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S.N. Singh

Climate Change and Crops

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In loving memory of my parents

Preface

Despite lingering uncertainties over the years, the climate change is globally viewed as an alarming threat to our sustainable development. That is why the print and electronic media are flooded with news of evils of climate change in different parts of the world. Dispelling the earlier doubts, the fourth assessment report of IPCC (2007) established a clear link between global warming and growing human activities. Although global warming is not a new phenomenon, it has been in place, since the life began on the planet Earth. Presence of greenhouse gases, like CO_2, CH_4, N_2O and water vapor etc., provided essential warmth for years for our survival on the Earth by maintaining a salubrious temperature of $+15°C$. This could be possible because these gases trap outgoing terrestrial radiations into the space and hence warm the Earth's surface.

Since the dawn of the industrial evolution of the mid-18th century, human activities have contributed significantly to rising concentrations of greenhouse gases in the atmosphere. Over the period of 1750–2000, CO_2 atmospheric concentration has increased by 31%, CH_4 by 151% and N_2O by 17%. Besides, other man-made greenhouse gases like HFCs, PFC and SF_6 were also pushed into the atmosphere to further enhance the global warming process. Continued abundance of GHGs in the Earth's atmosphere due to accelerated anthropogenic activities perturbed the thermostatic balance of Earth and pushed the surface temperature by 0.6°C in the 20th century and is predicted to further increase the temperature in the range of 2.4–6.4°C by the end of 21st century in the high scenario as per report of IPCC (2007). This may have adverse impact on the agricultural productivity globally. Agriculture is highly sensitive to climate variability and weather extremes, such as droughts, floods and severe storms. Rising tropospheric O_3 level, enhanced UV-B level due to depletion of stratospheric ozone layer, inundation of coastal areas due to thermal expansion of sea water and glacier melting are several concomitant factors which would also affect farm productivity at large. However, rising CO_2 in the atmosphere is reported to benefit crop production through enhanced photosynthesis rate in C_3 crops, provided water and nutrient supplies are not limiting.

Climate risk in the Asia/Pacific region may be ameliorated through two complementary strategies: adaptation and GHG mitigation. Adaptation options may include introduction of late maturing crop varieties or species, switching cropping sequences, sowing earlier, adjusting time of operations etc. A victim of climate change, agriculture can be also a potential source for one-fifth of anthropogenic emission of non-CO_2 greenhouse gases like CH_4 and N_2O and contributes 20% to Earth's enhanced radiative forcing. The emission of these gases can be attenuated by adopting new farming technologies. Mitigation of specially N_2O fluxes may also enhance N-use efficiency of the crop plants, leading to higher yields on one hand and on the other; it may check ground water contamination with nitrate. If such efforts are made on a global scale, they will at least postpone the danger of climate change for the time being.

The present edition was envisaged to compile the latest findings of researches carried out by the experts on the different aspects of climate change in relation to agriculture, both as source and victim. In this endeavor, I got overwhelming response of the scientists around the world and they contributed their articles enthusiastically for this volume. Hence, I feel pleasure in extending heart-felt thanks to all the authors. I would express my special gratitude to Dr. M.A.K. Khalil who not only contributed but also helped in reviewing of some chapters.

In addition, I would like to appreciate the multi-pronged services rendered by my own research scholars and staff, mainly Ms Sadhna Tiwari, Mrs. Babita Kumari, Mrs Ragini Singh, Dr. Larisha Tyagi and Sushil Kumar. Without their relentless efforts, this task could have not been completed as envisaged. Besides, Mr Dilip Chakraborty deserves special appreciation for his efforts and computer skills to put manuscript in the book format.

Lastly, I express my special thanks to my family members Mrs. Manorma Singh (wife), Mrs. Ragini Singh (daughter) and Pritish (son) for their patience, endurance and moral support during this period.

