EFFECTS OF INDUSTRIAL AGRICULTURE ON GLOBAL WARMING AND THE POTENTIAL OF SMALL-SCALE AGROECOLOGICAL TECHNIQUES TO REVERSE THOSE EFFECTS

A report to Via Campesina
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TABLE OF CONTENTS

EXECUTIVE SUMMARY

According to the Intergovernmental Panel of Climate Change agriculture is responsible for a significant portion of the increase of greenhouse gases. But not all agriculture has the same impact on global warming. In this report we review the literature on the contributions of agriculture to climate change and conclude that the industrial agricultural system is the main contributor to greenhouse gases, while sustainable smallholder agriculture can reduce greenhouse emissions. This conclusion supports La Via Campesina's call for food sovereignty and their arguments that smallholder sustainable agriculture can cool the planet.

Industrial agriculture already contributes significantly to global warming through greenhouse gas (GHG) emissions representing about 22% of total GHG emissions - more GHG emissions than the global transport sector. The alternative, agroecological methods for agricultural production used on small-scale farms, is far less energy consumptive and far less responsible for the release of GHG than industrial agricultural production methods. Furthermore, the alternative methods have the potential to sequester GHG. Reductions in GHG emissions through small-scale agroecological production are achieved in four broad areas when compared to the industrial agricultural system, and these are summarized below:

1) Transportation of agricultural inputs, outputs, and products contributes substantially to the overall greenhouse gas input from the transportation sector. According to the IPCC (2007), 13.1% of total GHG emissions derives from transport, some fraction of which is due to long distance transport associated with the industrial agricultural system. From literature figures, we estimate that the transportation sector of industrial agriculture emits about 4% of total GHG worldwide, a factor that would be substantially reduced with the conversion to a more small-scale localized food system.

2) Industrial agriculture utilizes techniques that result in significant changes in normal ecosystem properties that in times past have maintained a tenuous balance among the materials and fluxes involved in release of greenhouse gases. Industrial agricultural production emits three very important human-induced GHGs at significant levels: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). CO_2 is the most abundant GHG and is responsible for most human-induced climate change, but N_20 and CH_4 are also potent causes of global warming. Agricultural activities are responsible for approximately 50% of global atmospheric inputs of methane (CH_4) and agricultural soils are responsible for 75% of global nitrous oxide emissions, much of which is associated with the industrial system. As part of the industrial system, Confined Animal Feeding Operation (CAFOs) contribute approximately 18% of total GHG worldwide.

3) Large scale monocultures, so characteristic of the industrial system, continue to transform the world from landscape mosaics of small-scale high biodiversity production into massive industrial-like production, purposefully reducing biodiversity in search of the “optimal” production (or profits) on any given piece of land. The ecosystem services of, for example, tight nutrient cycles and natural control of pests, are consequently

disrupted, requiring industrial inputs that inevitably lead to increases in greenhouse gas emissions. In contrast, small-scale agroecological methods have great potential to sequester carbon in above-ground and soil biomass. Deforestation, mainly associated with the spread of large scale monocultures, is one of the major emitters of CO_2, and programs of community tree planting and agroforestry have great potential to reverse this trend.

4) Agriculture was developed to be an energy producing system (and remains so in more traditional forms of agriculture), but with the introduction of industrial methods it has been turned into an energy consuming system. The new industrial farmer replaces the thought-intensive technology in use for so many years with brute force energy application, made possible because we have an abundant store of fossil fuel energy. Consequently, energy in agriculture was converted from something that originally was the main *product* of agriculture to something that became a main *input* into agriculture -- a change from “using sun and water to grow peanuts” to “using petroleum to manufacture peanut butter.” It has been estimated that this industrial food system expends 10-15 energy calories to produce 1 calorie of food, an effective reversal of what had been the reason to develop agriculture in the first place.

Although precise figures of what fraction of global warming is due to industrial agriculture are difficult to calculate, it is nevertheless clear from the structure of the industrial system as compared to small-scale more traditional forms, as well as estimates of GHG emissions from particular sectors, that the fraction is considerable. Transforming the industrial agricultural system into localized small-scale diverse agroecological farms would reduce GHG emissions and could even reverse the trend by sequestring carbon in trees and soils. Therefore, the food sovereignty proposal of La Via Campesina would not only provide livelihoods for millions of smallholders around the world, but could also aid in cooling the planet for all.

INTRODUCTION: OVERVIEW OF GREENHOUSE GASES (GHGs) IN AGRICULTURE:

It is by now familiar knowledge that global modes of production, consumption and trade have generated enormous problems for the earth, including the transcendent problem of global warming. Increases in greenhouse gases associated with industrialization have been identified as the main cause of global warming (IPCC 2007). According to the Intergovernmental Panel of Climate Change (IPCC 2007) agriculture is responsible for a significant portion of the increase of greenhouse gases. But not all agriculture has the same impact on global warming. In this report we review the literature on the contributions of agriculture to climate change and conclude that the industrial agricultural system is the main contributor to greenhouse gases, while sustainable smallholder agriculture can reduce greenhouse emissions and therefore contribute to cooling the planet. In the first part of this report we provide an overview of the three main greenhouse gases and their links to agriculture. Then we examine how food transportation, agricultural production and land conversion affect greenhouse gases. We then analyse how the agricultural system has been transformed from an energy producer to an energy consumer. Finally, we conclude with a comparison between large-scale industrial agriculture and small-scale sustainable agriculture in terms of their potential to mitigate the impacts of global warming.

The most recent report of the IPCC concluded that atmospheric concentrations of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide ($N0_3$) far exceed the natural ranges over the last 650,000 years, and that GHG emissions have grown by 70% since 1974. Their report concluded that: 1) land use change (for agriculture and urbanization) was the second highest cause of global increases in CO_2 after fossil fuel use; 2) methane increases were very likely due to agriculture and fossil fuel use; and 3) nitrous oxide increases were due primarily to agriculture (IPCC 2007). A logical conclusion is that the industrial agricultural system is responsible to a great extent for the warming of the planet because: 1) industrial agriculture is fossil fuel intensive; 2) a large proportion of methane emissions come from confined animal feeding operations (CAFOS); 3) most of the nitrous oxide emissions come from nitrogenous fertilizer applications; and 4) large tracks of land in the tropics are being converted to large scale intensive monocultural plantations.

CO_2 is emitted from agricultural systems through a variety of mechanisms, including: 1) plant respiration; 2) the oxidation of organic carbon in soils and crop residues; 3) the use of fossil fuels in agricultural machinery such as tractors, harvesters, and irrigation equipments; and 4) the use of fossil fuels in the production of agricultural inputs such as fertilizers and pesticides. Carbon uptake can occur through photosynthesis as well as the accumulation of organic carbon in the soil. Plant carbon can enter the soil organic carbon (SOC) pool as plant litter and residues, root material and exudates, as well as animal excreta (when the animals eat the plants). These processes are highly dependent on agricultural management though, and many systems do not sequester carbon in soils for this reason (Marland et al. 2003). In terrestrial systems, SOC is the largest pool of carbon and globally contains over 1550 Pg C, where a Pg is equal to 1015 g or 1000 million metric tons (MMT). The soil inorganic carbon (SIC) pool contains 750-950 Pg C, and terrestrial vegetation is reported to contain an additional 600 Pg C (Batjes 1996; Houghton 1995).

Carbon moves from the atmosphere through plants, soils, and animals and back. It is returned from agricultural activities to the atmosphere through four primary routes: 1) changes in land use that release carbon from degraded soils and cleared forests; 2) processing of petroleum to make fertilizer at a rate of at least 40 million tonnes per year (Steinfeld et al. 2006a); 3) methane production from manure and fertilizers on crops; and 4) on-farm fuel use in production and transportation of plants (60 million tonnes) and animals (30 million tonnes) (Steinfeld et al. 2006b).

Agricultural methane (CH_4) is released by methane producing bacteria existing in the digestive tracts of ruminant animals (e.g. cattle) and manure piles of farm animals, as well as by soil microbial processes in farm production (e.g. rice grown under flooded conditions) (Smith et al. 2008).

N_2O is produced during the decay of animal manure as well as through the conversion of NO_3 by bacteria in the soil, including the breakdown of nitrogen-based fertilizers.

Industrial agriculture already contributes significantly to global warming through greenhouse gas (GHG) emissions. Agriculture represents about 22% of total GHG emissions, which is more GHG emissions than the transport sector (McMichael et al. 2007), but industrial agriculture may contribute even more to GHG emissions in the future. For example, the EU, a Kyoto Protocol signatory, is responsible for about 18% of global GHG emissions, and has set GHG emission reduction targets that depend on use of agrofuels instead of petrofuels. However, this means that the EU is ‘reducing its own emissions by raising emissions in developing countries that produce the feedstock oils (through increased deforestation and land use change, for example) and are not bound by emissions reduction targets, especially Indonesia and countries in Latin America’ (Smolker et al. 2008: 38).

The alternative agroecological methods for production used on small-scale farms are far less energy consumptive than the industrial agricultural production methods (Smith et al. 2008). According to Jules Pretty, industrial agriculture uses 6-10 times more energy than agroecological methods. Agroecological methods use less energy by depending on fewer outside inputs and less petrofuel-dependent infrastructure, but they also restore soils and nitrogen-fixating bacteria populations, reducing emissions up to 15%. Restoring grasslands and wetlands can also reduce emissions up to another 20% (Apfelbaum 2007). Producing food for local consumption reduces the distance food is transported (ie. "food miles"). This is becoming increasingly important since air freighted food has increased 140% since 1990, and the shipping industry emits twice as much GHG as the aviation industry (Intertanko 2007).

Calculating and/or equating GHG emissions from land-use change and various agricultural activities is difficult at present. Nevertheless there are some fundamental principles that, when combined with the minimal data that are available, leaves little doubt that industrial style agriculture is a major contributor to greenhouse gas emissions. Industrial agriculture has as one of its key features the concentration of production in areas that are “optimally” suited for specific agricultural commodities, which are inevitably restricted in geographic location and cause several problems. First, transportation of agricultural inputs, outputs, and products contributes substantially to the overall greenhouse gas input from transportation, cited by the IPCC as one of the most important sources of greenhouse gases. Second, the industrial system utilizes techniques that result in significant changes in normal ecosystem properties that in times past have maintained a tenuous balance among the materials and forces involved in release of greenhouse gases. Third, large scale monocultures, so characteristic of the industrial system, continue to transform the world from landscape mosaics of small-scale production into massive industrial-like production systems, purposefully reducing biodiversity in search of the “optimal” production (or profits) on any given piece of land. The ecosystem services of, for example, tight nutrient cycles and natural control of pests, are consequently disrupted, leading inevitably to increases in greenhouse gas emissions. Finally, there is a certain irony in the fact that a system that was designed to be an energy producing system (and remains so in more traditional forms of agriculture), has been turned into an energy consuming system, which is what has happened with the introduction of the industrial system into agriculture. It goes without saying that the massive increase in energy demands are satisfied mainly through the use of fossil fuels, with the well-known concomitant consequences. In the rest of this report we summarize recent literature on each of these four issues.

