant residue and its accessibility to soil organisms. The plant residues and the degree of decomposition are vital factors for SOC sequestration (Blanco-Canqui and Lal, 2004). Plant roots and residues furnish the soil with organic matter (OM). As SOM decomposition proceeds, organic particles associate with the soil matrix to form aggregates and sequester SOC (Blanco-Canqui and Lal, 2004). Contributing factors acting on the development of organic matter in the soil include soil properties (texture, clay content, bulk density, and soil mineralogy), climatic factors (temperature and moisture), and vegetation (residue input, plant composition) (FAO, 2001). The rate of organic matter decomposition is affected by the disruption of aggregates, which physically protect SOM. Aggregate breakdown by rain, tillage or other agricultural practices increases SOM exposure to air. The increased availability of oxygen and accessibility to microorganisms stimulates mineralization of the organic matter, and consequently a release of CO_2 to the atmosphere (FAO, 2001).

Researchers have suggested that legume-based cropping systems, with low C/N ratio organic residues, significantly increased the retention of soil C and N (Drinkwater et al., 1998; Gregorich et al., 2001; Mungai and Motavalli, 2006; Shiyomi and Koizumi, 2001; Vandermeer et al., 1998). Agroecosystem designs, such as intercropping, that incorporate legumes may increase SOC and N. Studies have been conducted to find biochemical characteristics that predict the decomposition rates of crop residues. Some of these include cellulose, lignin and polyphenol concentrations (Kumar and Goh, 2003; Nakhone and Tabatabai, 2008). Cellulose in plants is associated with hemicellulose and other structural polysaccharides, surrounded by a lignin sheath. Lignin, a complex polyaromatic matrix, is covalently associated with hemicellulose, preventing access of hydrolytic enzymes and acids from soil microbes to metabolize the plant residue. The highly ordered, crystalline structure of cellulose itself poses another obstacle to hydrolysis (Varga et al., 2003). Soluble compounds, such as amino acids and monomeric sugars are easily oxidized by soil microbes. More complex structural compounds, such as proteins take more time for microbial transformation. Celluloses, hemicelluloses and lignin are highly complex structures with high C/N ratios that are resistant to microbial enzymatic action. Thus, the C/N ratio of plant residues affects soil C and N by affecting the rate of SOM decomposition. Cattanio (2008) described N mineralization as negatively correlated with C/N ratio; the higher C/N, the slower N is transformed in soil. Organic residues with higher C/N persist for a longer time, and residues with lower C/N ratios are rapidly decomposed, quickly enhancing the formation of aggregates, but their action is transient. Slowly decomposing residues have a more gradual impact on aggregation, but their long-term effect on SOC sequestration is higher (Blanco-

Canqui and Lal, 2004). Thus, maize-soybean intercropping agroecosystems take advantage of the combination of residues with high and low C/N ratios. The C/N ratio of soybean is approximately 10:1, while the C/N ratio of maize is approximately 15:1 (Table 3.1).

Table 3.1: Chemical characteristics of soybean and maize residues (g kg^{-1}) (Adapted from Johnson et al., 2007; Liang et al., 2007; Nakhone and Tabatabai, 2008)

Residue*	Total N	Protein	Total C	Lignin	Hemicellulose	Cellulose
Soybean	41.6	240	427	60.7	763	170
Maize	28.6	123	437	117	418	399

*Whole plant dry matter

The C/N ratios of maize and soybean indicate that soybean residue is composed of structural compounds more easily available to microorganisms. Maize residue has a higher amount of lignin, hemicellulose and cellulose than soybean residue, which are complex structures high in the proportion of C and less available to microbial action. Maize residue, therefore, is more slowly decomposed, and C from maize residues will persist for a longer period of time in the soil.

To determine the effect of crop residues on SOM, crop residue C- and N-input is measured. Crop residue decomposition rates are measured; the most often-used method is the litterbag technique. In this method, a known amount of crop residue is enclosed in mesh bags and laid on the soil surface. A large number of litterbags are installed at the start of the experiment and sampled periodically over time. The decomposition rates are determined from the mass loss of the litter in the bag. This method is simple and inexpensive and is therefore widely used (Kurz-Besson et al., 2005). The litterbag method has been used to assess the effects of various environmental factors and experimental conditions on organic matter breakdown, such as quality and quantity of organic matter, climate, and soil properties (Knacker et al., 2003). Some systems in which this has been studied include maize-soybean rotations or conventional maize systems in which legume crop residue was fed to cattle, and the subsequent manure applied to the field as the primary N source. Agroecosystems that include legumes can maintain higher organic matter levels than continuous cropping systems with non-leguminous row crops, although crop rotation effects are also altered by tillage (Blanco-Canqui and Lal, 2004).

Soil moisture and temperature are the most important climatic factors in organic matter decomposition (Mungai and Motavalli, 2006; Knacker et al., 2003. Dry conditions usually reduce decomposition rates while temperature changes may either accelerate or slow down the decomposition. As soil water becomes limited, microbial, nutrient and gas movement are also decreased which may retard soil microbial processes (Mungai and Motavalli, 2006). Temperatures above 20 °C in combination with high moisture have been found to increase microbial activity whereas the activity of some earthworm species is highest at lower temperatures (Knacker et al., 2003). Other soil properties favoring rapid decomposition of plant litter include a near neutral soil pH and adequate nutrient availability allowing a diverse microbial population to be active in the mineralization of C and N in the soil (Mungai and Motavalli, 2006). During residue decomposition by soil microbes, C and N cycles in soil are strongly linked mainly because of the simultaneous assimilation by soil microbes of C and N. Assimilation of C depends on the rate of decomposition of plant material and N assimilation is then determined by C availability and the type of substrate, particularly its C/N ratio (Mary et al., 1996). The sources of N for microbes can be the plant residue itself, the mineral N already present in soil, and recycled soil microbial biomass (SMB) (Mary et al., 1996).

Research on intercropping has not received much attention until very recently, because the practice was thought to be suited only in smallholder farms in developing countries (Ahmed et al., 1982). In addition, the research conducted on these agroecosystems has focused mainly on yield and resource use. Therefore, the objectives of this study were to compare inputs of C and N between maize and soybean sole crops and intercrops. More specifically, the objective of this study was to measure SOC and N, crop residue inputs of C and N, and rates of decomposition of maize and soybean crop residue.

3.2 – Materials and Methods

3.2.1 Soil Sampling

Soil sampling occurred in January 2008 using a soil corer with an internal diameter of 5 cm. Soil samples were taken at depth increments of 0 – 10 cm, 10 – 20 cm, 20 – 40 cm, 40 – 80 cm and 80 – 120 cm for each treatment in each of the three blocks. Three samples were extracted from each repetition in each block and were composited into a single sample, using the coning and quartering technique (Schumacher et al., 1990) for a total of 12 samples per increment depth. Samples (approximately 500 g each) were weighed in the field for wet weight, and transported to the lab for air-drying. Soil moisture and temperature was noted at the time of sampling using the HH2 Moisture Meter with an ML2 ThetaProbe.

A sub-sample of approximately 20 g of soil was dried at 105 °C for 48 hours for bulk density analysis. Bulk density was calculated using the inner diameter of the core sampler and soil segment depth to calculate soil volume, and the oven dry weight of the soil. Roots, leaves and rocks were removed by hand before sieving (2 mm) the air-dried soil samples. The remainder of the air-dry soil was transported to Canada.

Soil carbonates were removed by acid fumigation prior to elemental analysis (Harris et al., 2001). For example, soil was ground to a fine powder (< 250 μm) using a ball-mill and 300 mg subsamples weighed into glass scintillation vials. Soil was moistened using 0.4 ml deionized water. Vials were placed into a desiccator with 100 ml of 12 N HCl. The desiccator was sealed and samples were exposed to HCl vapour for 72 hours. After 72 hours, the HCl was removed, the desiccator was resealed and the samples were subjected to repeated vacuum evacuation for 1 hour to remove all HCl vapour. Samples were then oven dried at 40 °C for 48 hours, ball-milled again, and then transferred to tin capsules for C and N analysis. Elemental analysis for SOC and TN was done on a Costech ECS 4010. Bulk density values were used to convert SOC and N concentrations (g kg^{-1}) into mass per area (g m^{-2}) for each depth increment.

Soil data were examined for homogeneity of variance and normality. All data were examined for homogeneity of variance. Data for soil characteristics were tested using a one-way ANOVA in SPSS (SPSS v. 15.0). Significant differences were further tested using Tukey's LSD multiple comparison test. For all statistical analyses, the threshold for determining significant differences was $P < 0.1$.

3.2.2 Crop Residue Sampling

Carbon- and N-input from crop residue was sampled during harvest within a 0.5 m x 0.52 m area in each replicate at Balcarce, Argentina. Crops were separated into above- and belowground plant components after removal of cobs and pods. Crop roots were dug from the soil and hand-washed in water to remove soil adhering to fine roots. Samples were dried at 65 °C for one week and weighed. Dried plant material was then ground in a Wiley mill (2mm). Dried litter was then ball-milled (< 250 μm) and approximately 5 μg weighed into tin capsules for C and N analysis using a Costech 4010 elemental analyzer. Concentrations of C and N (g kg^{-1}) were converted into mass per area (g m^{-2}) using the residue dry weight per area sampled in each treatment.

Crop residue input data were examined for homogeneity of variance and normality. All data were examined for homogeneity of variance. Data for soil characteristics were tested using a one-way ANOVA in SPSS (SPSS v. 15.0). Significant differences were further tested using Tukey’s LSD multiple comparison test. For all statistical analyses, the threshold for determining significant differences was $P < 0.1$.

3.2.3 Litterbag Sampling

The rate of crop residue decomposition was quantified in the field using the litterbag technique. Plant litter tissue was collected at harvest in April 2007 and dried for 7 days at 65 °C. Approximately 20 g of oven-dry soybean and maize crop residue was placed into 25 cm x 25 cm litterbags with a 2 mm mesh size. The maize litter was cut into pieces of approximately 10 cm to accommodate the litterbag size. For the maize sole crop treatments, only maize crop residue was used; for the soybean sole crop treatments, only soybean crop residue was used. For the intercrop treatments, maize and soybean crop residue was weighed in proportion to the number of rows of maize to soybean.

Ten litterbags per treatment were placed on top of the soil in between crop rows. One litterbag per treatment per block was collected each month between October 2007 and July 2008. Soil particles adhering to the crop residue were removed by hand and the residue was then oven-dried and weighed. Dry plant material from five months of litter available at that point was ground in a Wiley mill (2 mm) and transported to Canada. Dried litter was then ball-milled to a fine powder (< 250 µm) and approximately 5 µg weighed into tin capsules for C and N analysis using a Costech 4010 elemental analyzer. The data were expressed as % crop residue remaining and fitted to a single exponential decay function to calculate the decomposition rate constant.

$$W_t = W_0 e^{-kt} \tag{3.1}$$

The single exponential model assumes that all plant tissue decomposes at the same rate, where W_t is the percent of dry weight residue remaining at time t, in days, W_0 is the original percentage of dry weight crop residue, and k is the decomposition rate constant.

All data were examined for homogeneity of variance and normality. Data for soil characteristics were tested using a one-way ANOVA in SPSS (SPSS v. 15.0). Significant differences were further tested using Tukey's LSD multiple comparison test. For all statistical analyses, the threshold for determining significant differences was $P < 0.1$.

3.3 – Results and Discussion

3.3.1 Soil biophysical characteristics

Bulk density values (g cm^{-3}) increased slightly with depth in all treatments, but did not differ significantly (Table 3.2). The SOC concentration (g kg^{-1}) in the 2:3 intercrop was significantly lower compared to the other treatments at the 80 – 120 cm depth. In all other cases, SOC and TN concentration were not significantly different ($P< 0.1$) between treatments. A significant difference in SOC concentration was observed with depth in all treatments below 40 cm. The TN concentration (g kg^{-1}) differed significantly below a 20 cm depth for the maize and soybean sole crops and at 40 cm depth for the 1:2 and 2:3 intercrops. The C/N ratio was significantly lower in the 2:3 intercrop at 80 – 120 cm depth than the other treatments. In the maize and soybean sole crops and the 1:2 intercrop, there was a significant difference in C/N ratios between depths at 80 – 120 cm.

Table 3.2: Summary of soil properties in the soil profile in increments up to 120 cm depth at Balcarce, Argentina, including bulk density (g cm^{-3}), SOC and TN concentration (g kg^{-1}) and C/N ratio. Treatments were maize sole crop, soybean sole crop, 1:2 intercrop, and 2:3 intercrop. Soil was sampled January 22 - 23, 2008.