S.N. Singh

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Chapter 1
Crop Responses to Elevated Carbon Dioxide and Temperature

Mirwais M. Qaderi and David M. Reid

1.1 Introduction

Recent studies have shown that global atmospheric carbon dioxide (CO_2) has increased markedly due to human activities, including burning of fossil fuels and deforestation, and its current level (383 ppm) has far exceeded the natural range (180–300 ppm) seen over 6500 centuries (IPCC 2007). Rising of atmospheric CO_2 has caused the globally averaged surface temperatures to increase by 0.6 ± 0.2°C over the 20th century, while surface air temperature is estimated by models to warm 1.1–2.9°C "low scenario" or 2.4–6.4°C "high scenario" by the end of the 21st century relative to 1990 (IPCC 2007). A new global climate model predicts that in the coming decade, the surface air temperature is likely to exceed existing records (Smith et al. 2007). Global warming can be accompanied by shifts in precipitation patterns around the world (IPCC 2007).

Both, natural and cultivated plants are affected by different components of global climate change, including elevated CO_2 (Kimball 1983; Bazzaz 1990; Jablonski et al. 2002; Norby and Luo 2004; Ziska and Bunce 2006) and high temperature (Ferris et al. 1998; Challinor et al. 2005; Körner 2006; Morecroft and Paterson 2006). Elevated atmospheric CO_2 has positive effects on crop growth and productivity, both in terms of quantity and quality, by increasing photosynthesis and water use efficiency and decreasing transpiration through reducing stomatal conductance (Morison 1998; Long et al. 2004). Crop species are directly affected by increased atmospheric CO_2, which changes the plant physical structures and carbon: nitrogen balance (Torbert et al. 2004), and, in turn, affects growth, yield (Kimball et al. 2002a), tolerance to drought stress (Robredo et al. 2007) and susceptibility to pests and herbivores (Heagle 2003). Responses of crops to climate change are closely related to the local climate variability rather than to the global climate patterns and, therefore, crop responses to climate change vary with region and plant species (IPCC 2007). For instance, in the western Canadian province of Alberta,

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it is predicted that yields of canola (*Brassica napus* L.), corn (*Zea mays* L.) and wheat (*Triticum aestivum* L.) would increase, while in the province of Québec, yields of corn and sorghum [*Sorghum bicolor* (L.) Moench] might increase, but yields of wheat and soybean [*Glycine max* (L.) Merr.] would decrease. Similarly, in the Atlantic Provinces, yields of grain corn and soybean might increase, but no changes are expected to occur in barley (*Hordeum vulgare* L.) yield (Lemmen and Warren 2004). Also, there is an expectation that yields of temperate crops would increase under elevated CO_2 levels, but high temperature may offset this positive effect in determinate crops (Ferris et al. 1998).

A shift in growing areas of the world will probably occur due to global warming and as a result, crop productivity would increase in some areas, especially in temperate regions with extended growing season, but drastically decrease in others, such as equatorial regions (Hardy 2003). Also, some regions that currently support growth of a particular crop may in the future become unsuitable for the growth of that species because of warmer and drier conditions. In such conditions, a shift will have to be made for the production of another crop (Krupa and Groth 1999) as well as changes to human diet (Reddy et al. 1996).

The potential for crop productivity should increase with increased local average temperature over a range of 1–3°C, but decrease above this range (IPCC 2007) likely because of poor vernalization (Trnka et al. 2004), shortened phenological stages (Mitchell et al. 1993), decreased photosynthesis and increased transpiration and stomatal conductance (Nobel 2005). As indicated, increases in the frequency of both droughts and floods are projected to negatively affect local crop production (IPCC 2007), which may have a great impact on local, regional and global food supplies (Hardy 2003; Brouwer and McCarl 2006). Obviously, predictions based on the assumption that enhancement of crop productivity would be greater in drier ecosystems or in drier years have limited support (Nowak et al. 2004).

The separate and combined effects of elevated CO_2 and high temperature on plants have been studied, either in growth chambers, in greenhouses or in the field (Nijs et al. 1997; Morison and Lawlor 1999; Tuba et al. 2003; Norby and Luo 2004; Lawlor 2005). Over the past decade, there have been increasing interests in studying the effects of CO_2 on plants under field conditions, using free-air CO_2 enrichment (FACE) facilities (Kimball et al. 2002a; Nowak et al. 2004; Ainsworth and Long 2005). The main reason for using FACE has been that the results obtained from enclosure studies cannot always accurately portray the response of plants to the natural environment because of size limitation on these systems (e.g., using pots, which constrain root growth) and focus on the early stages of plant growth rather than on the whole life-cycle of plants (Long et al. 2004). However, Amthor (2001) reviewed 156 experiments on winter wheat and showed similarities in results between field and controlled-environment experiments. For example, yield increased in the controlled-environment-grown plants and field-grown plants by 12–14% and 8.0–8.6%, respectively, per 100 ppm increase in CO_2 concentration. Also, Kimball et al. (2002a) have shown that results from the earlier chamber-based studies are consistent with those from the FACE studies, and conclusions are accurate on the basis of both types of experimental approaches.