TRANSPORTATION

The IPCC (2007) contends that 31% of all GHG emissions derive from "land use", meaning essentially agriculture and forest clearing. Since most forest clearing today is done for the purpose of agriculture, it is reasonable to interpret the 31% figure as directly or indirectly a product of agriculture. What this analysis fails to take into account, however, is the massive amount of transport involved in moving agricultural inputs and outputs around the world. Again according to the IPCC, 13.1% of total GHG emissions derives from transport. Deriving a global estimate of just what fraction of this transport total is due to the industrial agricultural system is problematic. But if we presume, for example, that industrial agriculture and industry itself emit roughly the same proportional amount of GHGs as they emit in their normal operations (agriculture = 13.5%, industry 19.4%), and we allow the same proportion emitted for private travel as for residential and commercial buildings (7.9%), we can then estimate that the transportation sector of industrial agriculture emits 4.3% of total GHGs worldwide (i. e., 33% of total travel). This is obviously a very rough approximation, but it clearly indicates that transportation of agricultural inputs and outputs is a significant factor in agriculture's contribution to GHG emissions. Our estimate that industrial agriculture is responsible for 33% of GHG's associated with travel is supported by data from the UK, for instance, where it has been estimated that 28% of all road transport is devoted to agricultural activities (Pretty et al. 2005). In addition, while it is difficult to generalize, one life-cycle study of the US agricultural transport system noted that transport associated with agriculture as a whole contributes 11% of all agricultural GHG emissions from agriculture (Weber and Matthews 2008).

While studies explicitly about transport in agriculture have been numerous in Northern agricultural systems, we know of no studies that have compared transport emissions from smallholder farming to industrial farming. The considerable amount of variation in production systems and transport makes it difficult to compare agricultural emissions from transport on a worldwide scale. However, many studies have focused on smaller scales using the concept of 'food miles' to refer to the total distance food has to travel from the original production site to the place where it is consumed. The greenhouse gas emissions from air transport are considered particularly high, with estimates at 1.093 CO_2 equivalent to move one tonne of food one kilometer (Edward-Jones et al. 2008). In comparison, truck transport was estimated to contribute 0.15 CO_2 equivalent/tonne/km, while rail transport was estimated to contribute 0.01 CO_2 equivalent/tonne/km (Meisterling et al. 2007). In one study of the environmental cost of major food items consumed in the United Kingdom (Pretty et al. 2005), it was concluded that domestic transport accounted for the highest level of environmental cost from farm to point-of-sale due to high volumes in comparison to air or sea transport. In another major review in the UK it was suggested that, in addition to air transport, urban food transport (ie. people going to buy food or having food delivered), heavy goods vehicle delivery, and shipping all need to be considered to fully assess the GHG emissions from transport (Smith et al. 2008). They noted that air transport of food, which has the highest GHG emissions, has more than doubled in a decade (1992-2002). The use of food miles as a substitute for complete calculation of energy cost or GHG emissions has been criticized as being "simplistic." For example, Saunders et al. (2006) conclude that transport of certain food items from New Zealand (NZ) to the UK can make sense energetically if a more wholistic analysis of energy use is taken into consideration. So, for example, "The UK uses twice as much energy per tonne of milk solids produced than NZ, even including the energy associated with transport from NZ to the UK ... [reflecting] the less intensive production system in NZ." Saunders et al. suggest this is an interesting and important conclusion that emphasizes the efficiency of "less intensive" production systems. Of course it should be obvious that having the UK convert its dairy system to something more energy rational, as occurs in New Zealand, would be even more energy efficient.

Long distance transportation can be a major source of overall life cycle emissions for cereals, fruits and vegetables (Sim et al. 2007). One report looked at the ‘cradle to plate’ (ie. not including waste disposal) for food that is produced in New Zealand and exported to the UK, including apples, sheepmeat and butter. In their case study of butter (milk solids) they found that carbon emissions for direct inputs on-farm, including electricity, agricultural machinery and lubricants accounted for between 190–200 g CO_2/MJ of carbon emissons (Saunders et al. 2006). Fuel used for tractors, trucks, utilities and cars required 36.4 and 22.4 litres/ha, with a total energy use of 2,483 MJ/ha or 3,032 MJ/tonne of milk solids. Carbon dioxide emissions from all liquid fuels was 230 kg CO_2/ha or 280.4 kg CO_2/tonne of milk solids. Energy used for transport to the UK was estimated at 0.114 MJ per tonne km by shipping, the equivalent of 0.007 kg CO_2 per tonne km, while a fully loaded articulated truck within Europe (maximum of 44 tonnes allowed) was estimated to use 0.419 MJ per tonne km of energy and 0.027/ kg CO_2 emissions (Saunders et al. 2006). Shipping food from New Zealand to the UK (17,840 km) was estimated to emit 125 kg $C0_2$ per tonne. Other studies have estimated 0.2 MJ per tonne km for shipping internationally (Wells 2001; Webb 2004).

On-farm transportation and energy use are large contributors to CO_2-C emissions from industrial agricultural systems, as noted in the IPCC report (2007). Diesel used in industrial agriculture systems of the US amount to 59,000 million liters and are estimated to release 10.8 MMT C/yr. Estimates of CO_2-C emissions from agricultural machinery used in conservation, reduced, and no-tillage systems were 72, 45, and 23 kg C/ha/yr, respectively. This shows that reduced tillage also leads to decreasing on-farm machinery operations and C emissions from fuel. Data of fuel records from farmers estimate that crops under no-till used 45 l of diesel/ha/y, whereas other more field intensive crops may use up to 84 liters of diesel/ha/yr (Follett 2001).

A comparison of locally grown apples in Germany to imported New Zealand apples showed 27% difference in carbon emissions, taking into account the emissions of longer cold storage in Germany. Transport alone, which was by ship and truck for the imported apples, and by truck for the local apples, was 2.8 MJ/kg for imported apples and 0.81 MJ/kg for local apples (Blank and Burdick 2005). Transport difference was approximately 23,000 km.

Another life cycle assessment comparing organic and conventional wheat concluded that the global warming potential of one loaf of bread using conventional wheat flour (1 kg bread loaf) without transport is 190 g $C0_2$ equivalent, while one organic loaf of bread results in 160 g $C0_2$ equivalent. However, if the organic bread is transported 420 km away, that difference disappears (Meisterling et al. 2009).

GHG emissions from transport have frequently been omitted from the globalization ledger. Discounting the negative 'externalities' of transport has helped create a false economy for global agroindustrialization. It is a strategy that also enables a false economy for agrofuels projects, as noted by those who call for a full 'lifecycle analysis' of the impact of agrofuels (TNI 2007. "Much of the 'evidence' presented for agrofuels to reduce greenhouse gas emissions ignores the larger picture of 'land use change' (usually deforestation), soil erosion and nitrous oxide emissions" (TNI 2007:10).

We conclude from a review of relevant literature that transportation accounts for a small but significant amount of GHG emissions that can be attributed primarily to industrial agriculture. Evidence suggests that a conversion to locally-based distribution of products and sourcing of inputs could have a significant impact on reducing global carbon emissions. The calculations of GHG emissions is a complicated one that has to take into account not only transportation in the industrial system, but transportation in the locally based system that might replace it. Just how much savings would accrue to a transformation from the industrial to an agroecological system is not clear, although it is highly likely that GHG reduction would occur through the consequent change in transportation activities. One thing clear from the outset is that energy intensive, subsidized, industrial agricultural production in one region (e.g. maize in the United States) shipped thousands of miles to an area that formerly produced in a more energy efficient manner (e.g., Mexico), is not the way to fight global warming.

INDUSTRIAL STYLE AGRICULTURAL PRODUCTION

Industrial Agriculture and greenhouse gases

Industrial agricultural production emits three important greenhouse gases at significant levels: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (NO_2). CO_2 is the most abundant greenhouse gas and is responsible for most human-induced climate change, but N_20 and CH_4 also cause warming. Agricultural activities are responsible for approximately 50% of global atmospheric inputs of CH_4, and agricultural soils are responsible for 75% of global nitrous oxide emissions (Scheehle and Kruger 2006), indicating that agriculture is a significant emitter of the two latter greenhouse gases. The nitrogen and carbon cycles are closely coupled, so it is not surprising that alterations to the N cycle by industrial farming practices also affect the total C balance (Russell et al. 2009).

Agricultural alterations to the carbon cycle: CO_2 is emitted from agricultural systems through a variety of mechanisms, including: 1) plant respiration; 2) soil efflux, which results from the oxidation of organic carbon in soils and crop residues; 3) the use of fossil fuels in agricultural machinery such as tractors, harvesters, and irrigation equipment; and 4) the use of fossil fuels in the production of agricultural inputs such as fertilizers and pesticides.

Carbon uptake can occur through photosynthesis as well as the accumulation of organic carbon in the soil. Plant carbon can enter the soil organic carbon (SOC) pool as plant litter and residues, root material and exudates, as well as through animal excreta. These processes are highly dependent on agricultural management methods, and many systems do not sequester carbon in soils for this reason (Marland et al. 2003). CO_2 efflux, or respiration from soils, is a combination of microbial and root processes that transfer the C in soil organic matter (SOM) back to gaseous CO_2. Soil respiration rates are governed by factors similar to other soil functions: temperature, water content, microbes, and the composition of plant material decomposing in the soils.

Worldwide, soils contain about 70% of terrestrial organic C (Schlesinger 1997). It is estimated that intensive agriculture has contributed to the loss of about 25% of the original SOM content (IPCC 2007). This magnitude of SOM loss is not trivial because SOM contributes to soil water holding capacity, aeration, cation exchange capacity, and soil aggregation. It is often considered one of the most valuable indicators of overall soil quality. Climate has a major effect on the severity of SOM loss with industrial agriculture. In a meta-analysis of published studies, it was found that tropical forests are more sensitive to the effects of agricultural management than temperate systems (Ogle et al. 2005). Excessive N fertilization results in a net loss of SOM and soil C, probably as a result of altering the microbial community structure (Khan et al. 2007).

Soil C content is closely linked to microbial community structure and function in soils, and in particular, fungal biomass (Bailey et al. 2002). Since intensively managed soils utilize methods that decrease fungal abundance and favor bacterial domination, this microbial community shift may be one mechanism by which industrial agriculture decreases soil C sequestration capacity and ultimately becomes a net source of C in the global atmosphere. Soil organic matter is held in soils through soil aggregates, which are essentially clumps of soil mixed with SOM. Soil aggregation is enhanced by fungal abundance, and in particular, symbiotic mycorrhizal fungi. Arbuscular mycorrhizal fungi, found in the roots of most arable crops and about 80% of plants worldwide, are thought to contribute to C sequestration in soils in their residual tissues that make up significant portions of soil organic matter (Treseder and Allen 2000). Arbuscular mycorrhizal fungi derive most of their C from their associated plant. Thus, plants with these fungi in their roots must take more CO_2 from the atmosphere to support the C demands of the fungi. However, in large-scale agricultural systems, the quantities of applied inorganic N, pesticides, and frequent tillage decrease the diversity, abundance, and functioning of these beneficial microbes (Johansson et al. 2004, Oehl et al. 2004, Treseder 2004). Since these fungi have also been shown to act synergistically with N-fixing bacteria and other soil microbes that facilitate SOM accumulation, intensive agriculture results in a further depletion of soil nutrients, and a greater need for fertilizer application (Jeffries et al. 2003). Together, these cascading effects lead to greater greenhouse gas emissions and a net C loss from intensively managed agricultural systems.