	Depth (cm)	Treatment			
		Maize	**Soybean**	**1:2 intercrop**	**2:3 intercrop**
Bulk Density (g cm^{-3})	0 - 10	1.21 ^{A,a}	1.11 ^{A,a}	1.17 ^{A,a}	1.13 ^{A,a}
	10 - 20	1.24 ^{A,a}	1.24 ^{A,a}	1.18 ^{A,a}	1.15 ^{A,a}
	20 - 40	1.25 ^{A,a}	1.27 ^{A,a}	1.16 ^{A,a}	1.24 ^{A,a}
	40 - 80	1.33 ^{A,a}	1.35 ^{A,a}	1.27 ^{A,a}	1.19 ^{A,a}
	80 - 120	1.42 ^{A,a}	1.26 ^{A,a}	1.33 ^{A,a}	1.46 ^{A,a}
SOC (g kg^{-1})	0 - 10	31.5 ^{A,a}	33.3 ^{A,a}	30.9 ^{A,a}	28.9 ^{A,a}
	10 - 20	30.7 ^{A,a}	33.7 ^{A,a}	28.0 ^{A,a}	27.8 ^{A,a}
	20 - 40	25.7 ^{A,a}	30.4 ^{A,a}	25.7 ^{A,a}	21.9 ^{A,a}
	40 - 80	17.5 ^{B,a}	25.7 ^{B,a}	18.1 ^{B,a}	10.5 ^{B,a}
	80 - 120	12.9 ^{B,a}	22.8 ^{B,a}	13.3 ^{B,a}	5.5 ^{B,b}
TN (g kg^{-1})	0 - 10	1.9 ^{A,a}	1.7 ^{A,a}	1.7 ^{A,a}	1.5 ^{A,a}
	10 - 20	1.8 ^{A,a}	1.6 ^{A,a}	1.4 ^{A,a}	1.5 ^{A,a}
	20 - 40	1.1 ^{B,a}	1.2 ^{B,a}	1.1 ^{A,a}	1.4 ^{A,a}
	40 - 80	0.5 ^{B,a}	0.5 ^{B,a}	0.5 ^{B,a}	0.5 ^{B,a}
	80 - 120	0.1 ^{B,a}	0.1 ^{B,a}	0.1 ^{B,a}	0.3 ^{B,a}
C/N	0 - 10	16.6 ^{A,a}	19.6 ^{A,a}	18.2 ^{A,a}	19.3 ^{A,a}
	10 - 20	17.1 ^{A,a}	21.1 ^{A,a}	20.0 ^{A,a}	18.5 ^{A,a}
	20 - 40	23.4 ^{A,a}	25.3 ^{A,a}	23.4 ^{A,a}	15.6 ^{A,a}
	40 - 80	35.0 ^{A,a}	51.4 ^{A,a}	36.2 ^{A,a}	21.0 ^{A,a}
	80 - 120	129.0 ^{B,a}	228.0 ^{B,a}	133.0 ^{B,a}	18.3 ^{A,b}

Values followed by the upper case letter are not significantly different at $P < 0.1$ between depths. Values followed by the same lower case letter are not significantly different between treatments ($P < 0.1$).

Bulk density values from this study corresponded to values from other studies (Noellemeyer et al., 2008). In a study using a maize-soybean rotation, Al-Kaisi et al. (2005) also found that bulk density did not differ significantly between treatments at different soil profile depths. Sainju et al. (2008) likewise found that SOC concentration did not vary between treatments, but found a significant increase in SOC below 10 cm in a *Gossypium hirsutum* L.–(cotton)–maize rotation.

In this study, SOC (kg m^{-2}) decreased with depth up to 120 cm. The 2:3 intercrop had significantly lower ($P < 0.1$) SOC than the other treatments in the 80-120 cm depth increment (Table 3.3). The highest value for SOC occurred in the soybean sole crop with a mean of 4192.17 g m^{-2} at 10 – 20 cm depth. For all other treatments, the highest values of SOC occurred at the 0 – 10 cm depth. The lowest SOC value occurred in the 2:3 intercrop at 80 – 120 cm depth, where SOC was lowest for each treatment. At the 40 – 80 cm depth, there was a significant decrease in SOC.

Significantly higher levels of C in the top 40 cm and N in the top 20 cm of the soil profile from this study correspond to results from other researchers. Many long and short-term studies of maize production or maize-soybean in rotation have shown an increase in C in the surface layers with altered management practices (Mann et al., 2002). In a study of C levels in conventional till and no-till systems, Al-Kaisi et al. (2005) attributed increases in C up to 15 cm to a decrease in the rate of decomposition. Clapp et al. (2000) recently examined some of the interactions between maize harvest, fertilization, and C in a 13-year study in Minnesota, USA. Similar to the findings reported in Table 3.3, they found significantly higher levels of C up to a depth of 30 cm.

Table 3.3: Soil organic C and TN (g m^{-2}) in increments up to 120 cm depth in the soil profile at Balcarce, Argentina. Treatments were maize sole crop, soybean sole crop, 1:2 intercrop, and 2:3 intercrop. Soil was sampled January 22 - 23, 2008.

	Depth (cm)	**Treatment**			
		Maize	**Soybean**	**1:2 intercrop**	**2:3 intercrop**
SOC (g m^{-2})	0 - 10	3801.11 (93.30) ^{A,a}	3687.31 (238.69) ^{A,a}	3606.44 (208.26) ^{A,a}	3259.63 (154.61) ^{A,a}
	10 - 20	3820.21 (69.29) ^{A,a}	4192.17 (307.62) ^{A,a}	3317.25 (254.26) ^{A,a}	3205.71 (178.47) ^{A,a}
	20 - 40	3201.88 (382.37) ^{A,a}	3847.32 (243.00) ^{A,a}	2973.75 (173.44) ^{A,a}	2710.78 (236.01) ^{A,a}
	40 - 80	2314.20 (190.17) ^{B,a}	3478.92 (110.69) ^{B,a}	2299.18 (33.73) ^{B,a}	1244.57 (85.00) ^{B,a}
	80 - 120	1826.43 (117.37) ^{B,a}	3333.59 (380.74) ^{B,a}	1767.61 (12.52) ^{B,a}	805.18 (104.53) ^{B,b}
TN (g m^{-2})	0 - 10	229.27 (5.63) ^{A,a}	188.24 (12.58) ^{A,a}	198.41 (11.46) ^{A,a}	169.19 (8.02) ^{A,a}
	10 - 20	223.99 (4.06) ^{A,a}	199.03 (22.71) ^{A,a}	165.86 (12.71) ^{A,a}	172.97 (9.63) ^{A,a}
	20 - 40	137.05 (16.37) ^{B,a}	151.87 (29.94) ^{B,a}	127.28 (7.42) ^{B,a}	173.29 (15.09) ^{A,a}
	40 - 80	66.12 (5.43) ^{B,a}	67.68 (18.42) ^{B,a}	63.51 (0.93) ^{B,a}	59.27 (4.05) ^{B,a}
	80 - 120	14.16 (0.91) ^{B,a}	14.62 (4.38) ^{B,a}	13.29 (0.09) ^{B,a}	43.92 (5.70) ^{B,a}

Values followed by the upper case letter are not significantly different at $P < 0.1$ between depths. Values followed by the same lower case letter are not significantly different ($P < 0.1$) between treatments. Standard errors are given in parentheses ($n = 3$).

In longer-term studies, some researchers found losses of C in the surface layers. In a study during three years of no-till maize sole cropping systems, average losses of SOC from 2001 to 2004 ranged from 80 to 129 g C m^{-2} up to a 15 cm depth (Verma et al., 2005). Small but not significant decreases in SOC were measured for the 0 – 30 cm depth. The researchers suggested that continuous maize production resulted in either a decline or no detectable change in SOC during the first three years of no-till farming in the three cropping systems in their study (Verma et al., 2005).

The highest values of TN occurred in the maize sole crop treatment from 0 – 10 cm with 229.27 g m^{-2} (Table 3.3). The lowest value of TN occurred in the 1:2 intercrop from 80 – 120 cm with a mean of 13.29 g m^{-2}, where TN was lowest within all treatments. Significant decreases in N were found at the 20 – 40 cm depth for the maize and soybean sole crops and the 1:2 intercrop, and at the 40 – 80 cm depth for the 2:3 intercrop. These results correspond with Gregorich et al. (2001), who found that under a sole crop maize production system, significant losses of SOM also occurred at depths greater than 20 cm. Soils under legume rotation contained more SOM than those under maize monoculture, whereas in this study losses of N occurred below 20 cm in all treatments (Gregorich et al., 2001). Al Kaisi et al. (2005) found that there was a significant difference ($P < 0.1$) in N below a 15 cm depth. In addition, these researchers found that N varied with depth due to tillage effect below 5 cm. Peralta and Wander (2008) report that SOM levels are restricted when N is a limiting factor; competition for N results in increased SOM decay and ultimately degrade soils.

Enhancing levels of SOM occurs by increasing C and N input from crop residues to the soil and by decreasing SOM decomposition, thereby creating a net sink. Jarecki and Lal (2003) reported that soil C and N sequestration is attributed to management practices that minimize soil disturbance and erosion thereby decreasing aeration and thus microbial respiration. Changes in SOM levels are strongly affected by choice of cropping systems which increase crop productivity and enhance residue input levels (Jarecki and Lal, 2003). Intercrop agroecosystems may be one such system that increases SOM.

Soil can be managed to increase SOC and TN storage on a long-term basis (at least 10 years) by implementing conservation soil and crop management practices (Al-Kaisi et al., 2005). However, short-term management effects on SOC and TN dynamics are complex and often variable; some researchers found that there was a delayed response after implementation of management techniques (Al-Kaisi et al., 2005). There may be little or no increase in SOC and TN in the first 2 – 5 years, but a large increase after 5 – 10 years (Al-Kaisi et al., 2005). In a summary of longer-term experiments, West and Post (2002) suggested an increase in average annual SOC sequestration rate of 46 g C m^{-2} for maize-soybean rotations compared to continuous maize systems. In a maize-sobyean rotation at Balcarce, Studdert and Echeverria (2000) found that these crops in rotation caused a decrease in SOC to a depth of 17 cm of 60.3 g m^{-2} compared to the maize sole crop. Six et al. (2004) also analyzed data from long-term experiments, and concluded that average C sequestration in the first few years following changes in management practices were small or sometimes negative, as found in this study, followed by a gradual increase over time. They reported average C sequestration rates in temperate climates of 10 g m^{-2} in the top 30 cm of soil over 20 years (Verma et al., 2005). Ordonez et al. (2008) studied 10 ecosystems, including grasslands and conventional agricultural systems, in the central highlands of Mexico. The lowest SOC, sampled to a depth of 30 cm, was found in conventional agricultural systems, with 14.7 g m^{-2}. Their results suggest that while there is a potential C-sink in agricultural soils, that capacity is quickly reduced under management practices that reduce inputs of organic matter to the soil. Some researchers have even suggested that up to 85% of SOC is sequestered in the top 7 cm of soil (West and Post, 2002). Experiments conducted in Argentina have documented that enhancing and maintaining a high level of SOM is important to sustaining productivity of soils (Lal, 2005). In the Pampas, Diaz-Zorita et al. (1999) observed that losses of 1 Mg of SOM decreased crop yield by about 40 kg ha^{-1}. Thus it is important to incorporate crop residues and rotations for sustainable use of the soils of the Pampas (Lal, 2005). Intercrop agroecosystems, if adopted in the Pampas, may help restore SOM by increasing input of C and N to the soil.

3.3.3 Carbon and nitrogen input from maize and soybean residue

Nutrient input from combined maize and soybean in all treatments was calculated from crop residue collected at harvest (Table 3.4). The C-input (g m^{-2}) from shoots and leaves was greatest in the maize sole crop and then the 1:2 intercrop, which showed a significant greater C compared to the soybean sole crop and the 2:3 intercrop. The lowest input of C from both above- and belowground components was in the soybean sole crop. For total crop residue, the maize sole crop had the highest C-input, followed by the 1:2 and the 2:3 intercrops and then the soybean sole crop.

Table 3.4: Combined maize and soybean residue C- and N-input (g m^{-2}) from shoots, leaves and roots at harvest at Balcarce, Argentina.

	Treatment	Input from combined maize and soybean residue (g m^{-2})		
		Shoots and leaves	Roots	Total
C	Maize	795.12 (24.50) ^{a}	108.67 (18.90) ^{a}	903.79 (138.28) ^{a}
	Soybean	407.29 (31.73) ^{b}	94.54 (32.50) ^{a}	501.84 (144.82) ^{a}
	1:2 intercrop	667.22 (62.71) ^{a}	110.19 (14.54) ^{a}	777.41 (124.38) ^{a}
	2:3 intercrop	450.83 (33.94) ^{b}	101.49 (13.92) ^{a}	552.32 (69.62) ^{a}
N	Maize	12.44 (2.45) ^{a}	3.05 (0.25) ^{a}	15.49 (2.16) ^{a}
	Soybean	12.85 (3.17) ^{a}	1.25 (0.10) ^{b}	14.10 (5.47) ^{a}
	1:2 intercrop	10.81 (2.27) ^{a}	2.46 (0.40) ^{a}	13.27 (2.41) ^{a}
	2:3 intercrop	8.16 (1.45) ^{a}	2.19 (0.30) ^{a}	10.35 (1.24) ^{a}

Values followed by the same lower case letter are not significantly different ($P < 0.1$) between treatments. Standard errors are given in parentheses ($n = 3$).

The N-input was greatest in the maize sole crop, followed by the soybean sole crop, then the 1:2 intercrop and the 2:3 intercrop. The N-input from shoots and leaves was greatest in the soybean sole crop. The greatest N-input from roots was in the maize sole crop. The lowest N-input from shoots and leaves was in the 2:3 intercrop, and from roots was in the soybean sole crop, which was significantly lower than the other three treatments.