An ongoing argument exists among plant scientists who predict that the concomitant elevation of atmospheric CO_2 and air temperature will improve crop production and increase food supply for the increasing human population worldwide (Wittwer 1995) and those who are concerned about the negative impacts of high temperature, which offsets the positive effects of elevated CO_2 on crops in the future and, in turn, may lead to food shortage (Rosenzweig and Hillel 1998). The pros and cons of these two different opinions will be discussed later.

The purpose of this paper is to discuss the results of some of the experiments conducted to evaluate the separate and combined effects of elevated CO_2 and temperature on crop growth and physiology. The implications of such studies in crop productivity and food supply in the future will also be discussed. In this review, relevant data from various studies considering CO_2, temperature or other environmental factors have been used, regardless of the experimental systems, including controlled-environment growth chambers, greenhouses, open-top field chambers and free-air CO_2 enrichmment (FACE) facilities.

1.2 Effects of Elevated Carbon Dioxide on Crops

Effects of elevated atmospheric CO_2 have been examined on a number of crops (Kimball 1983; Cure and Acock 1986; Kimball et al. 2002a) in which the responses were related to other environmental factors (Fig. 1.1), including light, temperature, water, salinity and nutrients (Bowes 1993), especially nitrate and phosphate (McKee and Woodward 1994).

Plant physiological and biochemical responses (Bowes 1993) to elevated CO_2, known as the CO_2-fertilization effect (Dhakhwa et al. 1997), have been studied in plants with different photosynthetic pathways, mostly in C_3 species, but also in C_4

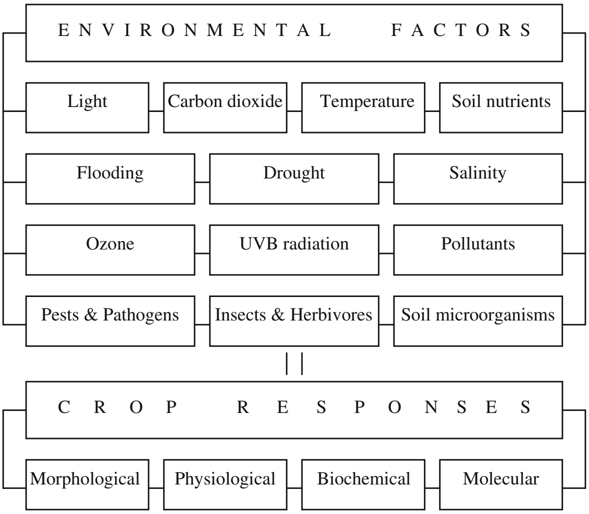

Fig. 1.1 Crop responses to CO_2, temperature and other environmental factors

and CAM plants (Bowes 1993). At present, the ambient CO_2 concentration is a limiting factor for plants with the C_3 photosynthetic pathway, and doubling of the atmospheric CO_2 will be beneficial to this group of plants, because the enzyme ribulose-1,5-bisphosphate carboxylase/oxygenase (Rubisco) can fix more CO_2 due to increased CO_2:O_2 ratio and that results in reduced photorespiration (Ziska and Bunce 2006). It has been indicated that plants can detect CO_2 concentration, but the mechanisms of such CO_2 signalling are poorly understood (Woodward 2002).

Carbon dioxide enrichment affects plant structure (Pritchard et al. 1999), transiently enhances the relative growth rate (RGR) of plants (Lambers et al. 1998) and increases biomass and yield (Kimball 1983). It alters the timing of developmental stages of plants (Bowes 1993), but accelerates growth and as a result, induces earlier leaf senescence (Heineke et al. 1999). Growing plants at elevated CO_2 concentration leads to increased leaf area, leaf area index (LAI), leaf area duration and leaf thickness as indicated by decreased specific leaf area (SLA) (Bowes 1993; Bray and Reid 2002), which is partly related to the accumulation of non-structural carbohydrates (Lambers et al. 1998).