Alterations to the N cycle — Nitrogen Fixation: Approximately 78% of the atmosphere is composed of nitrogen in the form of molecular nitrogen (N_2). This nitrogen is not biologically reactive due to its strong triple bonds that keep it in the N_2 form. A given molecule of nitrogen in the atmosphere might be floating around without changing for hundreds of years. However, through the action of some very specialized bacteria, called nitrogen-fixing bacteria, molecular nitrogen is converted to ammonia (NH_3) and brought into the biological system. Many leguminous plants (and a few non-legumes), such as soybeans and alfalfa, contain nodules in their roots to house these N-fixing bacteria. Since terrestrial plants are often limited by N, plants that grow with N-fixing bacteria in their roots have an advantage in nutrient-poor soils, since they come equipped with their own supply of N fertilizer. The abundance of N-fixing plants is highly variable, but their prevalence is somewhat correlated with nutrient poor soils, especially in areas with frequent disturbances. One major way that industrial agriculture has altered the N cycle is through the widespread cultivation of N-fixing plants, predominantly soy, alfalfa, clover, and peanuts.

A second way that industrial agriculture has manipulated the quantities of biologically available N is through the Haber-Bosch process to create synthetic fertilizers. In this process, humans have short-circuited the N cycle to force the conversion of N_2 to biologically usable compounds such as ammonia, ammonium nitrate, and urea. To break the strong triple bonds of N_2, high-energy inputs and temperatures of around 500 C are required. Consequently, synthetic fertilizer production consumes 3-5% of the world's natural gas and 1-2% of the world's annual energy supply (IPCC 2007).

Synthetic fertilizers and Nitrous Oxide (N2O) production*:* Nitrous Oxide (N_2O) is an atmospheric trace gas that has increased from 270 parts per billion by volume to 314 ppb in the past 250 years. N_2O emissions are a salient concern for global warming, as its global warming potential in the next 100 years is 298 times stronger than CO_2 per unit weight in trapping atmospheric heat (IPCC 2007). In other words, emitting 1 metric ton of N_2O is equivalent to emitting 298 metric tons of CO_2. Its lifetime in the atmosphere is also 114 years, making it not only potent, but extremely persistent. N_2O is also involved in the depletion of stratospheric ozone (Crutzen 1970), which is the ozone in the upper atmosphere responsible for filtering out 99% of the sun's damaging ultraviolet light and is also a greenhouse gas. Since the green revolution, N_2O has increased in the atmosphere by about 18%, largely due to conversion of synthetic fertilizers from large-scale, industrial agriculture.

Nitrification is an aerobic process, meaning it requires oxygen, by which ammonia (NH_4^+) becomes oxidized to nitrate (NO_3^-), with nitrite (NO_2) as an intermediate product. This process is completed by two sets of microorganisms: ammonia oxidizing bacteria and Archaea (Treusch et al. 2005). In unaltered soils, nitrification is a requisite step in decomposition, which is the breakdown of organic material into simpler forms that other plants and animals can use for nutrition.

Another source of soil-derived N_2O is through the process of denitrification. Denitrification completes the nitrogen cycle and returns NO_3^- back to N_2. However, N_2O is produced as an intermediary compound, much of which can be released to the atmosphere. Denitrification is accomplished by a wide range of bacterial species, and the specific composition of denitrifying bacteria can determine the rate at which N_2O is released back into the atmosphere (Cavigelli and Robertson 2001). The magnitude of N_2O fluxes from soils depends on complex interactions between the microbial community, the quality of plant litter entering soil organic matter (Millar and Baggs 2004, Guo et al. 2009), climate, and soil properties such as temperature, water content, and pH. Favorable conditions for soil N_2O production include, high water content (Liu et al. 2007) and excess quantities of soil N. Low soil pH slows the denitrification process, but may increase the relative amount of the N_2O fraction, particularly when NO_3^- is in excess. When urea and ammonium-based fertilizers are applied to soils, NO_3^- production is reduced, implying either an inhibition of nitrifiers or increase in denitrification rates (Nishio and Fujimoto 1990). N_2O release has been directly correlated to the quantity of NH_4^+ and NO_3^- applied to soils (Venterea and Rolston 2000, Zebarth et al. 2008a, b). It is thus not suprising that although natural soil processes produce about 6.6 Tg of N_2O annually, the quantity produced in industrial agriculture is estimated to be 208 Tg annually (IPCC 2007).

Fertilizer application frequency impacts N_2O release because large quantities of N_2O are produced for several days following fertilization. Since the over-fertilization of crops in large-scale agricultural systems is exceeding plant demand, the excess N promotes the growth of bacteria that release significant quantities of N_2O. Alterations in microbial communities from habitual fertilizer application can have surprisingly long-lasting effects. Intensive agricultural management of a site that had been abandoned 100 years ago continued to harbor greater abundances of nitrifier bacteria (Compton and Boone 2000). There are numerous studies documenting shifts in microbial community composition and function with agricultural intensification (Buckley and Schmidt 2001, Girvan et al. 2004, Wolsing and Prieme 2004, Ferreira et al. 2009). Nitrification and denitrification, in particular, have been shown to increase with increasing intensity of agricultural management regimes, with concomitant changes in the bacterial and fungal communities involved in N cycling (Wakelin et al. 2009). While we still do not know all of the species of bacteria and fungi that are impacted, it is clear that large-scale intensive management shifts bacterial communities in such a way that favors more rapid production of N_2O, which ultimately accelerates global warming.

Management Impacts on Greenhouse Gas Emissions: There are interactions between fertilizer and tillage practices that can lower or increase the amount of GHG emissions from agricultural fields. Studies of GHG fluxes in the US Corn Belt showed that continuous corn-cropped rotations contributed significantly to N_2O emissions, driven by pulse emissions after N fertilization in concurrence with major rainfall events. These emissions range upward from 3-8 kg/ha/yr. More complex systems, such as corn-soybean rotations and the restoration of prairies, showed diminished N_2O emissions and contributed to global warming mitigation (Hernandez-Ramirez et al. 2009). In another tillage and fertilizer study, total growing season non-CO_2 emissions were equivalent to 0.15-1.9 Mg CO_2/ha/yr (Ventera et al. 2005).

No-till cropping has also been shown to enhance carbon storage, aggregation and associated environmental processes with no significant ecological or yield trade-offs. No-till agriculture in corn-soybean-wheat rotations in Michigan showed an accumulation of 26 g C/m^2/yr over 12 years in 0-5 cm soil depth (Grandy et al. 2006). Looking at a range of cropped and unmanaged lands in a long-term analysis, Robertson et al. (2000) found that conventional systems were net emitters of greenhouse gases, and that all but the conventionally managed system accumulated soil carbon over the decade since establishment. No-till was shown to accumulate 30 g soil C/m^2/yr while cover crop organic was shown to accumulate 8-11 g soil C/m^2/yr. N_20 fluxes were also three times higher in the industrial agricultural sites versus the unmanaged sites (Robertson et al. 2000). This type of study shows that tillage makes a large difference in the offsets of greenhouse gases, and that creating agricultural systems that more closely resemble natural systems may very well help the agricultural system attain net negative or neutral global warming effects (Robertson et al. 2000).

Animal production and greenhouse gases

***Confined Animal Feeding Operations (CAFOs*)**: Production of livestock employs over a billion people, predominantly the world's poor, to produce about one-third of human dietary protein and 40% of all agricultural income. Furthermore, global trade in livestock products is rapidly growing, as changes in food preferences increase the demand for meat and milk (FAO 2008). These trends create pressure to raise cows, pigs, and chickens in large-scale confined spaces and feed them grains, soybeans, or residues raised on released or newly deforested land. Livestock production uses 70% of agricultural land and 25% of earth's land area so increased production leads either to new land or concentrated animals. For example, there are 450,000 animal feeding operations in the U.S. where animals are concentrated in limited spaces, feed is brought to them, and manure is deposited (U.S. EPA 2009a).

The total impact of livestock on greenhouse gas emissions includes production-related activities such as deforestation, overgrazing, feed-crop production with fertilizers (therefore, manufacture of fertilizers), as well as CAFOs. There are over 14,000 U.S. operations classified as *large* CAFOs by the U.S. EPA (2009b); their effects on greenhouse gases, other air pollution, water pollution, livestock health, anti-biotic use, human health, and animal welfare are reaching crisis proportions (Gurian-Sherman 2008).

Soils are the main repository for carbon on the land, with over 1100 billion tonnes—more than twice the amount of carbon held in vegetation (Sundquist 1993; Sundermeier et al. 2005). Loss of soil organic matter, including CO_2 to the atmosphere, plus emission of methane and nitrous oxide from animal respiration, manure, and fertilizers, cause 18% of human induced warming effect (FAO, 2006). When forested areas are cleared for grazing they sequester less carbon above ground, but also in the soil, depending on grazing practices. Soils under feed-crop production fare worse as they no longer function as net carbon sinks. The enormous quantities of organic C bound in plant biomass and soil organic matter are liberated and respired back to the atmosphere as CO_2 (Dixon et al. 1994 a, b, Wassenaar et al. 2006). Clearing and burning forests for grazing or cropland may release over a billion tonnes of CO_2 carbon per year (IPCC 2001). Fifty percent of soil carbon may be lost in the first decade following forest clearing (Nye and Greenland 1964); additional C is released by oxidation of biomass lagged from previous years (Houghton 1991). CO_2 emissions increase with cultivation of feed crops and liming acidic soils.

Carbon dioxide emissions from fossil fuel used for production and transport of fertilizers for feed-crops that feed animals in CAFOs and confinement dairies probably exceed analogous transportation costs for pasture fed animals. Shipment of soybean cakes from Brazil to Swedish dairies, for example, costs 32,000 tonnes of CO_2 emissions per year for transportation by ship (Cederberg and Flysjö 2004). Transport of meat is responsible for over 800,000 tonnes of CO_2/yr (Steinfeld et al. 2006 b). In total, livestock production may be responsible for 2.4 billion tonnes of CO_2 emissions per year (Steinfeld et al. 2006 a);; Sundquist 1993). Thus, livestock effects exceed transportation effects on the earth's climate.

Greenhouse gases from manure.-- Thirty-seven percent of all methane emissions are attributable to livestock, especially those animals fed grains. This figure includes emissions from liquid storage of manure, enteric fermentation by ruminants, and burning of fossil fuel to make and transport fertilizers for feed crops. Enteric foregut fermentation of fibrous food and exhalation of gas by the world's beef and dairy animals, may emit 80 million metric tonnes of methane per year—that is, 80% of agricultural methane emissions (U.S. EPA 2005, Steinfeld et al. 2006 a, b). Emissions vary with the mass of animals and their food type. Most of the world's swine and poultry are raised in intensive (concentrated) systems, and most U.S. beef is finished in feedlots resulting in significantly increased greenhouse gas production. CAFOs produce 60% of the animal manure output in the U.S. (US EPA 2006).