Values of C- and N-input from maize and soybean residue in this study are slightly higher than reported by other researchers. In an 11-year old system started in 1984 at Balcarce, Argentina, Studdert and Echeverria (2000) found that in maize sole crop systems, approximately 320 g C m $^{-2}$ of aboveground maize residue and 120 g C m^{-2} of soybean residue was returned to the soil. In this study, approximately 795.12 g C m $^{-2}$ of aboveground maize residue and 407.29 g m^{-2} of soybean residue was returned to the soil. In a summary of crop residue contribution to SOC, Wilhelm et al. (2004) found that leaving both maize and soybean biomass residues on the soil increased C-input by 56 g C m^{-2}, nearly twice the contribution as leaving only maize residues, which increased C-input by 32 g C m^{-2}. This input is comparable to findings in the experiment at Balcarce, where increases in C-input occurred in the 1:2 and 2:3 intercrop systems relative to the soybean sole crop of approximately 176 and 51 g C m^{-2}, relatively. However, contrary to the findings of Wilhelm et al. (2004), input from both intercrops in this study decreased relative to the maize sole crop. In a review of crop residue management of major grain crops in the USA from 1940 to 2000, Johnson et al. (2006) found that approximately 133 g m^{-2} of root-derived C and 120 g m^{-2} of shoot and leaf C from soybean sole crops is returned to the soil. These researchers report approximately 380 g m^{-2} of root-derived C and 298 g m^{-2} of shoot and leaf C from maize sole crops. These values are slightly lower than the values reported in this study, but also indicate that maize residues return a greater amount of C to the soil than soybean sole crops.

A breakdown of the nutrient inputs from separate components of maize and soybean crop residues from this study is presented in Table 3.5. The greatest C-input from maize residue occurred in the maize sole crops, followed by the 1:2 intercrop and the 2:3 intercrop, both from above- and belowground components. Similarly, the greatest C-input from soybean shoots and leaves occurred in the soybean sole crops, followed by the 1:2 intercrop and the 2:3 intercrop. From soybean root residue, there was slightly greater C-input in the 2:3 intercrop than the 1:2 intercrop, but the greatest input occurred in the soybean sole crop. Like input of C, the greatest N-input from maize residue occurred in the maize sole crops, followed by the 1:2 intercrop and the 2:3 intercrop, both from above- and belowground components. Also the greatest N-input from soybean residue occurred in the soybean sole crops, followed by the 1:2 intercrop and the 2:3 intercrop both from above- and belowground components.

In a study conducted in Turrialba, Costa Rica, Chang and Shibles (1985) found that there was no significant difference in N-input from legume residue between sole crop and intercrop plots. However, there was a significant difference in N-input from maize residue in intercrop plots versus sole crop plots. In addition, there was a linear relationship between greater N-input and a higher proportion of legume plants. In a study of cereal-legume rotations in the Philippines, Herridge and Bergersen (1988) reported that more N in the form of nitrates remained in soil following legume crops than after cereal crops. This might be due to less N being taken up by the legume, increased mineralization or due to added N from legume crop roots to soil (Li et al., 2005). The impact of root C and N on SOM turnover in the temperate zone as well is described by several authors. In a study by Johnson et al. (2006) using data from major grain crops in the USA, total root-derived C contributed

1.5 to 3 times more C than shoot-derived C. In Connecticut, Hooker et al. (2005) likewise found greater C-input from roots than shoots and leaves, and explains the difference by preferential consumption of root material by microbes.

Table 3.5: Input of C and N (g m^{-2}) from maize and soybean residue components from shoot and leaf, and roots at harvest at Balcarce, Argentina.

	Treatment	Input from maize and soybean residue (g m^{-2})		
		Shoots and leaves	Roots	Total
C input from maize residue	Maize	795.12 (24.54) ^{a}	108.67 (8.90) ^{a}	903.79 (138.40) ^{a}
	Soybean	–	–	–
	1:2 intercrop	523.10 (105.62) ^{b}	79.66 (40.18) ^{b}	602.77 (129.44) ^{b}
	2:3 intercrop	317.21 (97.51) ^{b}	70.74 (20.64) ^{b}	387.95 (99.36) ^{b}
C input from soybean residue	Maize	–	–	–
	Soybean	407.29 (101.71) ^{a}	94.54 (15.63) ^{a}	501.84 (131.71) ^{a}
	1:2 intercrop	144.12 (31.29) ^{b}	30.53 (5.85) ^{b}	174.65 (59.72) ^{b}
	2:3 intercrop	133.62 (27.25) ^{b}	30.74 (3.04) ^{b}	164.36 (70.00) ^{b}
N input from maize residue	Maize	12.44 (3.84) ^{a}	3.05 (2.50) ^{a}	15.49 (2.45) ^{a}
	Soybean	–	–	–
	1:2 intercrop	6.90 (8.66) ^{b}	2.00 (1.04) ^{a}	8.90 (2.27) ^{b}
	2:3 intercrop	4.37 (3.79) ^{b}	1.59 (0.59) ^{b}	5.97 (1.45) ^{b}
N input from soybean residue	Maize	–	–	–
	Soybean	12.85 (1.00) ^{a}	1.25 (0.43) ^{a}	14.10 (3.43) ^{a}
	1:2 intercrop	3.91 (0.85) ^{b}	0.47 (0.05) ^{b}	4.37 (1.61) ^{b}
	2:3 intercrop	3.78 (0.49) ^{b}	0.60 (0.03) ^{b}	4.38 (0.76) ^{b}

Values followed by the same lower case letter are not significantly different ($P < 0.1$) between treatments. Standard errors are given in parentheses (n = 3).

Ma et al. (2002) found a significant difference ($P < 0.1$) in residue input from maize in rotation with soybean (530 g m^{-2}) compared to a maize sole crop (460 g m^{-2}) in a four-year trial at Ottawa, Canada. Total plant N from maize residue also showed increases when in rotation with soybean (10.6 g m^{-2}) versus sole crop residue (7.4 g m^{-2}). In this study, N-input from maize residue was greater in the maize sole crop than either intercrop, however measurements by Ma et al. took place after four years whereas the system from this study was in its first year of establishment.

In terms of long-term soil quality and carbon sequestration, Mann et al. (2002) found that optimal increases in SOC occurred when maize litter was left on the soil surface at harvest and fertilization was mixed with the crop residue. Similarly, Karlen et al. (1994) found that doubling the input of maize litter at their study site in Iowa resulted in long-term increases in aggregate stability and total SOC after 10 years (Mann et al., 2002). In Table 3.6, various crop residue attributes are described that lead to C and N sequestration, including combining maize and soybean crop residues, as in the intercropping agroecosystems from this study.

Table 3.6: Plant quality attributes chosen for different management purposes (Adapted from Cadisch and Giller, 2001)

Objective	Time Scale	Optimal Residue Quality
C and N sequestration	Immediate and long-term	High lignin content, low C/N
Erosion control	Immediate	High N content
Reduction of pollution	Immediate	Low soluble C and N
Nitrogen supply	Immediate and long-term	Mixed residues, low and high C/N, high lignin

For effective short- and long-term sequestration, a combination of high-lignin, high C/N ratio crop residues with low C/N, labile residues is recommended (Cadisch and Giller, 2001). Crop residue management can play a significant role in replenishing SOC and N, but plant materials contain a wide range of C and N compounds that have different decomposition rates affected by many soil factors (Ajwa and Tabatabai, 1994). Thus, changes in soil moisture, temperature, oxygen content, pH, nutrient availability, and other soil factors can alter the decomposition rates of plant residue and the mineralization rate of soil organic matter (Al-Kaisi et al., 2005).

3.3.2 Decomposition of maize and soybean litter

Crop residue becomes stabilized to soil humus at varying rates of decomposition and to varying degrees through time (Sollins et al., 1996). In this study, half-lives ($t_{½}$) and rates of decay (*k*) of the crop residue were measured over 312 days (Table 3.7). A single model of exponential decay was used to determine half-lives and rate constant values, which is the most widely used in decomposition studies over the short-term, including this study of under one year (Kurz-Besson et al., 2005). This model assumes that all material will be decomposed at the same rate, but is appropriate for this study because of the short duration of the sampling period. A two-compartment model of decay can be used for longer-term studies of more than one year which describes a rapidly-decomposing fraction of crop residue with short half-lives, followed by a more recalcitrant fraction of residue with very long half-lives (Bahri et al., 2008; Kurz-Besson et al, 2005; Sollins et al., 1996).

Table 3.7: Decay rate constant (k) and half-lives ($t_{½}$) of maize and soybean leaf and stalk residue in maize and soybean sole crop and 1:2 and 2:3 intercrop agroecosystems at Balcarce, Argentina. Litterbags were collected, dried and weighed monthly from October 2007 to July 2008.

Treatment	*k* (days^{-1})	$t_{½}$ (days)	r^2
Maize	0.0167 (0.0002) ^{a}	41.39 (0.37) ^{a}	0.8647
Soybean	0.0183 (0.0004) ^{b}	37.78 (1.16) ^{b}	0.8583
1:2 Intercrop	0.0177 (0.0003) ^{a}	39.07 (1.46) ^{a}	0.8867
2:3 Intercrop	0.0177 (0.0003) ^{a}	39.12 (2.35) ^{a}	0.8959

The *k* values and half-life values (n=3) followed by the same lower case letter are not significantly different at $P < 0.1$. Standard errors are given in parentheses.

Soybean sole crop residue had the shortest half-life of 37.78 days, which was significantly different than the other treatments at P<0.1. Crop residue from the 1:2 and 2:3 intercrop treatments had half-lives of 39.07 and 39.12 days, respectively. The longest half-life was for the maize sole crop residue, at 41.39 days. As expected, the fastest rate of crop residue decay (0.0183) was found in the soybean sole crop, which was significantly different than the other treatments. The 1:2 and 2:3 intercrop residues had the second fastest rate of decay (0.0177 days), followed by the maize crop residues (0.0167 days).

Half-lives and rate constant values from this study were compared to other studies of decomposition with maize and soybean crop residues. Mary et al. (1996) reported a *k* value for maize crop residue of 0.012 days, a slightly slower rate of decay than reported in this study. Increasing values of k indicate increasing rates of litter mass loss or nutrient release. Kalburtji and Mamolos (2000) compared the aboveground parts of soybean and maize crop residue and found that N was more easily released in soybean stems than maize stalks. Verma et al., (2005) compared half-lives of maize and soybean and found that soybean residue decomposed 10–24% faster than maize residue. These values correspond to results from this study, where soybean residue decomposed approximately 10% faster than maize residue. In an incubation experiment on maize and soybean residue decomposition Rampoldi et al. (2008) observed higher *k* values after 56 days for soybean residue, and the lowest *k* values for maize residue. These researchers attribute differences in C/N ratios to adequately explain the differences found. A higher content of N and a lower content of C favored significantly faster decomposition, corroborating the stimulating effect of N on microbial activity (Rampoldi et al., 2008).

The amount of crop residue remaining in litterbags after 312 days is presented (Figure 3.1). The least amount of residue remaining occurred with soybean litter, which is expected as soybean litter had the fastest rate of decomposition, the shortest half-life, and a low C/N ratio. This is followed by residue from the 1:2 intercrop, which had the second fastest rate of decay, and the second least amount of remaining residue. The greatest amount of crop residue remaining after the sampling period was maize litter, which had the slowest rate of decomposition, the longest half-life, and a high C/N ratio. In addition, maize litter had the least amount of N remaining at the end of the sampling period. The decrease in the % biomass remaining observed on day 149 is likely due to higher than average levels of precipitation during the two weeks prior.

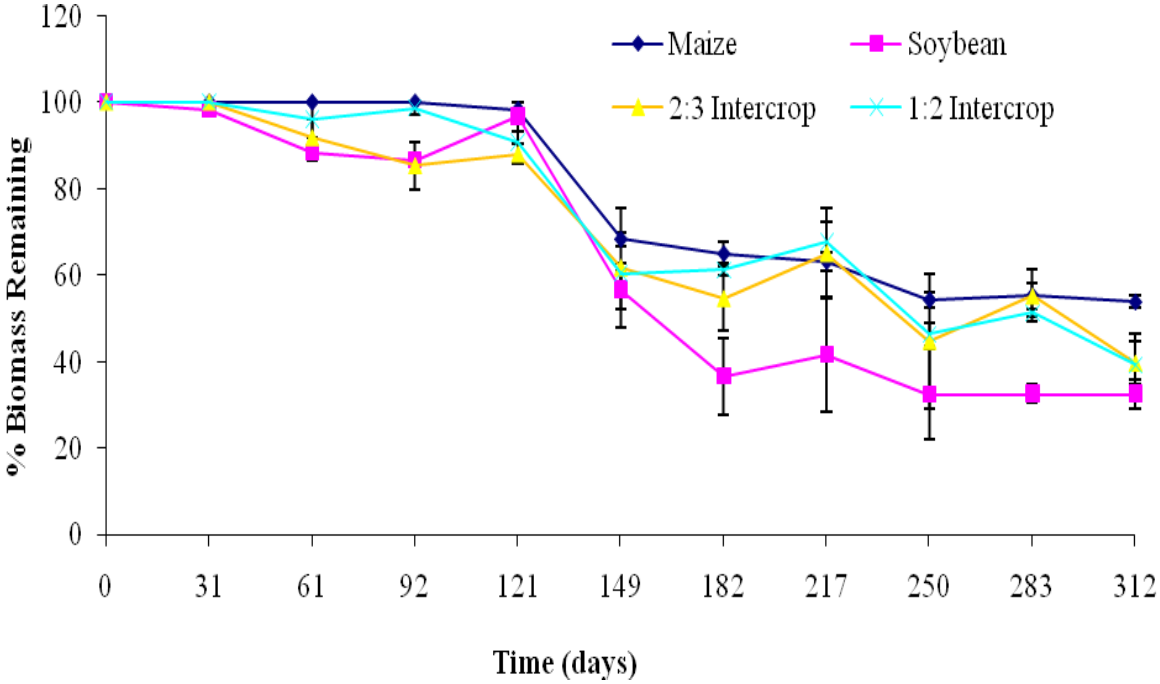

Figure 3.1: Leaf and stalk crop residue remaining after 312 days. Treatments were: maize sole crop, soybean sole crop, 1:2 maize:soybean intercrop, and 2:3 maize:soybean intercrop. Crop residue was dried for one week at 65 °C prior to placement in litterbags. Cobs and pods were previously removed. Day 1 corresponds to litterbag placement in the field on September 21, 2007. Error bars indicate standard error.