Elevated CO_2 causes plants to produce more number of mesophyll cells and chloroplasts as well as longer stems and extended large roots with altered branching patterns (Rogers et al. 1992; Bowes 1993). High carbon gain might increase root length, diameter and number (Lee-Ho et al. 2007), and stimulate lateral root production in plants grown under elevated CO_2 (Pritchard and Rogers 2000). A shift in biomass allocation from leaves to roots can occur under CO_2 enrichment (Stulen and Den Hertog 1993). On the basis of FACE experiments, Kimball et al. (2002b) have reported that in some agricultural crops, elevated CO_2 stimulated growth of roots more than that of shoots.

Elevated CO_2 can increase the number of flowers, fruits and seeds (Bowes 1993; Jablonski et al. 2002), which results in greater individual seed mass and total seed mass, but lower seed nitrogen concentration (Jablonski et al. 2002). It also increases seed yield but decreases grain and flower protein, as shown in various wheat cultivars (Ziska et al. 2004). Seed quality of some species, grown under elevated CO_2, can be affected through changes in lipid metabolism. For example, elevated CO_2 altered wheat grain lipids (Williams et al. 1994) and doubled the number of mitochondria in wheat leaves, compared to the ambient CO_2 level (Williams et al. 1998).

Elevated CO_2 affects growth through changes in chemical composition of plants, as shown in twenty-seven C_3 species, including nine crops (Poorter et al. 1997). They reported that elevated CO_2 caused an accumulation of non-structural carbohydrates, decreased organic nitrogen compounds and minerals and increased concentration of soluble phenolic compounds in leaves. Also, CO_2 enrichment can affect nitrogen and phosphorus, which are required for the photo-oxidative and photoreductive C cycles (Rogers et al. 1999). Plants grown under elevated CO_2 have higher nitrogen use efficiency (NUE) and photosynthetic N use efficiency (PNUE) (Tuba et al. 2003).

Elevated CO_2 stimulates photosynthesis in various intensities during different phenological phases (Mitchell et al. 1999), and its direct consequence is increased dry matter production (Lawlor and Mitchell 2000; Ziska et al. 2004). There are many studies indicating that the initial stimulation of photosynthetic rates declines with exposure to elevated CO_2 (Bowes 1993; Moore et al. 1999; Stitt and Krapp 1999). Acclimation of plants to elevated CO_2 has thoroughly been described (Heineke et al. 1999; Moore et al. 1999). A decline in photosynthesis can be accompanied by a reduction in Rubisco content (Moore et al. 1999) and an adjustment in leaf carbohydrate signalling (Heineke et al. 1999; Moore et al. 1999), such as increased starch content and a decrease in nitrogen concentration (Stitt and Krapp 1999). Some of these alterations that lead to decreased photosynthesis might be caused by restriction of root growth due to limited physical rooting space in pots, although results from different studies have been inconsistent (Berntson et al. 1993). In such situations, two important regulating factors are: (1) reduced nutrient availability (McConnaughay et al. 1993), particularly nitrogen and phosphorus, and (2) sugar sensing and signalling (Rolland et al. 2002). In the latter case, under elevated CO_2, high sucrose levels can act as signals that modify the activities of sources and sinks (Taiz and Zeiger 2002) and downregulate biosynthetic activity. Reduced photosynthesis may also be related to the utilizing capacity of plants for the extra photosynthate, which is produced under CO_2 enrichment (Arp 1991; Reekie et al. 1998).

On the other hand, Garcia et al. (1998) have found little evidence of a decline in photosynthetic capacity of spring wheat under field conditions, using free-air CO_2 enrichment. In their study, photosynthesis increased significantly and substantially for the entire life of the crop. Despite the acclamatory loss of photosynthesis per unit leaf area, changes in morphological characteristics, such as greater leaf area, can increase plant biomass and yield (Bowes 1993).