CAFOs are most potent as sources of methane and nitrous oxide when manure is stored in liquid form, promoting anaerobic breakdown. Greenhouse gases are diffused rather directly from reservoirs of liquid manure. Methane emissions from decomposition of pig and dairy manure in anaerobic holding reservoirs or tanks constitute 4% of global methane emissions, or about 10 million metric tonnes (US EPA 2005, Hao et al. 2001). Livestock contribute about 37% of the methane and 65% of the nitrous oxide, as well as 30 million tones of ammonia per year because efficiency of nitrogen retention is low: approximate values range from dairy (40%), poultry (34%), pigs (20%), to beef (5%) (Van der Hoek 1998, Smil 2002).

Comparison of gases from livestock and manure in alternative systems: Concentrated feeding operations contribute significantly more methane than well-managed pasturing. Properly managed pastures are carbon sinks when manure is cycled into the soil by natural processes. For example, grass-fed beef pastured with rotation, on fields thick with grass and leguminous forbs, actually sequester significant amounts of carbon and nitrogen with much cost savings in overhead. Similar claims are made for dairy cows, field- or uncrowded, shed-raised hogs, and free-range or mobile-enclosure fed chickens (Salatin 1998).

Pasturing is not automatically favorable to greenhouse gas balances, however, because overgrazing can cause 25-80% declines in soil carbon (Asner et al. 2003). Therefore, sustainable livestock production avoids unnecessary emission of greenhouse gases by pasturing in a way that animals drop their manure on healthy pastures. This system thus utilizes natural biodiversity and the nutrients to build rich soil. Optimal livestock production with optimal carbon sequestration is served when vegetation is sufficiently dense that “no rain hits bare soil” (Pete Farrell, interview, The Land Institute, 2008). Maintenance of good soil cover is possible with intensive grazing if animals in appropriate densities are moved from pasture to pasture with proper timing. Pasture-raised, grass-fed animals are healthier and often more profitable, and their manure makes efficient and effective fertilizer.[1] According to Smil (2002) the ruminant inefficiency noted above is worse in animals fed on crop concentrates made from corn and soybeans, and less in pasture raised animals.

Dairy management can also benefit from a shift to grass-fed cows and natural manure management, but the nature of the transition is not without problems that require careful management on the land and in governments. Competition between small producers and industrial-scale production is currently biased in favor of large producers. Corporations that control the big industrial dairy operations have long profitted from heavily subsidized, confinement-style dairy farming systems. Corporations are now extending their reach to cheaper products from small operations, for example, exploiting poor Pakistani families (Gura 2008).

Specific problems with nitrogen management on pastures: Animal waste deposited on bare soil is oxidized to nitrite and nitrate, with the possibility of saturation and N_2O release from anaerobic patches of urine. The different potential for emissions depends on the substrate, which is obviously different for feedlots and grassy pastures. Manure spread on cropland may also release greenhouse gases, notably N_2O (Smith et al. 2008). Tropical soils are especially prone to N_2O loss. Tropical ecosystems tend to be limited by phosphorus. Nitrogen fertilizer in phosphorus-limited systems generates 10-100X more NO and N_2O than the same fertilizer added to N-limited ecosystems (Hall and Matson 1999).

Manure loses nitrogen at a higher rate than mineral fertilizers, which vary from 30- 70% in their efficiency (Smil 1999). Soil emissions from manure are the largest livestock source of N_2O worldwide, but their relative amounts differ greatly depending on conditions (Mosier et al. 2004). N_2O emissions from applications of slurried manure to fields were lower after storage for 6 months or anaerobic digestion (Amon et al. 2002).

The biggest single cause of nitrous oxide pollution is industrial poultry production. Over 8 billion birds per year are confined and crowded into huge buildings in the U.S., for example, with litter-based manure management systems and intensified N_2O emissions, as well as other problems such as water pollution, bird health, antibiotics, and human health (US EPA 2007). A less crowded approach to poultry production is one of the most critical needs in fashioning a sustainable food system free of the above problems.

Mitigation**:** The world's huge livestock populations and the inherently high levels of greenhouse gas emissions from them provide opportunities to influence climate change. For example, any change in the carbon flux in soils, though small, can be significant because of soil volume (Rice et al. 1999, Smil 1999). Although market forces favor intensification of livestock production (Vlek et al. 2004)[1], alternative methods may emit fewer greenhouse gases and provide richer returns for livelihood, health, and sustainability, where deforestation, feed-crop dependence, and land degradation are avoided.

Mitigation methods include fewer livestock, intensification without crowded confinement, conservation tillage, organic farming (Casey and Holden 2006), crop rotation, cover crops and green manures, reduced compaction, and water management to irrigate in ways that prevent erosion. Despite these obvious means for reducing serious problems, reduced chemical-fertilized feed grains would be the most effective way to reduce greenhouse gas pollution in animal agriculture (Fless et al. 2002). As much as 18 million tonnes of CO_2/yr are emitted in cultivation of 1.8 million km^2 of maize, soybean, and wheat for livestock feed (Sauvé et al. 2000), and the problem is rapidly growing. There were about 750,000 cattle in the world in 1950 in contrast to about 1.53 billion in 2001 (International Erosion Control Association), so the potential for altering the balance between nitrous oxide emissions vs. sequestering nitrogen in the soil is large.

It is crucially important to use policy mechanisms related to the Kyoto Protocol's "Clean Development Mechanisms" to provide stimulus and support for small farmers willing to raise livestock sustainably. There is a large potential for sequestration of carbon in cropland soils. Eighteen million tonnes of CO_2 are emitted from the 1.8 million km^2 of arable land cultivated with maize, wheat, and soybeans for livestock feed. The world's degraded soils are responsible for 50-66% of the carbon lost by human activities (Lal 2004a). Potentially, 0.3 to over 1 tonne of carbon/ha/yr could be sequestered through application of methods designed to restore degraded soils (IPCC 2000, Steinfeld et al. 2006 a, b). Organic methods are spreading rapidly, in part because of their explicit design for soil restoration. Organic and other environmentally sustainable farming systems can also reduce nitrous oxide emissions by avoiding overproduction of manure and limiting stocking densities to the amount of land available for manure application (Kotschi and Müller-Sämann 2004). Livestock diets are being developed to reduce methane and nitrous oxide emissions.

Environmental services should be considered in all policy initiatives related to the complex combination of livestock production. Sequestration of carbon and nitrogen are goals compatible with soil and water management, and protection of biodiversity. Individually tradable land use, or secure rights to land, water, and pastures, as well as fair market regulation, must be part of the policy incentives to meet these goals (Leonard 2006). Policy makers must be made aware that deforestation is not caused by the world's poor. Most deforestation is promoted by large investors and occurs in plots of hundreds of hectares. Virgilio Viana, former environmental minister of the Brazilian state of Amazonas, said in an interview that the cause of deforestation is poor governance (Living on Earth, National Public Radio, U.S., January, 2009). Many of the current subsidies must be removed and replaced by incentives for small, sustainable operations (Carvalho et al. 2004).

Environmentally realistic externalities should be incorporated into fees for inputs and waste management. Subsidies for deforestation and carbon credits for large-scale plantings of soybeans and oil palms, while small farmers are criminalized, is bad policy. Small livestock producers as a group constitute a huge and potentially powerful agricultural system capable of sequestering carbon and nitrogen by farming sustainability, if allowed to do so, or especially if provided with a system of incentives to do so.

The world's soils have lost over 42 gigatons of carbon in the past 250 years, but they still have the capacity to recover up to 66% of the amount lost (Lal 2004a). The maximum achievable carbon sequestration in dryland soils is estimated to be about 1 billion tonnes per year (Lal 2004b), but the agricultural and ecological benefits would be great (Dregne 2002, Whitmore 2000). Rotational grazing, with care to keep the livestock numbers within limits conducive to healthy growth of grasses and forbs will be the primary tool for sustainable livestock production (IPCC 2000, Haynes and Williams 1993). Assuming that improved practices would allow recovery at of at least 0.3 tonnes of carbon/ha/yr (0.1-1.3 tonnes/ha/yr [IPCC 2000]), about 270 million tonnes of carbon/yr can be restored if even 60% of global arable land were to be worked with conservation tillage (in which at least 30% of plant residue is left on the soil) over the next few decades (Lal 1997). Organic trials with maize and soybeans demonstrated yields comparable to conventional methods, with the added benefit of increased drought resistance because of improved soil chemistry (Vasilikiotis 2001).

The world's efforts to reduce emission of global greenhouse gases cannot afford to continue with industrial agricultural business as usual, but must support small farmers to work land with practices that restore soil. Supporting small farmers on secure land, while protecting forests from giant clear-cutting projects, and abolishing factory livestock feeding operations will contribute to climate change mitigation as well as food sovereignty (de Haan et al. 2001).

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[1]Vlek et al. (2004) proposed that carbon sequestration would be best managed with intensified agricultural production on the better lands, using increased fertilizer inputs. They argued that the additional CO_2 from the extra fertilizer would be outweighed by sequestered carbon and avoided loss of organic carbon from deforestation. But fertilizer use is only one option for intensification. Others are better land and water management and better choices of livestock (Steinfeld et al. 2006). The CAFO form of intensification is not suited to many production locations because they require balancing more difficult chemical, biological, and social factors in ecosystems.

BIODIVERSITY, MONOCULTURES, AND LAND CONVERSION

Overview of land use changes

Agricultural Intensification and biodiversity reduction: Large scale monocultures continue to transform the world from landscape mosaics of small and diverse farms into large areas of extremely low biodiversity (Tilman 1999), both in terms of the planned diversity (the crops, animals and varieties that are intentionally included in the system by the farmers), as well the associated diversity that live or temporary use the agroecosystem (Vandermeer and Perfecto 2005a). Today, entire regions of the world are dominated by virtual monocultures, and the world's food supply depends on very few crops. Indeed, only 4 annual grasses (barley, maize, rice and wheat) occupy close to 600 million ha, representing almost 40% of the global cropland (Tilman 1999). Recently there has been a dramatic increase in a few crops such as soybean and African oil palm that are grown almost exclusively in monocultures (Donald 2004), contributing further to the loss of biodiversity in the rural landscapes. According to the International Assessment of Agricultural Knowledge, Science and Technology for Development (IAASTD 2009), the highest emissions of GHG from agriculture are associated with these intensive monocultures.

Industrial agriculture deliberately maintains the agroecosystem in a simplified (low diversity), disturbed state (Tilman 1999). Subsequently many of the ecological interactions and ecosystem processes that provide ecosystem services in diverse systems are disrupted (Matson et al. 1997, Altieri 1999, Tilman 1999, Vandermeer et al. 1998), requiring the application of agrochemicals and contributing further to GHG emission.