In a 105-day decomposition study, Chen et al. (2007) found that significantly higher N favours a decomposer community composition which enhances residue decomposition and SOM and humus formation. Similarly to Bahri et al. (2008), Ågren et al. (2001) found that maize residue is rapidly transformed into recalcitrant material. Crop residues with lower levels of available N, such as maize litter, decompose more slowly, because when all available N in polysaccharides is mineralized, the remaining maize litter is the recalcitrant lignin fraction. Although maize residues decompose slowly relative to soybean residues, they are an important contributor to long-term SOC build-up. Mann et al. (2002) found that when maize residue was removed at harvest from continuous no-till maize plots, SOC remained nearly unchanged over time, but increased about 14% in plots where residue was returned to the soil.

The concentration of C remaining in litterbags after 149 days from the four treatments is presented. The amount of C in crop residue from the collected litterbags declined from day 31 to day 149 in all treatments. After 31 and 61 days, the amount of C remaining in the maize sole crop was significantly lower than in other treatments. After 121 days, the amount of C in the soybean treatment was significantly lower than the maize treatment or either intercrop treatments. At the end of the sampling period, the greatest amount of C was found remaining in litterbags of the 2:3 intercrop treatments and the least amount of C was found remaining in the maize sole crop litterbags.

Table 3.8: Carbon remaining (g m^{-2}) in litterbags collected monthly after 149 days. Treatments were: maize sole crop, soybean sole crop, 1:2 maize:soybean intercrop, and 2:3 maize:soybean intercrop.

Day	Carbon (g m^{-2})			
	Maize	**Soybean**	**1:2 intercrop**	**2:3 intercrop**
31	92.35 (17.50) ^{a}	124.99 (8.22) ^{b}	157.68 (2.40) ^{b}	150.56 (10.72) ^{b}
61	102.59 (6.08) ^{a}	115.52 (3.49) ^{b}	149.52 (4.12) ^{b}	164.98 (4.10) ^{b}
92	96.67 (7.52) ^{a}	105.12 (10.69) ^{a}	168.48 (64.12) ^{a}	128.76 (6.72) ^{a}
121	78.69 (8.61) ^{a}	58.34 (9.50) ^{b}	106.91 (10.36) ^{a}	125.02 (4.50) ^{a}
149	72.10 (7.94) ^{a}	90.30 (5.89) ^{a}	112.40 (8.96) ^{a}	101.28 (3.72) ^{a}

Values followed by the same lower case letter are not significantly different ($P < 0.1$) between treatments. Standard errors are given in parentheses ($n = 3$).

Soybean litter is more quickly decomposed than maize residue, because it has a lower C/N ratio which makes it more available to microbial action. Kalburtji and Mamolos (2000) found fewer residues from soybean remained in the field after harvesting than maize. This means that N immobilization would be lower from soybean residues than from maize residues when incorporated into the soil. Blanco-Canqui and Lal (2004) stated that residues with low C/N ratios are rapidly decomposed, but its effect is transient, whereas slowly decomposing residues have a greater long-term effect on SOC sequestration. Drinkwater et al. (1998) stated that incorporation of leguminous crop residues increased both C and N retention in temperate agroecosystems. The systems studied included 1) a conventional maize-soybean rotation and 2) a maize system receiving N directly from legumes through incorporation of leguminous residue before maize planting. Higher levels of SOC in the second system were attributed to significant increases of soybean-derived C, which because it is quickly mineralized (Drinkwater et al., 1998).

The concentration of N remaining in litterbags from the four treatments is shown in Table 3.9. In general, the amount of N in litterbags declined from day 31 to day 149. After 61, 92 and 121 days, the amount of N remaining in the soybean treatment was significantly lower than the other treatments, suggesting that N mineralization occurred more quickly with soybean residue alone than either maize residue, or soybean and maize residue combined. After 149 days, there was a significantly lower amount of N remaining in litterbags from the maize sole crop than the other treatments. Although the most significant decreases in N occurred in the soybean sole crop, it was the maize sole crop that had the least N remaining in litterbags after the 149 days. Varvel & Peterson (1990) reported greater N concentrations for sole crop maize systems than in any other cropping system including a rotation with soybean (Kalburtji and Mamolos, 2000). In addition, they described losses of N from soybean crops due to increased microbial fixation of N_2O compared with other systems. This effect is likely because assimilation of N by plants is quite inefficient (approximately 50%), so much of the N is lost from the system from leaching in groundwater, from erosion, or via GHG emissions (Burket and Dick, 1997; FAO, 2001).

Table 3.9: Nitrogen remaining (g m^{-2}) in litterbags collected monthly after 149 days. Treatments were: maize sole crop, soybean sole crop, 1:2 maize:soybean intercrop, and 2:3 maize:soybean intercrop

Day	Treatment			
	Maize	**Soybean**	**1:2 intercrop**	**2:3 intercrop**
31	1.89 (0.13) ^{a}	1.73 (0.10) ^{a}	2.52 (0.12) ^{a}	2.80 (0.32) ^{a}
61	2.46 (0.10) ^{a}	1.92 (0.06) ^{b}	2.96 (0.16) ^{a}	3.12 (0.13) ^{a}
92	2.30 (0.06) ^{a}	1.79 (0.06) ^{b}	2.44 (0.04) ^{a}	2.80 (0.20) ^{a}
121	1.98 (0.10) ^{a}	1.41 (0.16) ^{b}	2.35 (0.16) ^{a}	2.78 (0.34) ^{a}
149	1.50 (0.13) ^{a}	2.24 (0.03) ^{b}	2.48 (0.12) ^{b}	2.52 (0.32) ^{b}

Values followed by the same lower case letter are not significantly different ($P < 0.1$) between treatments. Standard errors are given in parentheses ($n = 3$).

In a study of nutrient-release from maize, soybean and sunflower litter, Kalburtji and Mamolos (2000) reported that release of N is related to the C/N ratio of the soil and litter. A low N concentration in their study led to lower litter N concentration which resulted in a higher C/N ratio of the litter. The C/N ratio of the litter decreased with time. There was a negative relationship between the initial C/N and the *k* value of litter mass loss. Gosz et al. (1973) reported that when the C/N ratio had dropped to 30:1 during decomposition, net release of N to the soil started. The C/N ratio of maize and soybean litter remaining in litterbags in this study is presented (**Error! Reference source not found.**10).

Table 3.10: Ratios of C/N in litterbags after 149 days. Treatments were: maize sole crop, soybean sole crop, 1:2 maize:soybean intercrop, and 2:3 maize:soybean intercrop.

Day	Treatment			
	Maize	Soybean	1:2 intercrop	2:3 intercrop
31	48.8 ^{a}	72.2 ^{a}	62.6 ^{a}	53.8 ^{a}
61	41.7 ^{a}	60.2 ^{b}	50.5 ^{a}	52.8 ^{a}
92	42.0 ^{a}	58.7 ^{a}	69.0 ^{a}	46.0 ^{a}
121	39.7 ^{a}	41.4 ^{a}	45.5 ^{a}	45.0 ^{a}
149	48.1 ^{a}	40.3 ^{a}	45.3 ^{a}	40.2 ^{a}

Values followed by the same lower case letter are not significantly different ($P < 0.1$) between treatments. Standard errors are given in parentheses (n = 3).

Decomposition of litter that is incorporated into the soil proceeds more rapidly in materials rich in N and low in lignin recalcitrant C compounds (Kalburtji and Mamolos, 2000). The C/N ratio reported from other studies was approximately 30:1 for maize and soybean biomass, slightly lower than in this study. Mary et al. (1996) measured C/N ratio in maize biomass and found a 14:1 ratio in maize roots, and a 130:1 in stalks and leaves. The C/N ratio declined more rapidly in soybean litter than litter from maize sole crops or the intercrops. This data agrees with other researchers who found that litter mass more easily lost from soybean stems than maize stalks (Kalburtji and Mamolos, 2000). Maize and soybean residues are commonly managed in one of the following ways; they are incorporated into the soil, burned, or left on the surface. Straw of many crop species are rich in cellulose, hemicellulose and lignin. The chemical composition of straw is a key factor for the process of decomposition since high concentrations of polyphenols and lignin lead to low decomposition rate (Kalburtji and Mamolos, 2000). Results from this study corroborate that the C/N ratio of crop residue is one of the determining factors in residue decomposition.

Some researchers have shown that intercropping of maize or other cereal crops with soybean improves soil fertility when crop residues are incorporated (Nzabi et al., 2000). For example, Giller (1999) found that in a rotation of maize following soybeans, crop yield increased compared to sole crop maize due to effects of improved soil fertility. Likewise, in a study in Kenya, maize intercropped with soybeans yielded more than other sole cropped treatments (Nzabi et al., 2000). Shrader and Pierre (1966) observed significant increases in maize yield when grown in rotation with soybeans. These findings agree with trends observed in this study, because soybean sole crop plots had the highest SOC, but lower N than maize sole crop plots or the intercrop plots. The lower rate of decomposition of maize litter implies that these crop residues are stored in soil for a longer period than those from soybean, contributing to longer-term SOC and N levels (Kalburtji and Mamolos, 2000). Maize sole crop plots had the lowest SOC levels after one year of measurement, but approximately equal or greater N than soybean sole crop plots at each depth increment. The 2:3 intercrop plot had a higher k value than the maize sole crop, and greater SOC in the 0 – 10 cm surface layer, suggesting increased C and N input, and thus increased soil fertility. The 2:3 intercrop also had greater N than soybean sole crop plots at 0 – 10 cm and 80 – 120 cm despite a lower k value than soybean litter. This relationship was also reported by other authors (Kalburtji and Mamolos, 2000).

3.4 Summary and Conclusions

It was the objective of this study to quantify differences in soil biophysical characteristics, decomposition dynamics and crop residue input between sole crops and intercrops. There were no significant differences between treatments for SOC and TN. There was a significant decrease in SOC below 40 cm and a significant decrease in TN below 20 cm in all treatments. The C/N ratio was significantly greater at the 80 – 120 cm depth in all treatments, signifying very low N-availability at this depth. There was no significant difference in bulk density between treatments. Decomposition rates were measured between October 2007 and July 2008. There were no significant differences between treatments, however some trends were observed. The maize sole crop had the longest half-life and the slowest rate of decay, but had a greater input of crop residue-derived C and N than the other treatments. The soybean sole crop had the shortest half-life and fastest rate of decay, but lower inputs of crop residue-derived C and N. Reduced rates of residue decomposition and increased crop residue input results in accumulation of SOM, potentially increasing soil productivity and improving soil ecological functions. Both intercrop plots had intermediate rates of crop residue C and N inputs, and moderate rates of decay, thus these plots may accumulate greater SOM in time. Although maize sole crops had the highest input of C and N from crop residues over 149 days, the slow rate of decay means that maize residues have a more gradual impact on accumulation of SOM. On the other hand, the soybean sole crops quickly enhanced the SOM pool and had the second highest input of N from crop residue, but the effect is transient, as the N-rich residues are quickly decomposed. Both intercrop plots had intermediate input of C and the lowest input of N from combined crop residues. Intermediate rates of decomposition in both the intercrop plots indicate that a combination of recalcitrant crop residue with labile crop residue may contribute to both short- and long-term SOM.

Numerous researchers have shown that C and N sequestration must be quantified over the long-term. The study conducted at Balcarce was in its first year of implementation, thus the lower SOC and N in the intercrop plots compared to the sole crop plots do not indicate that they are less likely to accumulate in years to come. Continued research is needed to help determine the factors that affect the overall effects of crop residues on the soil environment. In addition, because almost no research exists on the environmental advantages of complex agroecosystems, continued long-term research is necessary. Results of this study are expected to help scientists to advise farmers and policy-makers on the benefits of maize-soybean intercrop agroecosystems to increasing SOM.

Chapter 4 – Greenhouse Gas Fluxes

4.1 Introduction

The amount of cropland worldwide has increased by 8% since the 1960s, to its current level of approximately 1.4 billion hectares (Smith et al., 2007). Agroecosystems, including intercrops, agroforestry, as well as rangelands and pastures, share 30% of the total land area in the US, 45% in Europe, and 38% worldwide (Fuhrer, 2003). Intensive agriculture has contributed to elevated greenhouse gases (GHG) (Haile-Mariam et al., 2008). Annual GHG emissions from agriculture are expected to increase in coming decades due to escalating demands for food and shifts in diet. Developing countries constitute the regions with the largest share of agricultural GHG emissions and are also the regions with the largest expected rates of increase in emissions (Smith et al., 2007). Globally, agricultural GHG emissions have increased by 14% between 1990 and 2005 (Smith et al., 2007). In addition, agricultural N_2O emissions are expected to increase by 35–60% up to 2030 due to the demand for increased fertilizer use and increased animal manure production (Smith et al., 2007). In Latin America, agricultural products are the main source of exports (Smith et al., 2007). As demands for food increase, annual emissions of GHG from agriculture may escalate further, but improved management practices may also help with emissions reduction (Smith et al., 2007).

Current GHG emission rates may escalate in the future due to population growth and a greater demand for food, which could result in higher emissions of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) as a result of increasing reliance on livestock production and nitrogen fertilizers. Implementation of new mitigation practices will be essential to prevent an increase in emissions from agriculture (Smith et al., 2007). In addition, soil carbon (C) may be more vulnerable to loss with climate change. In order to characterize this, soil-surface gas emissions must be measured. This is useful for several purposes; to assess the impact of land management on the atmospheric environment, and to develop and test predictive emission models (Rochette and Bertrand, 2007).