Elevated CO_2 reduces transpiration by partially closing the stomata and decreasing stomatal conductance (Morison and Gifford 1983; Bunce 2000), which decreases the ability of plants to dissipate heat load through nonphotochemical mechanisms under extreme temperature events (Shaw et al. 2005). On the other hand, sub-ambient CO_2 level stimulates stomatal opening or inhibits stomatal closure (Assmann 1999). Reduced stomatal opening leads to improved water use efficiency (Guy and Reid 1986; Clifford et al. 2000) and as a result, lowers water stress in plants (Kimball 1983). Wilson and Bunce (1997) have reported that both leaf temperature and leaf-to-air vapour pressure difference played a role in the reduction of stomatal conductance in soybean. Improved water status of plants, due to partial closure of stomata, causes a higher turgor pressure, which stimulates leaf expansion (Lenssen and Rozema 1990). Also, plant water use efficiency is strongly affected by stomatal density (Woodward and Kelly 1995). Both stomatal density and stomatal index of leaves, which are negatively correlated with elevated CO_2, have decreased over the past 100 years (Woodward 1987).

Decreased transpiration at elevated CO_2 with no increase in leaf water potential in wet soil indicates decreased hydraulic conductance. In some crops (e.g., corn and soybean), grown at elevated CO_2, both reversible and irreversible decreases occurred in hydraulic conductance, which could have been related to decreased transpiration (Bunce and Ziska 1998). Robredo et al. (2007) have also shown that hydraulic conductance decreased markedly in barley plants grown under elevated CO_2 than those grown under the ambient CO_2 level.

In response to elevated CO_2, the rate of respiration is increased in some species, but decreased in others (Poorter et al. 1997). As shown previously, elevated atmospheric CO_2 leads to a reduction in mitochondrial respiration and doubling of the current CO_2 level will reduce the respiration rate by 15–18% per unit dry weight (Drake et al. 1999).

Elevated CO_2 can affect plant hormones and its promoting effect on ethylene production through increasing the amount of the enzyme 1-amino-cyclopropane-1-carboxylate (ACC) oxidase has been shown in different plant species (Sisler and Wood 1988; Smith and John 1993), including sunflower, *Helianthus annuus* L. (Dhawan et al. 1981; Finlayson and Reid 1994). Woodrow and Grodzinski (1993) have reported that prolonged growth under elevated CO_2 increased endogenous ethylene production in the leaves of tomato (*Lycopersicon esculentum* L.) plants, compared to the leaves of those plants grown under lower CO_2 levels.

Ethylene can affect many aspects of plant growth and development (Khan 2006), including leaf orientation and carbon partitioning, as shown in tomato (Woodrow et al. 1988). It also influences root growth of sunflower, which is affected by CO_2 concentration (Finlayson and Reid 1996). From their study on rice (*Oryza sativa* L.), grown under elevated CO_2, Seneweera et al. (2003) concluded that increased ethylene production is central in promoting accelerated development, which coincides with enhanced rates of tiller appearance and the release of auxiliary buds leading to increased grain yield under CO_2 enrichment.

Overall, crops will benefit from doubling of the atmospheric CO_2, which affects various aspects of plant metabolism by acting as activator and substrate for Rubisco, promoting stomatal closure, and influencing respiration and hormone levels (Bowes 1993). All of these alterations can lead to increased crop growth and yield. However, one must not forget the confounding effects of higher temperatures and other factors, such as water and mineral nutrient supply (Fig. 1.1).

1.3 Effects of High Temperature on Crops

Elevated air temperature accelerates various aspects of plant metabolism (Larcher 2003), and can affect crops through changing the timing and rate of physiological processes, including the rate of organ development, respiration, senescence and their source-sink relations (Farrar and Williams 1991). Plant development can be accelerated by high temperatures, which lead to earlier shifts to the next ontogenetic stage (Badeck et al. 2004).

Crop responses to temperature depend on the specific optimum temperature for photosynthesis, growth and yield (Conroy et al. 1994). If the temperature is below optimum for photosynthesis, a slight increase in temperature may lead to increased plant growth and development, but if the temperature is close to maximum, a small increase in temperature can negatively affect crop growth and in turn, decrease yield (Baker and Allen 1993). Also, the character of temperature increase and the developmental stage of the crop determine crop responses to higher temperatures (Porter and Gawith 1999).