Diversity effects on soil processes: Soil organisms contribute a wide range of essential services to the sustainable function of agroecosystems. They have a role in nutrient cycling, soil carbon sequestration and GHG emissions (as discussed earlier), building soil physical structure and maintaining water regimes (Swift and Anderson 1999). These soil processes and properties are regulated by a highly diverse soil community of microbes and invertebrate animals (Giller 1996). Healthy soils are comprised of a highly diverse soil biota including representatives of all groups of micro-organisms, fungi, green algae, and cyanobacteria (Lee 1991). Other important members of soil biota include earthworms, termites, ants, some insect larvae and other soil animals that help aerate and turn the soil and create bioturbation (Lavelle 1997). Agricultural intensification and monocultures frequently lead to a decline in the diversity of the soil biota, and the replacement of soil ecosystem functions by chemical and mechanical inputs. Although the decline in soil biota with agricultural intensification can lead to the loss of ecosystem services, the causal relationship still remains to be quantified for most processes, and it is not clear how this impacts the function of the agroecosystem (Swift 1997). However, from studies of keystone organisms such as earthworms, termites, mycorrhizal fungi, and nitrogen-fixing bacteria it is evident that the reduction in the diversity of the soil biota can have profound effects on the biological regulation of decomposition and nutrient availability in soil (Beare et al. 1992, Matson et al. 1997), although this relationship is not always present (Wardle et al. 2000). Furthermore, there is evidence that plant diversity affects some soil ecosystem processes. For example, the rate of loss of limiting nutrients from terrestrial ecosystems has been found to be lower at high plant diversity and is also impacted by plant species composition (Naeem et al. 1996, Tilman et al. 1997). Likewise the recovery of soil carbon and nitrogen from highly degraded soils can be accelerated if fields are planted with a high-diversity mixture of appropriate plant species (Knops et al. 2002). Given that, it is not surprising that diverse agroecosystems have been shown to sequester more C in soil and biota than those with reduced biodiversity (Lal 2004c, IAASTD 2009). Therefore, the transformation of diverse landscape mosaics and diverse agricultural systems to large-scale monocultures not only reduces the GHG sequestration potential of the soil but also increases the need for fertilizer application, which in intensive monocultural systems is almost always synthetic fertilizer, adding even more to GHG emission.

Diversity effects on pests and diseases: The transformation from diverse agroecosystems to monocultures also alters the community composition of the pest complex including herbivorous insects and their natural enemies, as well as the microbial pathogens that attack plants (Power and Fletcher 1996). A fundamental principle in ecology and epidemiology is that the distribution of the host population influences the severity and extent of a disease or pest outbreak. In a homogeneous host population (where all individuals are susceptible) diseases and pest spread faster than in a heterogeneous population (Otten et al. 2005). The reduction in biodiversity typically results in greater crop losses due to insect pests in monocultures compared to diverse agroecosystems (Russell 1989, Andow 1991, Finckh et al. 2000). This trend is particularly strong for specialized herbivores and results from interference with host-finding and insect movements in heterogeneous environments (Bach 1980, Andow 1991). However, it is also well documented that diverse systems have higher levels of natural enemies (predators and parasitoids) that can control insect pests (Letourneau 1987, Russell 1989, Perfecto at al. 2004). The effect of plant diversity on microbial pathogens is less predictable than on insects and generalizations are difficult because the effect of plant diversity depends on several dispersal processes, infection efficiency and rate of disease progress (Matson et al. 1996, Mundt 1992). However, pathogens transmitted by insect vectors tend to have lower incidence in polycultures due to the effect of plant diversity on the movement of the insect vectors (Power 1991).

Greater plant genetic diversity also leads to lower incidence of plant pathogens (Browning and Frey 1969, Wolfe 1985, Mundt 2002). Both multiline cultivars and varietal mixtures have been shown to retard the spread and evolution of fungal pathogens and viruses. For example, in a large scale experiment in China over a 5 township region, disease susceptible rice varieties planted in mixtures with resistant varieties had 89% greater yield and the rice blast disease was 94% less severe than when they were grown in monocultures (Zhu et al. 2000). Furthermore, this effect is likely to apply also at the landscape level. Metapopulation theory and mesocosm experiments indicate that landscape mosaics with non-susceptible hosts (other species of non-susceptible varieties of the same species) slow the spread of an epidemic as compared to a homogeneous landscape, both in plants (Gilligan 2002; Park et al. 2001, 2003; Otten et al. 2005), and animals (Swinton et al. 1998).

Plant diversity and the stability and productivity in agroecosystems: The relationship between diversity and productivity of ecosystems is not well understood by ecologists yet. Although a substantial body of literature exists suggesting that diversity increases the productivity of ecosystems (Tilman 1999, Naem and Li 1997), there are also objections to this assertion (Huston et al. 2000). On the other hand, the evidence that certain crop combinations result in higher yields is uncontested. For example, the combination of grasses and legumes almost always yields more than the monocultures of either one (Vandermeer 1989). The mechanisms of this polyculture overyielding effect is well understood, and includes reduced competition and facilitation mainly through the nutrient release of the legume (Vandermeer 1989). The traditional maize/bean combination is one of the best known examples of this phenomenon and the overyielding of this intercropping system involve reduced competition for nitrogen (due to the nitrogen fixation of the *Rhyzobium* bacteria in bean nodules), reduce pest and diseases, and the provision of a climbing structure to the bean plant by the maize. A four-year experiment with maize and faba bean found that maize overyielded by 43% and faba bean by 26% in phosphorous deficient soils. This study demonstrated that the mechanism for overyielding in the faba bean was reduced competition due to deferential rooting depth of the two crops, and the overyielding of the maize resulted from its uptake of phosphorous mobilized by the acidification of the rhizosphere via faba bean root release of organic acids and protons (Li et al. 2007).

All else being equal, the stability (year to year variation) of the total rate of plant production in an ecosystem depends on plant species diversity and composition (Naeem and Li 1997, Schultz et al. 1983), stability of primary productivity being greater for more diverse ecosystems (Tilman 1999). Applied to agricultural systems this means that diverse agroecosystems minimize year-to-year variance in yields, therefore providing an insurance value to biodiversity. Models of stability and productivity in ecosystems have shown that the effect of adding a species to a monoculture system will be larger than adding the same species to a multispecies system, suggesting that the effects of diversification will be largest when increased diversification is applied to monoculture systems (Norberg et al. 2001).

Given that both productivity and the stability of yields are generally improved in more diverse systems, the implications for GHG are indirect but clear. More diverse farming systems, because they promote higher productivity and stability may sometimes reduce the need for synthetic fertilizer application, therefore reducing the impact of agriculture on GHG emissions.

Loss of landscape level diversity: The intensification of agriculture not only results in a reduction of species and genetic diversity at the farm level but also at the landscape level. The expansion of the large monocultural plantations and intensive agricultural systems implies the homogenization of the rural landscapes and the loss of diverse features such as live fences, riparian corridors, woodlots, hedge-roads, patches of natural forests and other natural habitats (Altieri 1999). We now know that these landscape features also provide ecosystem services to the agroecosystem (Tscharntke et al. 2005). For example, several studies have shown an increase abundance of natural enemies and more effective biological control in agricultural plots surrounded by wild vegetation (Boatman 1994, Fry 1995). This habitat may be important as overwintering sites for natural enemies of pests and can provide increased resources in the form of alternative host/prey and pollen for predators and parasitoids (Landis 1994). Likewise nearby forests and natural habitats have been found to increase pollination services of native bees (DeMarco and Coelho 2004, Kremen et al. 2004). Biodiversity at the landscape level, including diversity of functional groups, is also important for the resilience (the capacity to recover after disturbance) of the agroecosystem (Bengtsson et al. 2003, Loreau et al. 2003). As rural landscapes become more simplified and dominated by vary large monocultures, these ecosystem services have to be substituted with chemicals or mechanical inputs, thus increasing the GHG footprint of agriculture.

Diversified agroecosystems to curb GHG emissions: Given the information in the previous section it seems evident that the long-term sustainability of agroecosystems and the ecosystem services they generate depend on the conservation of biodiversity at both the farm and the landscape level (Matson et al. 1997, Bengtsson et al. 2003, Loreau et al. 2003, Tscharntke et al. 2005). It is also obvious that the current industrial agricultural system reduces diversity at the farm level and creates simplified low diversity landscapes that may contribute significantly to GHG emissions and therefore, to global warming. On the other hand, small scale diversified family farms, by maintaining diversity at the farm and landscape levels, contribute to the maintenance of ecosystem processes and services reducing the need to rely on imported inputs (Altieri 1999). The GHG emissions that are the result of pesticide and fertilizer production could thus be greatly reduced by the diversification of farms and rural landscapes. Fortunately there are still many small-scale low-input family farms, especially in tropical regions.

Agroforestry Systems: Agroforestry is the production of livestock or food crops in combination with growing trees, either for timber, firewood, or other tree products (Montagnini and Nair 2004). Some of these systems, especially the traditional ones, can contain high species diversity within a small area of land (Leakey 1999, Kumar and Nair 2006). Not only do they provide diversity of crops in time and space, but also protect soil from erosion and provide litter for organic material and soil nutrients (Young 1994, Jama et al. 2000), reducing the need for synthetic fertilizer.

Evidence is emerging that agroforestry systems have great potential for increasing above ground and soil C stocks, reduce soil erosion and degradation, and mitigate GHG emissions (Mutuo et al. 2005). In agroforestry systems the standing stock of the carbon above ground is usually higher than the equivalent land use without trees (Smith et al. 2008). In a review of 42 studies it was estimated that the C sequestration potential of agroforestry was 2.6, 3.9, 6.1, 10 Mg C/ha/yr for the semi-arid, temperate, sub-humid and humid regions respectively (Schroeder 1994). In a study of 10 year-old agroforestry systems with *Erythrina poeppigiana*, sequestered C was attributed to 0.4 Mg C/yr in coarse roots, 0.3 Mg C/yr in tree trunks, 1.4 Mg C/yr in tree branches and leaves added to the soil as mulch, and 3.0 Mg C/yr from crop residues. The latter two contributions resulted in an annual increase of SOC pool by 0.6 Mg C (Oelbermann et al. 2004). For smallholder agroforestry in the tropics, potential C sequestration rates range from 1.5-3.5 Mg C/ha/yr (Montagnini and Nair 2004). Furthermore, in degraded soils in the sub-humid tropics, improved fallow agroforestry practices have been found to increase top soils C stocks up to 1.6 Mg C/ha/yr above continuous maize cropping. These soil carbon accretions have been linked in particular to increasing C in water stable aggregates (Mouto et al. 2005). As for other land-use systems, the extent of C sequestered will depend on the amounts of C in standing biomass, recalcitrant C remaining in the soil, and C sequestered in wood products. Agroforestry systems with perennial crops, such as coffee and cacao, may be more important carbon sinks than those that combine trees with annual crops (Mantagnini and Nair 2004).

The potential of agroforestry for aiding in the curbing of GHG emissions is not limited to carbon sequestration. A review of agroforestry practices in the humid tropics shows that these systems were also able to mitigate N_2O and CO_2 emissions from the soils and increase the methane sink strength compared to annual cropping systems (See table; Mutuo 2005). In a study of the Peruvian Amazon, a tree-based agroforestry system emitted less than a third $N_2$0 than a high (fertilizer) input annual cropping system, and half of the low input cropping system (Palm et al. 2002). However, in a study of improved follows in Kenya, CO_2 and N_2O emissions increased after the addition of legume residues (Millar et al. 2004), highlighting the importance of using low-quality organic inputs and increasing nutrient use efficiency to optimize the reduction in the GHG footprint of the system (Mutuo et al. 2005). Data from several countries strongly suggest that agroforestry systems can partially offset CH_4 emissions, while conventional high-input systems exacerbate CH_4 emissions (See table; Tomich et al. 1998, Tsuruta et al. 2000, Hairirah et al. 2001).