Carbon dioxide is released largely from microbial decay of plant litter and soil organic matter, methane is produced when organic matter decomposes in anoxic conditions, and N_2O is generated by the microbial transformation of nitrogen in soils and manures. This is enhanced where available nitrogen (N) exceeds plant requirements, especially under wet conditions (Smith et al., 2003). It is important that all three gases be investigated together to arrive at a verifiable estimate of net global warming potential. In temperate climates, much research has focused on agricultural emissions of CO_2, the largest contributor to atmospheric change, and N_2O, which has the highest global warming potential (Follett et al., 2005). Agricultural GHG fluxes are complex and heterogeneous, but the active management of agricultural systems offers possibilities for mitigation (Smith et al., 2003).

The C and N cycles operate in soils and both provide indices of the quality of the soil (Walker et al., 2001). The concentration of CO_2 in the soil and the exchange with the atmosphere depends on the rate of input of C to the soil and the respiration of microorganisms and plant roots (Nusier and Rousan, 2008). Soil aeration is a vital process because this controls how much O_2 diffuses into the soil (Nusier and Rousan, 2008). Plant roots and microorganisms utilize O_2 in the soil and release CO_2 during decomposition. The organic C is obtained from plants and soil microbial biomass (SMB) and oxidized to CO_2. Adequate soil respiration by plant roots and microbes requires that the soil itself be adequately aerated to allow diffusion of O_2 into the soil, and release of CO_2 to the atmosphere. This exchange of gases between the soil and the atmosphere changes under different soil temperature conditions or in response to different agricultural practices, such as tillage, that affect aeration. The CO_2 concentration in the soil air is often 10 to 100 times the level of 0.03% found in the atmosphere (Nusier and Rousan, 2008). Several studies suggest that increased CO_2 causes increased plant growth at low or high levels of nutrients (Nusier and Rousan, 2008).

The concentration of CH_4 in the atmosphere is responsible for 20% of the radiative forcing of the Earth, and has increased by 155% from about 700 ppb from 1750 to 1783 ppb in 2004 and is presently increasing at the rate of 5 to 13 ppb per year (Lal, et al., 2007; Mosier et al., 1997). Principal anthropogenic sources of CH_4 include fossil fuel exploitation, rice paddy cultivation, ruminant animals, biomass burning, and landfills. Soil is the only known biological sink for CH_4 (Henckel et al., 1999; Regina et al., 2007). The concentration of CH_4 in the Northern Hemisphere is about 100 ppb more than in the Southern Hemisphere, indicating either greater sources or lower sink strength in the Northern Hemisphere (Mosier et al., 1997). The concentration of CH_4 decreases proportionally with depth, suggesting that consumption is limited by the rate of diffusion of CH_4 into soil (Mosier et al., 1997).

Agriculture is considered to be responsible for about two-thirds of the anthropogenic sources globally. Aerobic soils oxidize 10–15% of annual CH_4 emissions to the atmosphere (Mosier et al., 1997). Dunfield et al. (1993) described the microbial pathways of CH_4 oxidation in soil; organisms that consume CH_4 are methanotrophs. All known methanotrophs are obligate aerobes, because the enzyme responsible for the initial step in CH_4 oxidation is a methane monoxygenase (MMO) which requires molecular O_2. Research has shown that CH_4 and NH_3 are competitive substrates for most MMO, which indicates that methanotrophs may contribute to nitrification (Mosier et al., 1997). The sink for CH_4 in soils depends on moisture and N levels (Johnson et al., 2007).

Nitrogen is frequently the most limiting element to terrestrial plants, and the rate of mineralization of this element profoundly affects primary productivity (Liu et al., 2007). Nitrogen is fixed from the atmosphere to the soil by bacteria, such as *Azotobacter* spp. and *Rhizobium* spp., which are found in the root nodules of legumes (Walker et al, 2001). Nitrogen is converted to ammonia, which then forms ammonium ions (NH_4^+) in soil. Ammonium is oxidized to nitrite (NO_2^-) and nitrate (NO_3^-) by soil bacteria, through nitrification. Plants and autotrophic bacteria use NH_4^+ and NO_2^- to synthesize organic compounds, such as amino acids, and ultimately converted to nitric oxide (NO), and N_2O by denitrification, and released back to the atmosphere. Nitrous oxide is an obligatory intermediary product of denitrification, and also as a by-product of nitrification. In order to meet the food demands of an expanding human population, increasing amounts of inorganic N fertilizers are applied to agricultural lands, which represents the largest anthropogenic input of N (Ding et al., 2007). This increased N-loading to soil increases the potential for N_2O emissions.

Soil management practices are one of the major factors that can influence the soil–atmosphere exchange of GHG which contribute to global climate change (Smith et al., 2003). Measurements of soil gas fluxes for different tillage treatments and cropping systems are, therefore, important for identifying management practices that can positively impact GHG emissions. However, the magnitude and accuracy of determination of gas emission vary spatially and gas emission itself is affected by such factors as seasonal climatic conditions, air and soil temperature, sampling frequency, and cropping systems (Rochette and Bertrand, 2007; Smith et al., 2003). Sey et al. (2008) describe in situ GHG flux measurements as relatively labor intensive, but detailed characterization of soil gases is important. Sampling in situ provides information about where GHG are produced within the soil profile and at what point in the growing season, and it allows more accurate prediction of feedbacks at the ecosystem level of C and N balances resulting from agricultural practices such as tillage, fertilizer applications, and cropping systems. As a result of this extensive variability in sequestration potential and emissions associated with changes in agricultural practices, there is a need to evaluate both short- and long-term emissions that are region- or soil-specific (Omonode et al., 2007).

No research has yet evaluated GHG emissions from intercrop agroecosystems. Therefore, the purpose of this study is to quantify and compare GHG emissions in sole crop and intercrop agroecosystems. The specific objectives of this study are to quantify emissions of CO_2, CH_4, and N_2O throughout the growing season in the maize sole crop, soybean sole crop, the 1:2 intercrop and the 2:3 intercrop agroecosystems.

4.2 – Materials and Methods

Hutchinson and Livingston (2001) describe the chamber-based technique as a valuable and cost-effective approach for studying gas fluxes between the soil and the atmosphere. Although to some degree the chamber interferes with natural gas flux from the soil by creating an artificial environment, specific chamber design may minimize such differences (Xu et al., 2006). To maintain equilibrium and avoid variations in pressure between the static pressure of the inside of the chamber and the ambient air outside the chamber, a vent tube is connected to the chamber lid. Thus, disturbances in pressure associated with chamber deployment and sampling, or with changes in the temperature of the headspace are eliminated (Hutchinson and Livingston, 2001). Any CO_2 loss through the vent tube is not likely to be significant if proper internal diameter and length of the vent tube are considered. Diffusion loss to be negligible if the vent tube is designed properly; for example, a vent tube with a diameter of 9 mm and a length of 15 cm, the loss was less than 0.04% of the total flux over a 30-min measurement period (Hutchinson and Livingston, 2001). In addition, wind can affect soil CO_2 efflux by enhancing the mixing of the atmosphere and removing respired CO_2 accumulated at the soil surface. In a non-steady state closed chamber-based soil flux system, although it is well mixed, the chamber CO_2 concentration cannot be maintained at ambient levels. This is because the chamber CO_2 concentration must be allowed to rise in order to compute the slope of dCO_2/dt, which is needed to calculate CO_2 flux. Thus, the soil CO_2 diffusion gradient is altered and may cause underestimation in F_{CO2}. To minimize this error, nonlinear curve fitting is used to account for the change in the gradient (Hutchinson and Livingston, 2001). Shortening the measurement period also is used sometimes to reduce the impact of increasing chamber CO_2 concentration, especially when data are analyzed by linear regression (Xu et al., 2006). Hutchinson and Livingston (2001) stated that by taking multiple samples throughout deployment period, they were able to account for non-linear accumulation of the target gas and to test the statistical credibility of the resulting estimate of the flux. Chamber techniques have been used to estimate soil-surface gas emissions for many decades and remain the most commonly used approach. Kabwe et al. (2002) showed that the dynamic closed chamber technique yielded accurate measurements of fluxes. They are relatively inexpensive to build and use, and can be adapted to a wide range of field conditions and experimental objectives (Rochette and Bertrand, 2007).

4.2.1 Field Sampling

Two GHG gas sampling chambers per repetition in each block were placed between maize and soybean rows at a distance of 2 m from the plot edge to minimize any border effects. Sampling took place with a gas–tight syringe and samples were drawn from the chamber at time 0, 15 and 30 minutes. Greenhouse gases were always sampled between 8 am to 12 pm and 1 pm to 2 pm to minimize diurnal variation. Air samples of 25 ml were injected into 12 ml evacuated vials (Labco Ltd., High Wycombe, UK). Soil moisture content and temperature to 10 cm were noted at sampling time using the HH2 Moisture Meter with an ML2 ThetaProbe. Gas samples were transported to Canada and gas concentration of CO_2, CH_4 and N_2O were determined by gas chromatography (GC)

(Agilent 6890N). Gases were separated on an HP high-speed capillary column (25m × 0.32mm × 10μm) held at a constant temperature of 30 °C. The temperatures and injection volume were adjusted to obtain optimum peak resolution.

Emissions of GHG (μg m^{-2} h^{-1}) were measured over the course of the 2007-2008 growing season at Balcarce, Argentina. Measurements of GHG fluxes from the four treatments were taken before seeding of maize or soybean (October 10, 2007), bi-weekly throughout the growing season (October 23, 2007 to April 8, 2008), then after crop harvest (April 18, 2008).

4.2.2 Chamber Design

Greenhouse gas sampling chambers were constructed from a non-reactive PVC (Figure 4.1) and based on a design by Omonode et al. (2007); Parkin et al. (2003) and Rochette and Bertrand (2007). The chamber had a 10 cm collar in the soil and an above-ground height of 15 cm. A vent tube was inserted into the insulated chamber lid to maintain pressure equilibrium. The headspace volume was 0.0118 m^3.

4.2.3 Flux Calculation and Statistical Analysis

To account for problems associated with GC drift, samples from individual chambers were run in sequence (t_0, t_{15}, t_{30}) and standards were run periodically throughout the sample run (every 20 samples). The gas standard had the following composition: 99 ppm CO_2, 5 ppm CH_4, and 10 ppm N_2O. Non-linear regression was used to calculate the slope of the concentration vs. time. Emissions calculated from the rate of change of the concentration (dCO_2/dt) of the analyte of interest (CO_2, CH_4 and N_2O) in the chamber headspace were converted from μL analyte L^{-1} to μL analyte m^{-2} hr^{-1} by multiplying the slope by the chamber volume (L) and dividing by the chamber surface area (m^2)

(Parkin et al., 2003). To convert the flux value from a volumetric basis to a mass basis, the ideal gas law is applied (Eq. 4.1).

$$PV = nRT \tag{4.1}$$

Where R, the gas law constant, is expressed in L atm Mol^{-1} K^{-1}, and pressure (P), volume (V), moles (n) and temperature (T) have corresponding units of Atm, L, Mol, and K, respectively. Applying equation 4.1 requires calculation of actual air temperature at the time of measurement, which was achieved by using data from the INTA meteorological station Figure 4.2). Temperature values were calculated as 273 K + daily recorded air temperature (in ˚C). Resulting values (µmol m^{-2} h^{-1}) were converted to µg m^{-2} h^{-1} by multiplying by the molecular weight of the analyte.

Greenhouse gas flux data were examined for homogeneity of variance and normality. Data were analyzed using the univariate general linear model (SPSS version 16.0). Significantly different main effects were further analyzed using Tukey's multiple comparison test. Cumulative production rates of CO_2, CH_4, and N_2O were calculated by taking the growing season mean emissions values, and using the pre-seeding production rates values for the non-growing season.

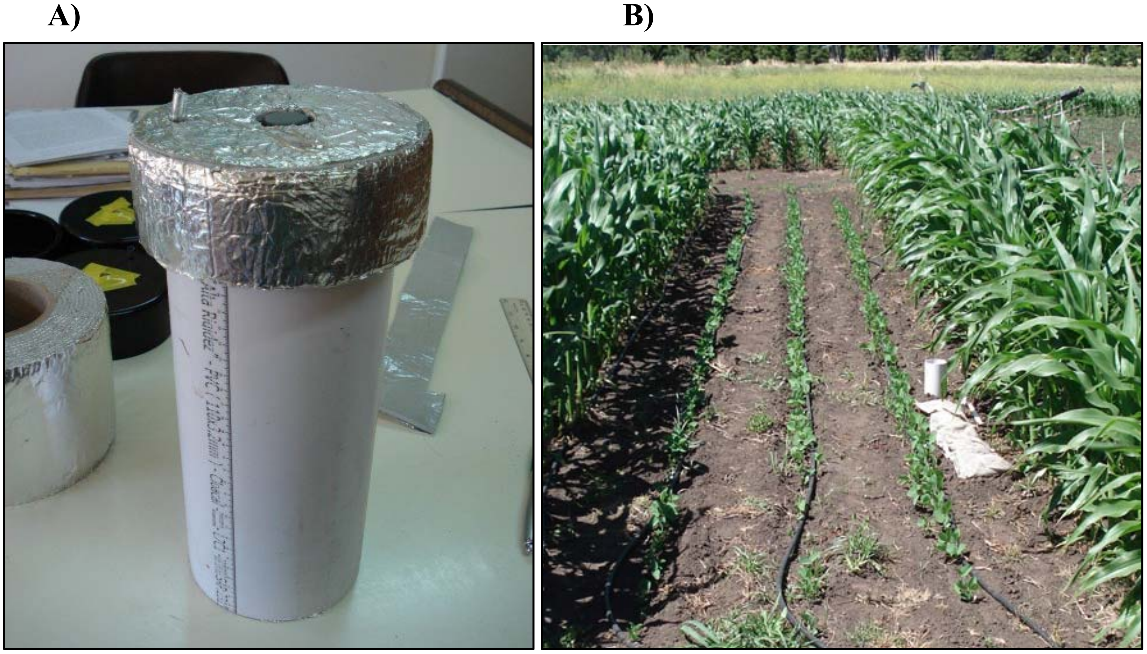

Figure 4.1: PVC gas flux chamber; A) construction of chamber, including anchor, showing dimensions and insulated lid with septa and vent tube, and B) chamber anchor placement between rows in the 2:3 intercrop treatment.