High temperatures reduce plant biomass and adversely affect crop reproductive efforts (Polowick and Sawhney 1988; Ferris et al. 1998), particularly flowering and fertilization (Morison and Lawlor 1999). As shown in a number of species, including bean (*Phaseolus vulgaris* L.), linseed (*Linum usitatissimum* L.), tomato, corn, wheat and canola, high temperature stress reduced seed yields (Polowick and Sawhney 1988; Young et al. 2004). Also, *Brassica* species that have vigorous growth between 10° and 30°C, with the optimum temperature of ~20°C, are very sensitive to high temperatures at the blooming time, even with ample moisture availability. Extended periods of high temperature (over 30°C) can result in severe sterility and high yield losses as well as in low oil content and poor seed quality if it happens during the period of seed filling (Sovero 1993). Young et al. (2004) showed that, in canola, high temperature reduced pollen viability and germinability, induced flower and fruit abortion and as a result, negatively affected seed production. In annual crops, such as peanut (*Arachis hypogaea* L.) (Challinor et al. 2005) and wheat (Ferris et al. 1998; Wheeler et al. 2000), even a brief period of high temperature near to the flowering stage may lead to severe yield reduction due to pollen sterilization. Mitchell et al. (1993) indicated that reduced yield in wheat could be due to shortened phenological stages and in turn, less time for the accumulation of resources for grain formation. High temperature can also affect yield quality. As reported in cereal grains, high temperature reduced protein content and altered types and proportions of proteins and lipids (Lawlor 2005).

High temperatures decrease plant biomass and yield by decreasing photosynthesis and increasing transpiration and stomatal conductance (Nobel 2005). A sharp increase in the basal chlorophyll fluorescence indicates a blockage of the photochemical reaction centre of photosystem II under high temperature that affect photosynthesis (Hällgren et al. 1991). Plants tend to produce smaller leaves and more extensive root systems under higher temperature to offset leaf water loss and to increase water intake (Gliessman 1998). Also, plants mitigate overheating by leaf rolling and drooping and vertical leaf orientation (Larcher 2003; Nobel 2005) or by transient wilting (Chiariello et al. 1987; Nobel 2005). Such adaptive mechanisms likely reduce leaf exposure to incident light and in turn, may lead to decreased photosynthesis. Reduced net photosynthesis due to increased temperature can in part be related to Rubisco deactivation (Crafts-Brandner and Salvucci 2000). In contrast to elevated CO_2, high temperatures lead to increased photorespiration by decreasing the affinity of the enzyme Rubisco to CO_2 relative to oxygen (Lawlor 1998).

Plant hormones, such as abscisic acid (ABA), indole-3-acetic acid (IAA) and ethylene, are involved in the responses of plants to high temperature (Nilsen and Orcutt 1996) by balancing transpiration through affecting stomatal conductance (Dodd and Davies 2004). High temperatures increase ABA, but decrease IAA (Nilsen and Orcutt 1996) and ethylene evolution (Yu et al. 1980). IAA is a growth promoting hormone and its lower concentration under higher temperature can, in canola, negatively affect plant growth and that results in lower dry matter (Qaderi et al. 2006).

1.4 Interactive Effects of Elevated CO_2 and Temperature on Crops

Concerns about the enrichment of atmospheric CO_2 and its association with global warming that will likely have potential impacts on crop production, have drawn attention of plant scientists to study the combined effects of elevated CO_2 and high temperature on plants, especially crops (Idso et al. 1987; Rawson 1992; Morison and Lawlor 1999). As mentioned earlier, crops respond positively to elevated CO_2 and negatively to high temperature, but what about the interactive effects of these two environmental factors? Previous studies have shown that elevated CO_2 can partially ameliorate some of the adverse effects of environmental stresses, including high temperature (Qaderi et al. 2006), salinity (Bray and Reid 2002) and ultraviolet-B radiation (Qaderi and Reid 2005) on crops. Even though elevated CO_2 can mitigate the detrimental effects of the above-optimal temperatures on crop growth and yield, certainly temperatures near the upper limit for crops will negatively affect yields, regardless of CO_2 concentration (Polley 2002).