Table 2. Average fluxes of N_2O, CH_4 and CO_2 in cropping systems, agroforestry practices and forests in slash-and-burn systems in the Peruvian Amazon and lowland humid tropics in Sumatra, Indonesia.

Land-use system	Land-use system	N_2O emission ($\mu g\ N\ m^{-2}\ h^{-1}$)	CH_4 flux ($\mu g\ C\ m^{-2}\ h^{-1}$)	CO_2 emission ($\mu g\ C\ m^{-2}\ h^{-1}$)	Source
Cropping systems	High input cropping	31.2	15.2	84.0	Palm et al. (2002)
	Low input cropping	15.6	− 17.5	66.6	Palm et al. (2002)
	Cassava/*Imperata*	7.1	− 14.8		Tsuruta et al. (2000)
Agroforestry systems	Shifting cultivation	8.6	− 23.5	67.5	Palm et al. (2002)
	Multistrata agroforestry	5.8	− 23.3	62.6	Palm et al. (2002)
	Peach palm	9.8	− 17.0	66.4	Palm et al. (2002)
	Jungle rubber	1.0	− 12.0		Tsuruta et al. (2000)
	Rubber agroforests	12.5	− 27.5		Tsuruta et al. (2000)
	R. agroforests (cloning)[1]	–	− 29.0		Tomich et al. (1998)
Forests	Forest	9.2	− 28.8	73.3	Palm et al. (2002)
	Forest	5.0	− 31.0		Tsuruta et al. (2000)
	Logged forest	7.2	− 38.2		Tsuruta et al. (2000)

[1]Rubber agroforestry with cloning.

A global analysis of biological and economic data from 94 nations representing diverse climatic and edaphic conditions reveals a range of integrated land-use systems that could be used to establish and manage vegetation on marginal or degraded lands as a way to sequester carbon. Give the demonstrated potential of agroforestry systems for reducing GHG emissions and increasing their uptake, it is not surprising that these systems include agrosilviculture, silvopastoral, and agrosilvopastoral systems (Dixon et al a or b??. 1994).

Afforestation versus Agroforestry: Afforestation, the establishment of forests on land that is not a forest, typically takes the form of large-scale monocultural tree plantations. A study that analyzed the changes in carbon cycle that could be achieved with a global, large-scale afforestation program, concluded that 350 million hectares would be available for plantations and these will only be able to sequester a total of 104 Gt C over a 100 year period (Nilsson and Schopfhauser 1995). Afforestation studies of agricultural land in the US show that soil C can change from -0.07 to 0.55 Mg C/ha/yr on deciduous sites, and from -0.85 to 0.58 MG C/ha/yr under conifers. Soil N changes under afforestation ranged from -0.1 to 0.025 Mg N/ha/yr However, even after 20-50 years of afforestation in many sites, the C/N ratios remained more similar to agricultural systems than native forests, showing that chemical replenishment of soils is a slow process (Paul et al. 2003). This may suggest that maintaining highly diverse agroforestry sites are more beneficial to the maintenance of nutrient rich soils than afforestation of previously intensively farmed land.

Deforestation and Other Land Conversions

The world's forests and savannahs have long helped maintain the global carbon cycle in balance, but the industrial agriculture system has contributed both directly and indirectly to the dismantling of this particular ecosystem service. Although the small-scale farmer is frequently blamed for forest clearing, a closer examination reveals a complex link between deforestation caused by small-scale agriculturalists and that caused by large-scale farms, especially in a context of globalized food production and food commodification (Vandermeer and Perfecto 2005a). In response to increasing demand for food and the latest hot commodity (biofuels), industrial agriculture is rapidly advancing on the world's remaining native habitats with important consequences for climate change. Here we review the literature on the impacts of tropical deforestation on GHG emissions and uptakes, and analyze the drivers of deforestation.

Impact of deforestation on GHGs: Deforestation releases CO_2 and reduces its uptake by plants. The IPCC report estimates that land use changes, mainly deforestation, contribute 20% of the CO_2 emissions globally (Salomon et al. 2007). The other 80% is due to fossil fuel burning and cement production. Although worldwide cropland expansion was lower after 1950 than before, deforestation is occurring more rapidly in the tropics. Latin America, Africa, and South and Southeast Asia have experienced exponential increases in cropland expansion since 1950 (IPCC 2007, Chapter 2). By 1990 cropland and pastures covered 45.7 to 51.3 million km^2 (35-39% of global crop lands), and forest cover decreased by 11 million km^2 (Ramankutty and Foley 1999, Klein Goldewijk 2001). CO_2 emissions have continue to increase over the last few decades, and emissions associated with land use changes averaged over the 1990s is estimated to be 0.5-2.7 Gt C/yr (Solomon et al. 2007).

Carbon uptake and storage in the terrestrial biosphere arise from the net difference between uptake due to vegetation growth, changes in reforestation and sequestration, and emissions due to heterotrophic respiration, harvest, deforestation, fire, damage by pollution and other disturbance factors affecting biomass and soils (IPCC 2007: Salomon et al. 2007). As discussed in the previous section, the type of agriculture also contributes to CO_2 uptakes and emissions, with small-scale diverse agroecological and agroforestry systems contributing to a reduction in emissions and increased C uptakes. Studies of net CO_2 fluxes in the terrestrial biosphere show significant uptakes in mid-latitudes of the Northern Hemisphere, probably due to the stability of forests and forestry plantations in those regions (IPCC 2007: Salomon et al. 2007). Surprisingly, the IPCC (Salomon et al. 2007) concludes that the tropics are either carbon neutral or sink regions, despite widespread deforestation, a result that has been verified more recently (Lewis et al. 2009). This implies that tropical rainforests are net carbon sinks (Phillips et al. 1998, Malhi and Grace 2000). Thus if tropical deforestation continues unabated, not only will a significant reservoir of the Earth's carbon be released into the atmosphere, but a critical sink for near-term emissions will be destroyed.

Although deforestation has the strongest impact on CO_2 emissions, it also contributes to other GHGs. From data presented in the IPCC report (Salomon et al. 2007; Table 7.6) we estimate that CH_4 emissions from biomass burning represents 6-8% of the total anthropogenic emissions, which in turn represent 60% of the total. Most of these emissions come from deforestation, but some can also be attributed to pasture and firewood burning. On the other hand, current data from different countries confirm that upland primary and secondary forests are CH_4 sinks (average monthly CH_4 consumption rate of 30 *ug* C/m^2/ha). In contrast, intensive agricultural systems can decrease the sink strength by 50% or more or even revert to soil methane emissions (Palm et al. 2002, Mutuo et al. 2005). What we know today about CH_4 and forests suggest that converting tropical rainforests into intensive farming systems, which is what is happening in many tropical regions of the world, represents a triple impact on the CH_4 budget. First, the elimination of the CH_4 sink from the actual deforestation, then the emission of CH_4 from biomass burning, and third, the emissions of CH_4 from (fertilizer) intensive agriculture.

Conversion of tropical savannas: Tropical savannas have been heavily impacted by human activity, with large extensions of land converted from tree-grass mixtures to open pastures and agriculture (Solbrig et al. 1996). Dominant land-use in moist tropical savannas include beef cattle production and large scale intensive agriculture, although in Africa shifting and permanent cultivation is also practiced (Hoffman and Jackson 2000). The Brazilian *cerrado* is one of the world's biodiversity hotspots and is one of the most threatened savannah systems in the world, with more than 50% of its original 2 million km^2 transformed into pasture and agricultural lands for cash crops, mainly soybean (Klink and Machado 2005). It is estimated that up to 70% of the *cerrado* biomass is underground, depending on the dominant vegetation (Castro and Kauffman 1998). When this ecosystem, or other savannas are transformed to pastures or annual agriculture, the carbon stocks are altered, with the degree of the alteration depending on the extent of the modification. Planted pastures may accumulate carbon, if they are well managed, but most of the pastures in the *cerrado* are degraded and fail to serve as atmospheric C sinks

(Silva et al. 2004). The transformation of the cerrado to large scale soybean plantations has a strong impact on GHG emissions (See Case study 1, below, *Large-scale cattle pastures and monocultures in the Brazilian Amazon*). It has been shown that allowing cropland to revert to grassland will reduce the GHG emissions. Converting arable cropland to grasslands typically results in the accrual of soil C owing to lower soil disturbance and reduced C removal in harvested products. Compared to cultivated croplands, grasslands may also have reduced N_2O emissions from lower N inputs and higher rates of CH_4 oxidation (Smith et al. 2008).

Introducing grass species with higher productivity or C allocation to deeper roots has been shown to increase soil C. The establishment of deep-rooted grasses in savannas has been reported to yield high rates of C accrual (Fisher et al. 1994). Introducing legumes in grazing lands, and mixed crop system can also promote soil C storage (Soussana et al. 2004). This practice may also reduce N_2O emissions. Slowing degradation by alternative grassland management and by impeding desertification could conserve up to 0.5-1.5 Pg C annually (Dixon et al. 1994 a).

Drivers of tropical deforestation: Although the small-scale farmer is frequently blamed for forest clearing, most detailed studies of deforestation link deforestation activities of small-scale agriculturalists to pressure from larger scale agricultural activities, such as cattle ranching and extensive monocultures (Vandermeer and Perfecto 2005a, b).

The heavy promotion of industrial monoculture plantations and agrofuels as solutions to the current food and energy crises actually increase pressure on agricultural land, leading to more deforestation (Perfecto and Vandermeer, in press) and more GHG emissions (IPCC 2007). This has already led to massive land grabbing by transnational companies in developing countries, forcing farmers and indigenous communities off of their lands. ‘Idle’ land is being identified by states and firms for expansion of commercial agrofuels. However, growing evidence raises doubts about the concept of idle land. In many cases, lands perceived to be idle, under-utilized, marginal or abandoned by government and large private operators, actually provide a vital basis for the livelihoods of poorer and vulnerable groups through crop farming, herding and gathering of wild products (Cotula et al. 2008: 22-3; see also Paul et al. 2003: 29).

Tropical forest degradation and destruction occur in synergistic ways. In Latin America, timber extraction is usually directed at low-density, commercially valuable tree species, but damage to other trees can be almost double the volume of harvested trees (Veríssimo et al. 1992). In the dipterocarp-rich forests of Indonesia, logging operations can affect up to 77-87% of an area (Curran et al. 1999). Logging and drought increase forest susceptibility to fires, and initial fires in the understory create conditions that facilitate subsequent canopy burns, resulting in a positive feedback (Nepstad et al. 1999). As forest cover declines and becomes fragmented, local rainfall patterns change, and drying near fragment edges creates an additional feedback loop that degrades forest through fire and water stress (Nepstad et al. 2008). Logging roads provide access to farmers and ranchers who convert forests to pastures and croplands (Asner et al. 2006). Farmers often invest the income generated from selling the remaining trees in the landscape into further land conversion (Merry et al. 2002).