4. 3 – Results and Discussion

There were no significant differences in CO_2 emissions between treatments (Figure 4.3). The treatment that had the highest CO_2 emissions was the 2:3 intercrop, the highest value of which was 406.26 μg $m^{-2}h^{-1}$. The soybean sole crop had the next highest CO_2 emission value followed by the maize sole crop, then the 1:2 intercrop. Higher CO_2 emission rates for all treatments were observed during the non-growing season near harvest time, due to the large input of senescent crop residue. This is expected, as greater residue C-input would contribute to greater CO_2 fluxes from the soil. These results correspond with other authors, who attribute CO_2 emissions to the incorporation of crop residues during this period, and also due to differences in the quality of the residues, determining nutrient release to the soil (Baggs et al., 2006; Oorts et al., 2007).

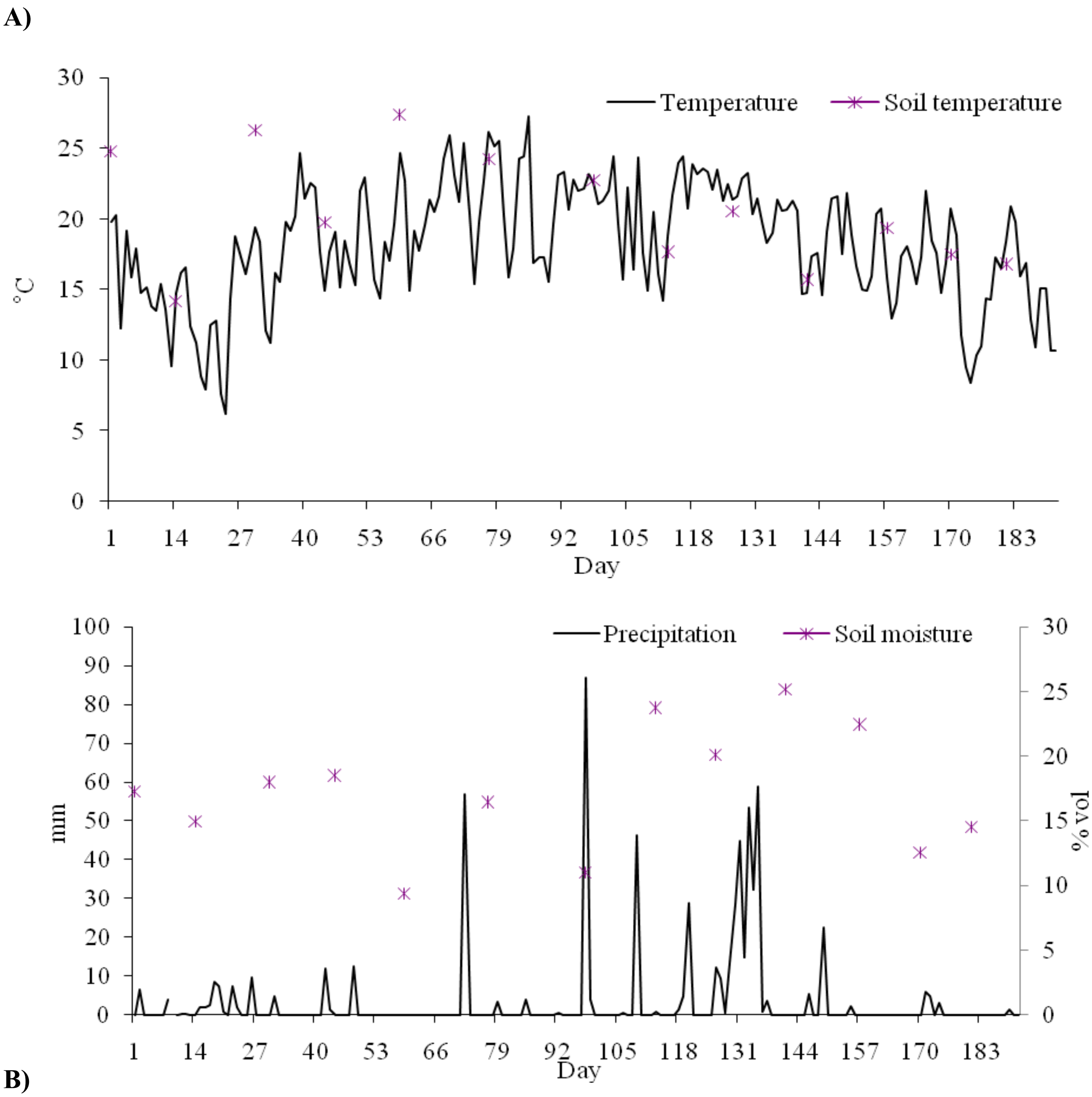

Figure 4.2: Weather data measured daily from the INTA meteorological station; A) mean air temperature (°C) and B) precipitation (mm). Soil temperature (°C) and soil moisture (% vol) were measured biweekly with GHG sampling at 10 cm from October 23, 2007 to April 18, 2008 using the HH2 Moisture Meter. Day 1 corresponds to the first sampling day on October 23, 2007.

Other researchers report that CO_2 emissions did not differ significantly between maize and soybean treatments, but observed trends in emissions throughout the growing season (Oorts et al., 2007). Seasonal differences are attributable to increases in root respiration, crop residue inputs, weather, planting and harvesting events. In the system at Balcarce, there was an increase in CO_2 emissions between pre-seeding (PS) and day 1 due to planting of short- and long-cycle maize five days earlier. Another increase in CO_2 emissions on day 30 may be due to planting of soybean on day 28. The slightly increased CO_2 on day 141 is attributable to both precipitation events and higher residue input.

Rastogi et al. (2002) described the relationship between soil moisture and CO_2 emissions. As precipitation increases, soil moisture increases, inducing greater soil respiration by microbes and thus greater production of CO_2. In late February and early March 2008, higher than average precipitation events at the study site likely stimulated increased microbial activity in the soil, causing greater CO_2 evolution on day 141. In addition, crop harvest of maize and soybean began on day 155, which would also have stimulated increased nutrient release to the soil and thus increased microbial activity. Oorts et al. (2007) studied CO_2 emissions in a long-term maize-wheat rotation under conventional and no-till in northern France. Similar to the results of this study, these resesearchers found that there was an increase in CO_2 emissions before crop harvest. Adviento-Borbe et al. (2007) found low CO_2 fluxes after planting but increased fluxes throughout May to July with increasing temperature and progressing growth of maize, reaching a maximum of 500 $\mu g\ CO_2\ m^{-2}\ h^{-1}$. In this study, the highest peak during the mid-season occurred at approximately 360 $\mu g\ CO_2\ m^{-2}\ h^{-1}$. Similarly, higher peaks in CO_2 values were observed during warmest days in the spring (Verma et al., 2005). The linear relationship between CO_2 emissions soil temperature and volumetric soil moisture at 10 cm was also observed by Ellert and Janzen (2008) in southern Alberta. These researchers found a significant increase in CO_2 emissions in the soybean sole crop than in rotation with maize. In their two-year study, Omonode et al. (2007) found mean emissions from maize sole crop were 16% higher than maize in rotation with soybean. Similar to results from this study, CO_2 emissions during the growing season were highest during the period when crop residue input was the greatest (Omonode et al., 2007).

Emissions of CH_4 ($\mu g\ m^{-2}\ h^{-1}$) did not show a significant difference between treatments (Figure 4.4). The highest emissions of CH_4 was observed in the 2:3 intercrop, with 0.76 $\mu g\ m^{-2}\ h^{-1}$ immediately following the planting of maize. Emissions of CH_4 were highest for all treatments on day 1 of GHG sampling. In general, all treatments provided a sink for C. These results are consistent with other studies. For example, Ellert and Janzen (2008) determined that CH_4 uptake was barely detectable in maize sole crop systems amended with soybean residues for an N source. Verchot et al. (2008) found a net CH_4 sink in soil in an agroforestry system in Brazil where maize was grown with leguminous tree species. Some researchers found that CH_4 uptake was less sensitive to crop type, field management practices, and fertilizer application rates, and more strongly influenced by tillage management (Koga et al., 2004). However, Verchot et al. (2008) suggested that CH_4 uptake increased in dry conditions. This is expected, because greater soil moisture would contribute to the anaerobic conditions which are necessary for increased methanotroph activity and CH_4 production.

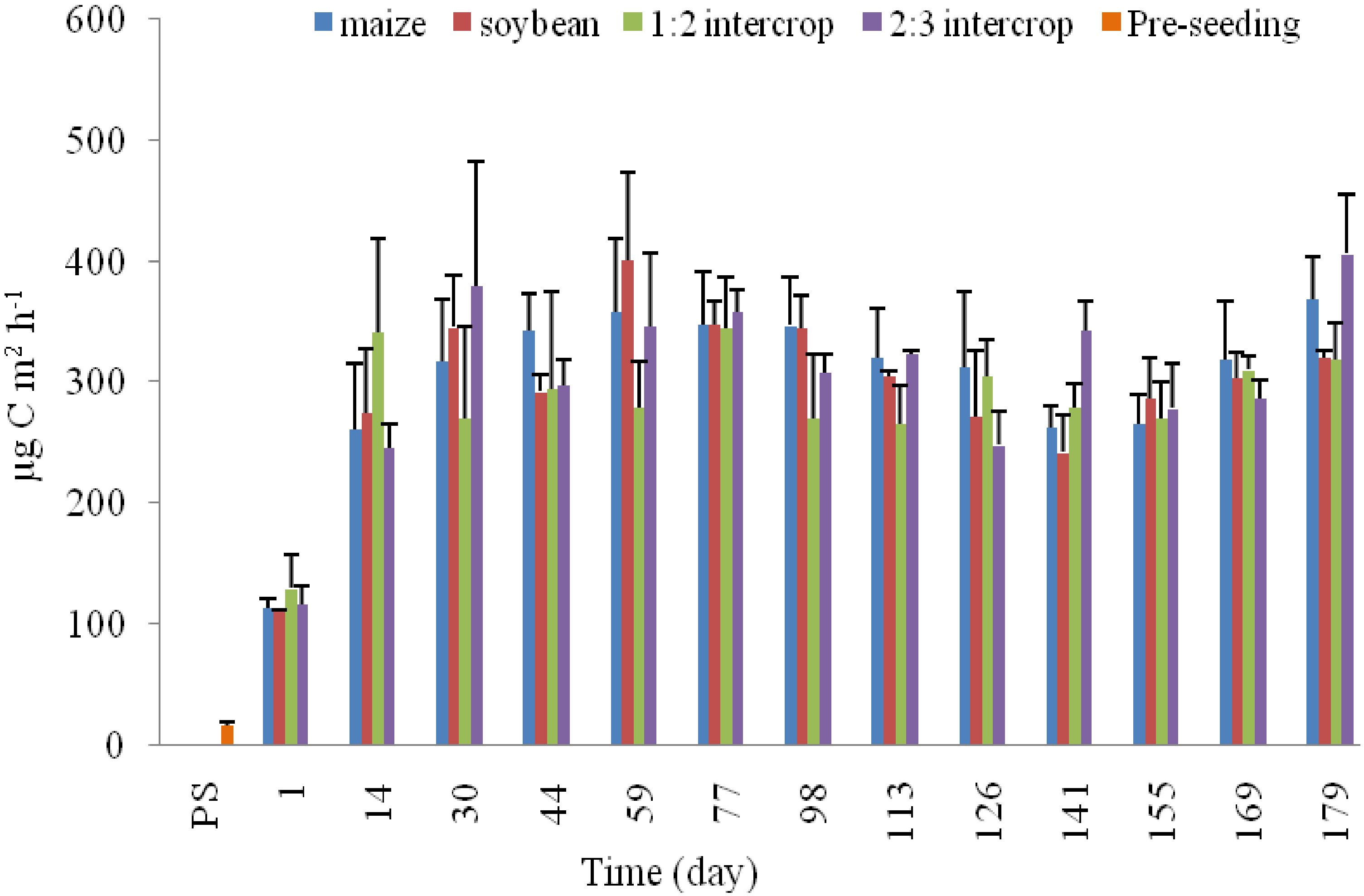

Figure 4.3: CO$_2$ emissions (µg m^{-2} h^{-1}) during the 2007-2008 growing season at Balcarce, Argentina. PS refers to measurements taken at pre-seeding. Sample day 1 corresponds to October 23, 2007. Error bars indicate standard error.