The role of small to medium-scale farmers versus large-scale industrial agriculture in deforestation is at the center of an ongoing debate in conservation (Wright and Muller-Landau 2006, Sloan 2007). While the proximate drivers of deforestation can be directly attributed to different types of agriculture, the underlying causes are much more complex. Small- and large-scale farmers are linked in complicated ways through markets, social forces, and policy, which all operate at different scales (Lambin et al. 2001). Blaming a family farmer for cutting down forest to grow cassava in Africa ignores the socioeconomic conditions that favor this type of production. Assigning blame also limits discussion of how policy can intervene to promote both farmers' well-being and forest conservation. Yet it is increasingly clear that the expansion of industrial agriculture in response to global demand for food and biofuels is having a detrimental effect on the world's remaining forests. To mediate global warming, we must understand both the direct and indirect impacts that large-scale industrial agriculture has on forests.

Among the proximate drivers of tropical deforestation, conversion to agriculture is the most important. The contribution of small versus large-scale agriculture to deforestation differs among continents. In tropical Latin America, 47% of deforestation from 1990-2000 was due to direct conversion to large-scale permanent agriculture, including livestock, soybeans, biofuel crops, industrial tree plantations, fruits, vegetables and cut flowers, whereas 13% was due to direct conversion to small-scale permanent agriculture (FAO 2009). In contrast, 59% of deforestation in tropical Africa during the same period was caused by small-scale permanent agriculture, as opposed to 12% by large-scale permanent agriculture. In tropical Asian and Pacific countries, the contribution of each type of agriculture was more balanced. Small-scale permanent agriculture caused 13% of deforestation while expansion of shifting cultivation into undisturbed forests depleted another 9%, compared to 29% caused by large-scale permanent agriculture. While large-scale industrial agriculture caused a significant portion of deforestation in Latin America and Asia, smallholders were the main contributors to deforestation in Africa.

Large-scale industrial agriculture drives deforestation in indirect ways as well. Far from the agricultural-forest frontier, industrial agriculture can 'push' smallholders to the frontier by consolidating land and replacing labor-intensive production with capital-intensive production (Angelsen and Kaimowitz 2001). Large-scale, labor-intensive agriculture at the frontier can 'pull' migrants to work on commercial plantations, as in the case of resettlement projects in southeast Asia (Lambin et al. 2001). Infrastructure in the form of roads, electrification, health services, etc. stimulates development and attracts land-seeking families. Public perception and some conservation policies tend to ignore these complexities regarding deforestation. The two case studies below serve to elucidate the complex ways that large-scale agriculture plays a role in tropical deforestation.

Regional case studies: Perhaps the two regions where industrial agriculture expansion is having the most impact on native habitat, and where the stakes are highest for global GHG emissions, are the Amazon and Southeast Asia. The Amazon rain forest represents one of the largest and most dynamic reservoirs of terrestrial carbon in the world. Fearnside (2000) estimates that Brazilian Amazonia represents 38% of the total carbon stock in the tropics. Amazonian trees contain 90-140 Pg C, which is roughly equivalent to 9-14 years of current global, annual emissions due to all human activities (Nepstad et al. 2008). The forests of Southeast Asia represent the second largest carbon stock. Deforestation rates in these two regions are among the highest in the world, largely due to the advancement of industrial-scale monocultures and, in the Amazon, because of extensive cattle ranching. In Case study 1 we analyze the expansion of cattle ranching and soybean plantations into the Amazonian region, and it's impact on GHG. In Case study 2 we analyze the situation in Southeast Asia where millions of hectares of rainforests are being converted to African oil palm for biofuel production.

Case study 1. *Large-scale cattle pastures and monocultures in the Brazilian Amazon*

Amazonian forests covered about 5.4 million km^2 in 2001, approximately 87% of their original extent (Soares-Filho et al. 2006). Sixty-two percent of the Amazon lies in Brazil, where deforestation rates vary widely from year to year. From 1988 to 2006 deforestation averaged 18,100 km^2/yr, reaching a maximum of 27,400 km^2/yr in 2004, and then declining to approximately 11,000 km^2/yr in 2007 (Malhi et al. 2008). Roughly 62% of deforested land consists of pasture, 6% cropland, and 32% secondary vegetation (Ramankutty et al. 2007).

Land distribution in Brazilian Amazonia is highly skewed in favor of large landowners. Landholdings >2000 ha comprise 47% of land converted from forest or *cerrado* to agriculture, even though they represent only 1% of all establishments (Chomitz and Thomas 2001). In contrast, holdings <20 ha constitute 54% of the total number of establishments but cover merely 1.5% of deforested area. The smallest farms (<10 ha) are primarily subsistence-oriented, with manioc and rice constituting 30-40% of production. Farms 20-100 ha in size diversify manioc with commercial products such as milk and bananas. Large and very large farms ranging from 100-100,000 ha in size are dominated by cattle and soybeans, with sugarcane becoming an important crop on the largest farms in this range. Establishments spanning >100,000 ha are used for silviculture. Large holdings are concentrated in southern Amazonia, whereas small (<20 ha), subsistence-oriented holdings are mostly scattered throughout western Amazonia (Chomitz and Thomas 2001).

As the overall dominant land use, large-scale cattle ranching continues to be the preeminent force behind deforestation in Brazilian Amazonia. Large and medium-sized ranches account for about 70% of clearing activity (Fearnside 2005). The 74 million head of cattle in Brazilian Amazonia occupy 84% of the total area under agricultural and livestock uses and have expanded 9%/yr, on average, over the last 10 years (Nepstad et al. 2008). Despite widespread use, extensive cattle ranching is highly unproductive on a per area basis. About 60% of pasture currently in use has a mean stocking rate of 0.95 cattle/ha, while 40% has less than 0.5 cattle/ha (Chomitz and Thomas 2001). Stocking rate declines with farm size. Many authors criticize cattle ranching because the social benefits fall dramatically short of the environmental costs (Kaimowitz et al. 2004, Nepstad et al. 2006). Nevertheless, large ranchers accrue benefits in the form of profits from land speculation and high returns to labor (Hecht 1993, Angelsen and Kaimowitz 2001).

Decades-old expansion of cattle ranching is now converging with rapid growth in industrial monocultures to meet the rising global demand for grains, meat, wood and biofuel crops. As land appropriate for such agro-industrial commodities becomes scarce in the USA, Western Europe, China, and many other countries, agribusinesses are looking to expand in regions with large agricultural frontiers like Amazonia. Soybean monoculture cultivation represents the most rapidly growing sector in Amazonian agriculture (Nepstad et al. 2006). Although soybean is primarily grown on areas that were once pasture or natural *cerrado*, some expansion is occurring directly on forest land. In Mato Grosso, the Brazilian state with the highest deforestation rate, 17% of total deforestation during 2001-2004 resulted from direct conversion of forest to large-scale mechanized cropland, primarily soybean (Morton et al. 2006). Area cleared for cropland and mean annual soybean price were highly correlated, suggesting that deforestation will increase as international demand for this commodity continues to rise (Morton et al. 2006). In addition to direct forest conversion, soybean expansion contributes to deforestation in indirect ways. As soybean producers buy up land from ranchers in the south, the newly capitalized ranchers move to the north where land prices are lower and they can expand their herds, pushing the agricultural frontier (Nepstad et al. 2008). Soybean production also stimulates massive government investment in infrastructure such as waterways, railways, and highways, which unleashes private investment, profiteering, and further land conversion (Fearnside 2001). A similar process of commodity market-driven soybean expansion is underway in the Bolivian Amazon (Hecht 2005). Tropical forest conversion to soybean for biodisel has an estimated carbon debt of >280 Mg CO_2/ha and will require 320 years to repay as compared to GHG emissions from petroleum based biodisel (Fargione et al. 2008) .

Case study 2: *Oil palm in Indonesia and Malaysia*

Mature oil palm reached 14 million ha worldwide in 2007, with Indonesia and Malaysia dominating world production. In Indonesia and Malaysia, 56% and 55-59%, respectively, of oil palm expansion occurred at the expense of forests during 1990-2005 (Koh and Wilcove 2008). These two countries are expanding oil palm plantations with the goal of supplying 20% of EU biodiesel needs. India plans 14 million ha of land for jatropha plantations, and Africa another 400 million ha (Holt-Giménez 2007, Vidal 2007: 3). Major investments are underway to convert millions more hectares of forest and other land uses to oil palm (Sheil et al. 2009). Palm oil plantations are encouraged by tax breaks, subsidies, and huge investments by China's National Offshore Oil Corporation, and by oil and agribusiness firms like Shell, Nestle Oil, Greenergy International, BioX, Cargill and Archer Daniels Midland (Smolker et al. 2008: 29).

Smallholders are responsible for 37% of the total area under oil palm production in Indonesia and 40% in Malaysia. Smallholders are defined as family-based enterprises producing on less than 50 ha, and include wage laborers on plantations whose involvement is not always voluntary (Vermeulen and Goad 2006). Conversely, large plantations comprise 63% of the total area under palm oil production in Indonesia and 60% in Malaysia.

Ethnographic research on oil palm plantations in Kalimantan, Indonesia, confirms the combined social and ecological effect of agrofuel expansion: ‘Forest and land availability have been greatly reduced, making it more difficult for the local communities to obtain non-timber forest products and leading to a lack of farming lands. As there are not enough farming lands, farming has become more intensive. The same lands are used continuously, so that the soil does not have enough time to regain fertility. As there is not enough arable land, many people have given up rice farming and a linear regression can be seen in the diversity of crops cultivated in relation to the proximity of the plantation. Availability of, and access to foods such as meat, vegetables and fruits has declined, so that more food has to be bought, leading to higher food expenses’ (Orth 2007: 51).

Converting lowland rainforest in Indonesia and Malasia to Palm biodisel will result in a carbon debt of approx 610 Mg of CO_2/ha that will take 75-93 years to repay (Danielsen et al. 2008, Fargione et al. 2008). Until then, producing and using palm bio-disel will produce more GHG releases than refining and using an energy equivalent amount of petroleum based biodisel (Fargione et al. 2008). If the original habitat was peatland, the time required would be 600-840 years (Danielsen et al. 2008, Fiergione et al. 2008).

Reducing emissions from deforestation and degradation (REED): Mechanisms to reduce carbon emissions from deforestation and forest degradation (REDD) have been gaining momentum as a way to combat global warming, fund forest conservation, and deliver economic benefits to rural populations. It has been estimated that cutting global deforestation rates 10% could generate up to $13.5 billion in carbon credits under the REDD initiative approved at the U.N. climate talks in Bali this past December (Ebeling and Yasué 2008). However, it has been recently demonstrated that converting forest to oil palm in Southeast Asia is more profitable than conserving it for a REDD project (Butler et al. 2009). Furthermore, with the inclusion of practices such a no-tillage, it might be possible in the future for a soybean farmer in Brazil that uses herbicide resistant transgenic soy and plants thousands of hectares in soybean monoculture, to receive payment for carbon credits. Indeed, soybean farmers in Canada are already receiving payment for their no-till practices. As discussed above, no-till practices reduce CO_2 emissions, while the system as a whole can have a net negative impact on GHG emissions.

REDD and other proposals such as the biochar initiative, and the promotion of no-till agriculture and climate resistant transgenic crops, can potentially benefit large-scale agribusinesses while further marginalizing small farmers. As we have seen with the rapid expansion of large scale industrial plantations for biofuels, solutions that favor agrobusiness have the tendency to results in massive land grabbing by transnational companies in developing countries, the displacement of farmers and indigenous communities out of their territories, and further increases in GHG.

FROM ENERGY PRODUCER TO ENERGY CONSUMER

Nature provided us with a spectacularly useful machine that harvests energy from the sun and makes it available for our use. And it is free. We have made use of this energy machine for our evolution and survival as a species to an extent that has never been rivaled by any other species in the 3.5 billion years of life on earth. It is a complicated process in which the energy raining down on the earth from the sun is captured in a green-colored chemical (chlorophyll), converted to energy-rich chemicals, and stored in the leaves, stems, trunks, fruits and seeds of plants. When we eat those seeds or fruits or leaves, we capture that stored energy and use it for our own purposes, to do all the things we have always had to do to survive and reproduce. It was a machine that provided energy effectively for free.

That is what nature gave us. Approximately 10,000 years ago we found a way to improve on nature. When we invented agriculture we precisely modified the way nature organizes our energy producing system to make it work better for us, and to make it generate even more energy for us. At first we saw in nature some underlying rules about how it is organized - some rules and laws that are at the base of keeping the system operative. Our agricultural methods were designed with those rules in mind to create a system in which nature was an even more efficient provider of our energy than it had originally been. Agriculture was a way to expend a small amount of additional energy (e.g., plant a seed) to get a great deal more energy produced (harvesting the seed heads).

All along the way small farmers acted as basic scientists, asking questions about their production processes and doing experiments. They were always trying something new to improve on their system, and sharing with their neighbors when something worked well. In this way agriculture has been, for the past several thousand years an immensely dynamic energy producing system with a small-scale structure that almost guaranteed continual adjustments and improvements occurring at a local scale through those scientists called farmers. The result was a myriad of agricultural systems the world over, each attuned to the particularities of local environmental factors and each functioning in a way to increase the efficiency of the extraction of energy from the sun.

But then we discovered something about that fundamental natural process of energy capture and storage. We discovered that for the past 600 million years or so, the decomposition of the storage materials of energy has been incomplete. That is, rather than a complete cycle of carbon (from carbon dioxide in the air, to storage in the bodies of plants and animals, to decomposition of those bodies after death to release carbon dioxide into the air again) maintaining a balance, and rather than the ecosystem using all the energy it produces, a whole lot of that energy and that carbon was stored below the ground. Imagine, for example, a tree that grows for 50 years, all the time storing the energy it gets from the sun in its trunk. We can burn that trunk for its energy (to warm the house or cook the meal) and get 50 years worth of energy in an hour or so. But now imagine a forest of trunks that accumulate energy for 50 million years, or 100 million years, storing the energy below the ground where its fallen trunks are protected against the action of fungi and bacteria. If we can find those trunks we can burn them and quickly get the energy that has been stored over those millions of years. This, of course, is what fossil energy is all about. We had discovered that we had not only an efficient energy production system (from plants and the sun), but that we also had a treasure trove of energy under the surface of the earth, put there over a long period of time by this same process.

This discovery led to the Industrial Revolution which was, from a very general point of view, the use today of energy produced long ago. It was this new energy system that allowed us to do all sorts of things, from using iron to make machetes, to traveling by bus, to construction of cities. As marvelous as that revolution was, it ironically has changed the fundamental energy system we use in agriculture. All the intricate and complex methods that farmers had been developing over millennia suddenly came into the view of industrialists. The men and women who figured out how to make automobiles looked at agriculture and thought, why not apply the same principles there. So, rather than slowly transforming the soil decomposition process so as to better release the right amount of nitrogen to the soil, the new industrialists asked, why not make the nitrogen using all the energy we have stored away in coal and petroleum and not worry about the decomposition process in the first place? Rather than timing rotations and planting designs to minimize the attack rate of an insect that could easily become a pest, the new industrialists asked, why not make a poison using all the energy we have stored away in coal and petroleum and not worry about the timing or rotations or pattern of intercropping? In short, the new industrial farmer said we can replace the thought-intensive technology we have been using for so many years with a brute force energy application, and we can do it because we have so very much of that energy stored under the ground and it is virtually free (Kirschenmann 2007).

This was the invention of the modern industrial agricultural system, which at its foundation makes a fundamental change in the way we get our most basic energy. We have moved from an "ecosystem function" of energy generation to one based on fossil fuels, thus converting an agricultural system whose main purpose was to *provide* energy to human beings, to a system that is a *net consumer* of energy. Paraphrasing Richard Lewontin, it was a change from "using sun and water to grow peanuts" to "using petroleum to manufacture peanut butter." As a consequence, it has been estimated that this industrial food system expends 10-15 energy calories to produce 1 calorie of food, thereby effectively reversing the reason for the invention of agriculture in the first place. According to the FAO "…on average, farmers in industrialized countries spend five times as much commercial energy to produce one kilo of cereal as do farmers in Africa. … to produce one kilo of maize, a farmer in the US uses 33 times as much commericial energy as his or her traditional neighbour from Mexico. … to produce one kilo of rice, a farmer in the US usex 80 times the commercial enery used by a traditional farmer in the Philippines! " (GRAIN, 2007)

One of the largest contributions of energy use in industrial agriculture is a result of dependence on synthetic fertilizers. It has been estimated that as much as a third of all energy consumption in crop production in the United States is from the production of chemical fertilizers (Gellings and Parmenter, 2004). A 1992 report from the Florida cooperative extension service states that, “commercial fertilizers are by far the largest users of energy in production agriculture,” Although the authors were obviously limiting their analysis to on-farm activities, their highlighting of synthetic fertilizers as the main energy consumer is important. Similarly, according to a report of the European Fertilizer Manufacturers Association, about 60% of the energy required to produce wheat in Europe is due to the production of fertilizers. The main synthetic fertilizer worldwide is nitrogen, and the main initial product for producing it is ammonia. Ammonia is produced primarily from natural gas, with some production (27%) coming from coal.

By the 1990s fertilizer manufacturers had gained significant improvements in energy efficiency in the manufacture of fertilizers, probably reaching maximum efficiency in the early 2000s. For example the proportion of energy used in the production of corn in Ontario, Canada decreased by 42% between 1975 and 1991. As of 2008, the production of synthetic fertilizer worldwide had gained efficiency through the capture of CO_2 from manufacturing processes, and using about a third of all CO_2 produced for the production of urea. India, for example, produced 88% of its nitrogen fertilizer in the form of urea, thus reducing CO_2 emissions from the overall process of fertilizer manufacture. But, of course, this CO_2 is still released as the urea is metabolized in the soil. In a study of maize production in Canada it was found that substituting swine manure for synthetic fertilizer reduced the energy input into the system by about 32%, mainly due to the energy saved by eliminating the energy requirements of producing synthetic fertilizer (McLaughlin et al. 2000). While an accurate figure of cost of synthetic fertilizer use for agricultural production around the world is elusive, there is no doubt that it is still significant. Small scale production systems that do not use synthetic fertilizer and do not import compost from off farm sources thus save on this small but significant energy expenditure.

There continues to be considerable debate around how much the industrial agricultural system has been transformed. Does industrial agriculture really use a lot more energy than it generates, or does it generate almost as much as it uses? Answering this question in an unequivocal fashion is difficult, but a great many studies have been completed that contribute to our understanding of the energy balance in industrial agriculture. We can, as a first approximation, presume that peasant agriculture is effectively neutral, that is, the energy input is less than the energy output. Whatever energy is input in the industrial system can be regarded then as above and beyond the effectively minimal amount used by small-scale farming. Consider, as a single example, the difference between industrial maize production as opposed to an alternative, small-scale producer as still practiced by millions of farmers and similar to the energy efficiency of the past. One estimate of energy utilization is illustrated in the following figure.

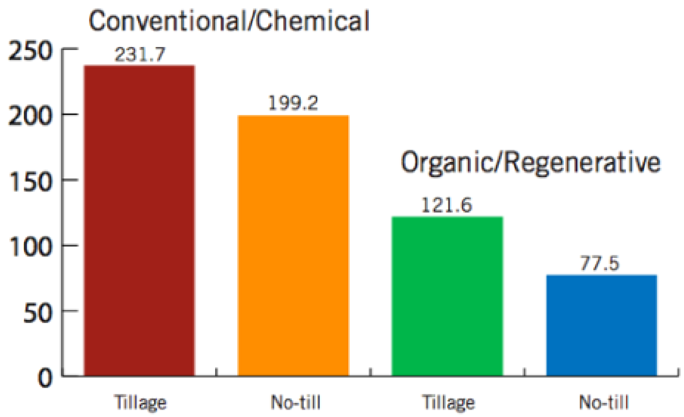

Figure 1. Comparison of industrial-style production (Conventional/Chemical) versus small-scale agroecological (Organic/Regenerative) with respect to energy used (units in ordinate are gallons of diesel per acre). Data are from Pimentel et al. (2002).

The classic study of Pimentel et al. (2002) concluded that in the case of corn and wheat, organic production in the United States was somewhere between 29 and 70% more efficient than industrial production. For the production of potatoes and apples, in contrast, the industrial system was 10-90% more energy efficient, due largely to excessive loss from insect pests under organic systems. Pimentel et al. (2002) also found that organic maize production in the United States was 30% more energy efficient than conventionally produced maize.

Several studies point to energy inputs contributing little to industrial agriculture when compared to organic methods. In a careful analysis of only the energy involved in adding synthetic nitrogen to systems of winter wheat and sugar beets in Germany, Kuesters and Lammel (1999) concluded that "… for each class of growing conditions, the highest output/input ratios [for energy] are achieved at low production intensities and decline with increasing production intensity." They go on to note, however, that the output/input ratios are only one way of measuring energy efficiency, and show that the net energy gained from using synthetic chemical nitrogen exceeds by more than five times the amount of energy embodied in the applied nitrogen. But looking at their data closely reveals that little gain in energy output is realized from any energy input from nitrogen, with the lowest levels of nitrogen always yielding significant energy in the form of biomass. Similarly, in an extensive analysis of energy efficiency in crop production in Danish agriculture, Dalgaard et al. (2001) concluded that, although the industrial model produced the most energy, the efficiency of that production was higher in organic production. They found a similar result for animal production. And, in a life cycle assessment of milk production in Sweden it was found that organic production decreased energy use compared to industrial production by 29% (Cederberg and Mattson 2000), while Denmark reported a 38% reduction in energy use by the organic system. Many other studies reach similar conclusions (e.g., Stoltze et al., 2000, Rigby and Cáceres 2001, Reganold et al. 2001).

The consensus from many studies is that overall energy use is lower in organic agriculture by a factor of approximately 20-40%. In light of the general consensus that production under organic-like systems is on average equivalent or even higher than under industrial production, we see little concern with balancing the energy saving aspects of organic production. Thus, the literature on organic versus conventional comparisons suggesting 20–40% difference in energy use can be taken as a first approximation of the expected increase in energy use when going from the small-scale agroecological production system to an industrial agriculture system.

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