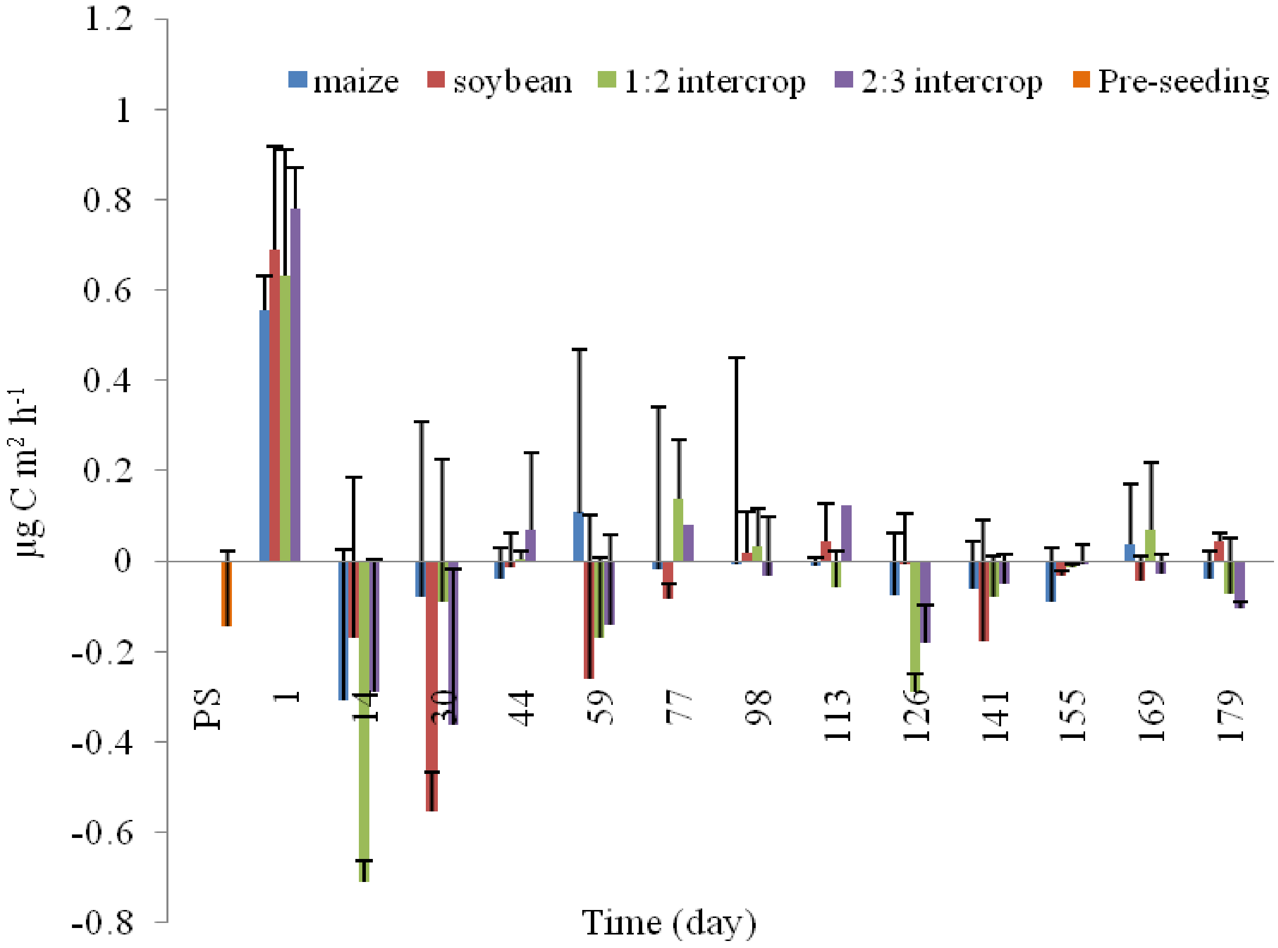

Figure 4.4: CH_4 emissions (μg m^{-2} h^{-1}) during the 2007-2008 growing season at Balcarce, Argentina. PS refers to measurements taken at pre-seeding. Sample day 1 corresponds to October 23, 2007. Error bars indicate standard error.

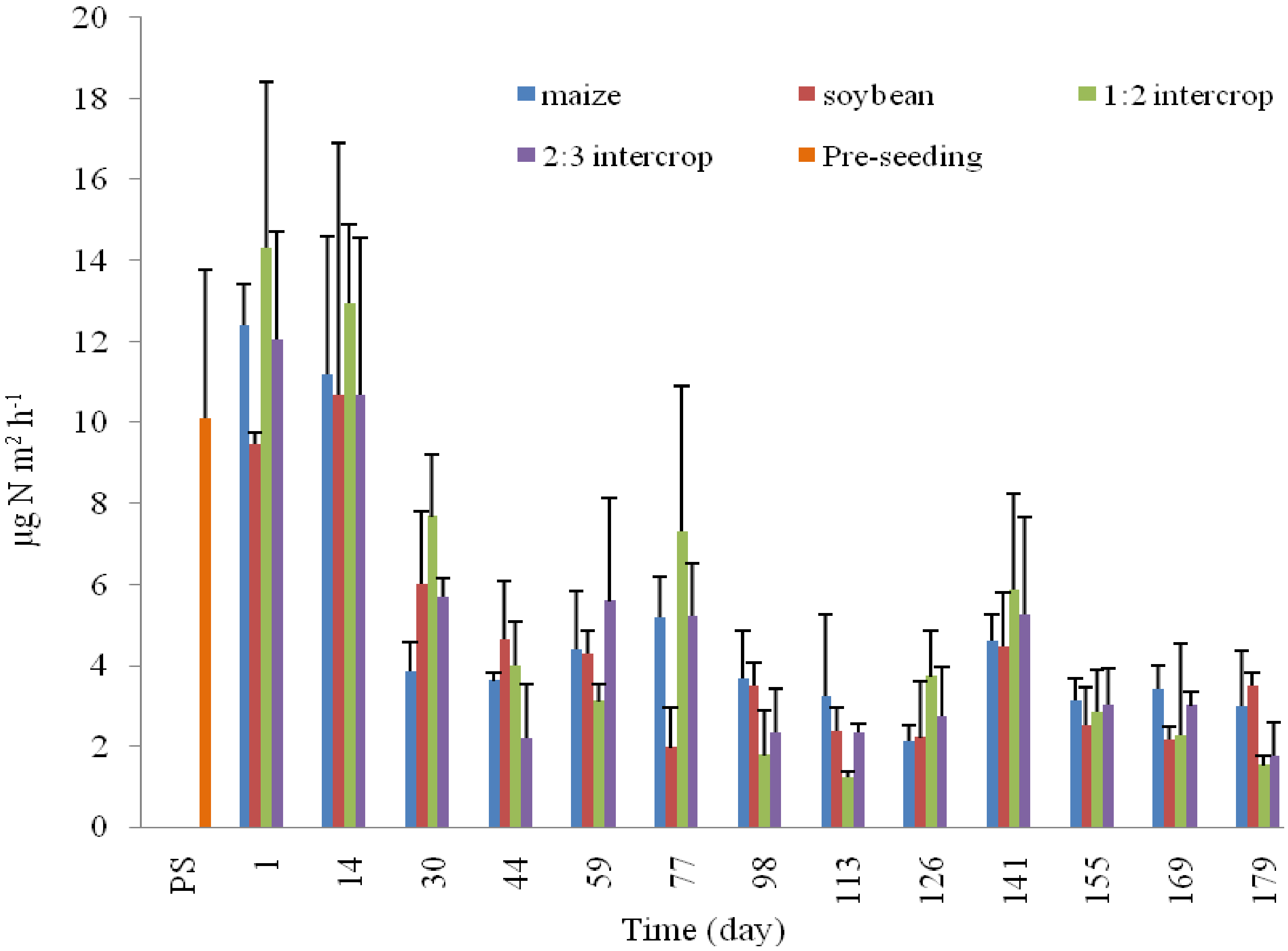

Figure 4.5: N_2O emissions (μg m^{-2} h^{-1}) during the 2007-2008 growing season at Balcarce, Argentina. PS refers to measurements taken at pre-seeding. Sample day 1 corresponds to October 23, 2007. Error bars indicate standard error.

Some systems with improved CH_4 uptake adversely affected N_2O emissions (Koga et al., 2004; Müller and Sherlock, 2004). Mosier et al. (1997) reported inhibition of CH_4 uptake by inorganic N. Some researchers have stated that factors governing a soil's CH_4 sink capacity include moisture and N levels (Johnson et al., 2007). As such, emissions of N_2O ($\mu g\ m^{-2}\ h^{-1}$) were measured in this study (Figure 4.5). However, no significant differences between treatments were detected. The highest N_2O emissions were observed at PS and during the first two weeks of measurements, from October until early November. After this time, in mid-November, N_2O emissions decreased and remained low throughout the rest of the growing season. Müller and Sherlock (2004) found a similar response in N_2O emissions consistent with the trend observed in this study. Ding et al. (2007) found consistently low N_2O emissions throughout the growing season, with only slight emissions peaks following fertilizer application.

In this study, the N_2O emission peaked at 14 $\mu g\ N_2O\ m^{-2}\ h^{-1}$ with a soil temperature of 15°C. Emissions then tailed off and remained near zero until later in the season when the soil temperature at 5 cm depth reached 10°C or more (Ding et al., 2007). In general, greater than 80% of N_2O emissions in temperate grassland ecosystems occur at higher soil temperatures, at high soil moisture, with easily metabolized C, and N-fertilizer application to the soil (Müller and Sherlock, 2004). Soil temperatures between approximately 10 and 15 °C positively influence nitrification and denitrfication rates (Müller and Sherlock, 2004). At the study site, emissions peaked within the first two weeks at approximately 14 $\mu g\ N_2O\ m^{-2}\ h^{-1}$, after which the soil temperature increased above 15 °C, suggesting reduced emissions would occur. The N_2O emissions after day 14 are markedly reduced throughout the rest of the growing season. Similar to the response observed with CO_2 emissions, a slight increase in N_2O emissions on day 141was caused by higher than average rainfall in February and March. Some researchers reported that rainfall events stimulated brief peaks in N_2O emissions, showing that soil moisture is an important factor, but cannot be used as the sole predictor in emissions (Müller and Sherlock, 2004). Baggs et al. (2006) found that N_2O emissions typically increased after addition of residues, since OM is readily decomposed and N_2O is subsequently produced during nitrification and denitrification.

Mosier et al. (2006) describe a distinct seasonal pattern in N_2O fluxes. The greatest N_2O emissions consistently occurred within days or weeks of fertilization and planting (Mosier et al., 2006). During the peak growing season, N_2O fluxes appeared to be less influenced by precipitation events, irrigation and forage harvest (Ellert and Janzen, 2008). Other researchers did not detect significant differences between maize and soybean treatments, but found a seasonal correlation in N_2O-N flux (Verchot et al., 2008). Roelandt et al. (2005) found a significant positive correlation between spring and summer temperatures and N_2O evolution.

Annual emissions rates of CO_2, CH_4 and N_2O were estimated and expressed as kg ha^{-1} yr^{-1} or g ha^{-1} yr^{-1} (Table 4.1). No significant differences between treatments were observed for CO_2. The highest annual emissions of CO_2 were from the 2:3 intercrop at 703.20 kg CO_2 ha^{-1}, followed by the maize sole crop, and then the soybean sole crop. The system with the least annual CO_2 is the 1:2 intercrop, with 657.45 kg CO_2 ha^{-1}. A significant difference in CH_4 in the soybean sole crop and the 1:2 intercrop compared to the 2:3 intercrop and the maize sole crop was observed. The largest CH_4 sink was in the 1:2 intercrop, consuming 107.17 g CH_4 ha^{-2} annually. The next largest CH_4 sink was the soybean sole crop, followed by the 2:3 intercrop and then the maize sole crop consuming only 4.61 g CH_4 ha^{-1}. No significant differences between treatments were observed for N_2O. The highest annual emissions of N_2O were from the 1:2 intercrop with 12.72 kg ha^{-1} yr^{-1} followed by the soybean sole crop and the maize sole crop. The lowest emissions of N_2O were in the 2:3 intercrop, with 6.81 kg ha^{-1} yr^{-1}. Annual emissions of N_2O were inversely proportionate to CO_2 emissions.

Verma et al. (2005) observed a flux from maize sole crops of approximately 600 kg ha^{-1} yr^{-1} CO_2 over three years, which approaches the values from the observations of this study. Oorts et al.

(2007) observed a higher annual flux of CO_2 in a maize-wheat rotation over 32 years. They estimated approximately 1150 kg ha^{-1} yr^{-1} from the system. Meyer-Aurich et al. (2006) found that the integration of soybean in rotation with maize results in a mitigation of 726 kg CO_2 ha^{-1} yr^{-1}. These values are much higher than the values from this study. The potential difference in CO_2 mitigated between the 1:2 intercrop and the maize sole crop is 45.68 kg CO_2 ha^{-1} yr^{-1}, and only 29.43 kg CO_2 ha^{-1} yr^{-1} with the soybean sole crop. The 2:3 intercrop would produce a greater CO_2 flux than either sole crop.

Table 4.1: Annual emissions of CO_2, CH_4, and N_2O from soil in maize, soybean, 1:2 intercrop and 2:3 intercrop treatments for the 2007-2008 growing season in Balcarce, Argentina.

	Treatment			
	Maize	**Soybean**	**1:2 intercrop**	**2:3 intercrop**
CO_2 (kg ha^{-1} yr^{-1})	703.13 (78.35) ^{a}	686.88 (57.91) ^{a}	657.45 (82.88) ^{a}	703.20 (62.50) ^{a}
CH_4 (g ha^{-1} yr^{-1})	-4.61 (29.62) ^{a}	-96.49 (25.38) ^{b}	-107.17 (25.55) ^{b}	-24.12 (24.00) ^{a}
N_2O (kg ha^{-1} yr^{-1})	11.11 (2.08) ^{a}	10.06 (3.02) ^{a}	12.72 (3.28) ^{a}	6.81 (1.49) ^{a}

Values followed by the same lower case letter are not significantly different ($P < 0.05$) between treatments. Standard errors are given in parentheses (n = 3). Negative values indicate a net sink.

The net annual fluxes of CH_4 are small, however significant differences between treatments were observed. Cultivated soils generally show much lower CH_4 uptake rates than soils under native vegetation (Mosier et al., 1997). In Costa Rica, Keller et al. (1993) found that land-use conversion of forest to pasture transformed a net sink of 3.3 g CH_4 m^{-2} y^{-1} to a net source of 1.8 g CH_4 m^{-2} y^{-1}. Gregorich et al. (2005) concluded that uptake by agricultural lands in eastern Canada could only offset about 4% of the CH_4 produced by dairy cows in the region. In this study, all four agroecosystems were net sinks for CH_4, although conversion from a 1:2 intercrop to a maize or soybean sole crop reduces the CH_4 sink potential in soil.

The tightly coupled biogeochemical cycles of C and N suggest that increases in SOM will cause increases in N_2O emissions due to greater soil N and microbial activity (Li et al., 2005). However, Ding et al. (2007) found that the lack of available labile SOC and CO_2 released from maize roots seemed to reduce the synthesis and activity of nitrifiers, reducing N_2O emissions in maize plots compared to soybean. In a two-year trial in Iowa, N_2O emission from soil under soybean was 4.4 kg N_2O ha^{-1} yr^{-1} and from soil under corn was 9.6 kg N_2O ha^{-1} yr^{-1} (Johnson et al., 2007). Interestingly, at the study site, emissions of N_2O were also reduced under soybean sole crop compared to maize sole crop. Data from Drinkwater et al. (2007) suggested that C sequestration in the field is offset by N_2O emissions associated with fertilizers. However, some researchers maintain that it is N_2O emissions that are offset by C sequestration in agricultural soils (Meyer-Aurich et al., 2006).

The majority N_2O emissions are estimated based on the soil microbial processes of nitrification and denitrification (Ellert and Janzen, 2008). The main cause of agricultural increases in N_2O emission to the atmosphere is the application of N fertilizers and manures. Soil N_2O emissions indicate an inefficient use of N in agricultural soils, which then enhances the radiative forcing in the atmosphere (Ellert and Janzen, 2008). In general, N_2O emission increases with increased N-inputs (Gregorich et al., 2005; IPCC, 2001). A positive relationship was reported between N fertilizer applied and N_2O emissions in Eastern Canada (Gregorich et al., 2005). In contrast, annual N_2O emission and N fertilization were poorly correlated on cropland when comparing reports from different management systems (Roelandt et al., 2005). Clearly, the uncertainty of emissions is large, especially when measured using manually operated chambers, the most commonly used system (Chapuis-Lardy et al., 2007).

4.4 – Conclusions

The objective of this investigation was to evaluate whether intercrop agroecosystems emitted fewer GHGs than sole crops. The results indicate that there is no significant difference in emissions in intercrop agroecosystems or sole crop agroecosystems throughout the growing season after one year of measurements. Emissions of GHG throughout the growing season tended to be correlated with precipitation events, temperature, soil moisture and temperature, and crop residue input. These factors increased decomposition and thus soil respiration rates, increasing GHG fluxes to the atmosphere.

The agroecosystem with the fewest cumulative annual C emissions to the atmosphere was the 1:2 intercrop. The 2:3 intercrop had the greatest C emissions and the lowest N_2O fluxes. There is a great deal of inconsistency among researchers regarding the tradeoffs of CO_2 mitigation and N_2O production. Some scientists argue that because the global warming potential of N_2O is so much greater than that of CO_2 or CH_4, there is a reduced advantage of C sequestration in the soil due to fluxes of N to the atmosphere. Others argue that even though legumes contribute considerably to emissions of N_2O, these are more than offset by fewer emissions from reduced fertilizer use and manufacturing of the fertilizer, and increased C-input to the soil.

The soil microbial processes involved in GHG emission to the atmosphere are not well understood. While the only sink for CH_4 is in soil, little information is known about the methanotrophic community involved and their interaction with microbes that contribute to other GHG such as CO_2 and N_2O. Soil disturbance contributes to aeration and thus the magnitude of the soil sink for CH_4. However, research shows CH_4 uptake rates for systems where soils are cropped with different crops and at different intensities are limited. In this study, no significant differences were detected on individual sampling days in the magnitude of the CH_4 sink, indicating that crop type may play a less significant role.

The influence of residue quality, as determined by factors including C/N ratio, on GHG emissions have yet to be determined in complex agroecosystems such as intercrops and agroforestry systems. Detailed studies are necessary to determine C and N transformations in soil and the related GHG emissions in complex agroecosystems. Investigations using isotope-labeling studies are necessary under field conditions to determine the process of GHG emissions. Researchers must be cautious when reporting emissions because they are variable in space and time.

Chapter 5 – Final Summary and Conclusions

5.1 Summary and Overall Conclusions

The purpose of this research was to determine whether intercrop agroecosystems sequester more C and N in soil and emit fewer greenhouse gases (GHG) than sole crop systems. Because no research had yet investigated the potential of intercrop agroecosystems in soil carbon (C) and nitrogen (N) sequestration and GHG mitigation, a better understanding of dynamics in these systems was needed. The specific objectives of this study were to compare intercrop and sole crop systems and determine which system was better able to sequester C and N and mitigate GHG. The four treatments studied included a maize sole crop, a soybean sole crop, a 1:2 intercrop and a 2:3 intercrop. It was hypothesized that soils of the intercrop agroecosystems would have greater soil organic C (SOC) than the soybean sole crop and greater soil total N than the maize sole crop. In addition, it was hypothesized that maize sole crops would have the lowest C and N-residue input, while soybean sole crops would have the greatest C and N-residue input.

An evaluation of soil biophysical characteristics showed that there were no significant differences in SOC and total soil N between treatments. Significant changes in the C/N ratio, SOC and total soil N were correlated with soil depth, and were not influenced by crop type. Some researchers suggested that changes in management practices aimed at increasing soil organic matter (SOM) may in fact decrease SOC and total soil N in the first 5 years. In addition to SOC and soil total N, bulk density and C/N ratio were characterized. There was no significant difference in bulk density between treatment or depth. The C/N ratio was significantly higher in all treatments at the 80 – 120 cm depth, except in the 2:3 intercrop.

The next part of the study investigated C- and N-input from crop leaf, shoot and root biomass. From combined total crop residue, the greatest C-input occurred in the maize sole crops. The 1:2 intercrop had the next highest input followed by the 2:3 intercrop and lastly the soybean sole crop. The greatest N-input occurred in the maize sole crops as well. The soybean sole crops had the next highest combined total crop residue, followed by the 1:2 intercrop and the 2:3 intercrop. Over the long-term, maize residues contribute greater input to influence SOC sequestration; on a short-term basis, soybean crop residues replenish the soil with SOC and SON that is immediately available for plant and microbial uptake.

Another part of the study investigated decomposition of residues in the four treatments using the litterbag technique. Maize residue has a high C/N ratio, and therefore is more recalcitrant than soybean residues, which are more available to microbes. However, after 149 days, C and N from residue remaining in litterbags was greatest in the 2:3 intercrop, followed by the 1:2 intercrop, then the soybean sole crop and the maize sole crop. This finding was unexpected, because it was hypothesized that C and N would remain in litterbags in the maize sole crop for a longer period of time. Soybean sole crop residue decomposed most quickly, maize sole crop residue decomposed the slowest, and combined residue from the intercrops decomposed at intermediate rates. As expected, the C/N ratio of litterbag residue from the soybean sole crop was the lowest, although the highest C/N ratio was found in the 2:3 intercrop and not the maize sole crop.

In general, the results indicate a trend that the 1:2 intercrop could accumulate greater SOM in the long-term compared to sole crop systems. Reduced rates of residue decomposition and increased crop residue input results in accumulation of SOM, increasing soil productivity and improving soil ecological functions. Increases in SOM contribute to overall sequestration of C and N and reduces GHG emissions. The 1:2 intercrop has the advantage of combining recalcitrant crop residue from maize that can persist in soil for many years with labile soybean crop residue that maintains SOM levels. This agroecosystem had an intermediate rate of decay, implying that less C and N was lost to the atmosphere as CO_2, CH_4, and N_2O.

The final part of this study measured GHG emissions in the four treatments from pre-seeding to crop harvest. No significant differences in any of the GHG measured were detected between treatments on bi-weekly sample days throughout the growing season. Emissions responded to weather events, including precipitation and warmer days during mid-growing season. Emissions tended to correspond to changes in soil temperature and moisture, which help regulate rates of microbial respiration, nitrification and denitrification. During harvest, greater input of crop residues caused increased levels of CO_2 and N_2O respired from the soil. Methane levels remained low or negative throughout the growing season in all treatments, independent of weather events or crop residue input.

Cumulative annual emissions were calculated for all treatments. No significant difference between treatments was detected for CO_2 or N_2O. The CO_2 emitted to the atmosphere was inversely proportional to the N_2O emitted to the atmosphere. The 1:2 intercrop emitted the least CO_2 and CH_4, but emitted the most N_2O. Interestingly, the 2:3 intercrop emitted the most CO_2 and CH_4, but emitted the least N_2O. The maize and soybean sole crop plots generally had intermediate emissions values, except for the maize sole crop which provided the smallest sink for CH_4. The 1:2 intercrop and the soybean sole crop provided a significantly greater cumulative annual CH_4-sink than the 2:3 intercrop and the maize sole crop. Although some researchers have expressed doubts on the ability of soils to consume enough CH_4 relative to animal production in order to significantly mitigate global climate change, CH_4-sinks in agricultural soils constitute a viable mitigation option to be used in addition to GHG sinks.

In Argentina, agricultural products are one of the main sources of exports. Significant changes in land use and management in the Argentine Pampas have occurred, with conversion from native grassland to cropland being the most significant. This has resulted in increased GHG emissions from soils. Cropland area and the use of fertilizers have also increased. Although there is a trend towards adoption of no-till agriculture, particularly in the Mercosur area (Brazil, Argentina, Paraguay, and Uruguay), there have been very few studies on complex agricultural systems, such as intercropping. There is not enough information that relates SOC dynamics to soil biophysical characteristics, including the mechanical properties of aggregates over the long-term. No previous research had investigated the potential of intercrop agroecosystems in SOC and N sequestration and GHG mitigation.

5.2 Recommendations for Future Research

Global climate change will affect agroecosystems at spatial and temporal scales. Increases in atmospheric GHG are likely to affect agroecosystems in many ways, changing agricultural productivity and shifting nutrient cycles, as well as increasing occurrence of invasive species, including weeds, insect pest invasions, and plant diseases. It is difficult to draw generalized conclusions based on results of experiments under such a wide range of conditions presented in the literature (Fuhrer, 2003).

Global C pools have been fairly well characterized, but global N sinks and sources still need to be identified. Also, N dynamics in soils, including N_2O emissions, have traditionally been based on N-inputs to the system, such as fertilizer and manure application (Smith et al., 2007). More in-field N_2O measurements, including isotopic analysis and full quantification of the different stages of the biogeochemical cycle of N would help explain N-losses from agroecosystems. Leaching is a major pathway of N-loss from a system, and therefore it is important to incorporate into estimates. In general, reliable assessments of GHG are difficult because of spatial and temporal variability and the simultaneous occurrence and removal of emissions (Smith et al., 2007). Research on management practices that increases SOM in croplands has focused mainly on tillage, rotation, fallow, erosion control, fertilizers, pesticides and irrigation. Legumes have been recognized to reduce the need for fertilizers, but what research that has been published on intercrop agroecosystems has focused mainly on yield.

Overall, the outlook for GHG mitigation in agriculture suggests significant potential. Short and long-term experiments are being carried out in many areas of the world; often are in relation with international agricultural research centers, promoting international scientific collaboration (Easterling and Apps, 2001; Smith et al., 2007). However, although GHG emission from soil in different systems has been researched for several decades, there are still geographic regions and agricultural systems that have not been well characterized. There is a need to estimate global warming potential across a wide range of agricultural systems. Ideally, a standard method of calculating global warming potential should be established. A standardized methodology is needed to improve the accuracy of determining changes in SOC, soil N and GHG emissions and reduce disparities between measurements by different researchers at different geographical locations (Verchot et al., 2008).

A lack of scientific consensus and disparity in the results, including those reported here, indicate that more research, including long-term studies, is necessary to determine whether GHG mitigation can occur in the soil in complex agroecosystems. At present, climate forcing GHG emissions are underestimated and crudely defined for agroecosystems (Smith et al., 2007). The knowledge gap regarding GHG emissions from intercrop agroecosystems indicates the extent of how little research at all has been conducted. Until now, most research has focused on tillage and crop rotations rather than the potential as a C and N sink. Further characterization of emissions is needed to improve databases on global emissions from different crops and different land-use systems.

In Latin America climate change mitigation has still not been considered as an issue for mainstream policy implementation. Carbon sequestration in agricultural soils would be the climate change mitigation option with the highest potential in the region (Smith et al., 2007). Most countries in the region have devoted efforts to capacity building for complying with obligations under the UN Framework Convention on Climate Change, including the Argentine government in establishing voluntary national emissions targets (Ramakrishna et al., 2003). Despite recent economic changes, Argentina's voluntary efforts in addressing climate change serve as a useful example of the role developing nations can play in the international effort.

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