Crop responses to elevated CO_2 are strongly related to temperature (Wheeler et al. 1994). Both of these environmental factors affect crop yields, in terms of seed quantity and quality. In a meta-analysis of FACE experiments, Ainsworth and Long (2005) have shown that the herbaceous crop yield increased on average by 17% and the above-ground dry matter production for trees increased 28%. In contrast, Lobell et al. (2006) developed statistical crop models, using outputs from multiple climate models and considering six California perennial food crops, and predicted that temperature-induced decreases in yields of these crops will likely occur by 2050. Also, Prasad et al. (2005) have reported that elevated CO_2 can increase yields of grain legume crops (e.g., soybean, dry bean (*Phaseolus vulgaris* L.) peanut, and cowpea [*Vigna unguiculata* (L.) Walp.)], but this beneficial effect is offset by negative effects of the above-optimum temperature, which leads to decreased seed yield and quality. Thomas et al. (2003) have shown that, in soybean grown under five temperature regimes and two CO_2 concentrations, carbon dioxide had no effect on seed composition and transcript, whereas temperature had a pronounced effect on mature seed composition (e.g., total oil, ratio of fatty acids, nitrogen, phosphorus, carbohydrate) and transcript (e.g., β-glucosidase) in developing seeds (Table 1.1).

Vu et al. (1997) have found that elevated CO_2 enhanced leaf photosynthetic CO_2 assimilation rates, but CO_2 enrichment and high temperature reduced Rubisco content in rice and soybean. Variation in the responses of these two crop species to these two environmental factors suggests an interspecific variation among C_3 plants in response to future global climate change. Ziska et al. (1996) studied the response of 17 rice cultivars to increased CO_2 and temperature and reported that at elevated CO_2, plant growth and yield were higher under lower temperature than under higher temperature regime and there was variation among rice cultivars.

Alterations in the air temperature and its interaction with elevated CO_2 can affect photosynthesis. Based on the biochemical models of photosynthesis (Farquhar et al. 1980), other models that simulate this interaction have been

Table 1.1 Responses of some crop species to the stimulatory effects of elevated CO_2 at high temperature

Crop	Environment*	References
Strong response		
Bean	CEC	Cowling and Sage (1998)
Peanut	GH	Vu (2005)
Rice	FACE	Borjigidai et al. (2006)
Soybean	GC	Sionit et al. (1987)
Sweet potato	GH	Cen and Sage (2005)
Wheat	OTC	Hakala (1998)
Weak response		
Cotton	EGC-SPAR	Reddy et al. (2005)
Cowpea	GC	Ahmed et al. (1993)
Bean	Sunlit CEC	Prasad et al. (2002)
Peanut	EGC-SPAR	Prasad et al. (2005)
Rice	GH	Ziska et al. (1996)
Soybean	Sunlit CEC	Thomas et al. (2003)
Wheat	C-GH	Mitchell et al. (1993)

* C, chamber; CEC, controlled-environment chamber; GC, growth chamber; FACE, free-air CO_2 enrichment; GH, greenhouse; OTC, open-top chamber; SPAR, soil-plant-atmosphere-research unit.

developed (Kirschbaum 1994; Mitchell et al. 1995). Long (1991) postulated that elevated atmospheric CO_2 concentration at higher temperatures would increase net CO_2 assimilation and carbon pools, which facilitate dissipation of excess light energy resulting in decreased photoinhibition and thus leads to increased plant productivity. As shown in Table 1.1, a positive interaction between elevated CO_2 and high temperature on photosynthesis of C_3 plants has been reported by some investigators (Hakala 1998; Vu 2005; Borjigidai et al. 2006), but not by others (Mitchell et al. 1993; Prasad et al. 2005; Reddy et al. 2005).

Sionit et al. (1987) reported that under CO_2 doubling, soybean yield was greater for plants grown under 26/20°C than for those grown under 22/16°C or 18/12°C of day/night temperature regimes. However, Prasad et al. (2002) showed that in kidney bean (*Phaseolus vulgaris* L.), elevated CO_2 did not offset the detrimental effects of high temperatures on reproductive processes and yield. Also, no direct beneficial effects of elevated CO_2 at high temperature was found in both heat-sensitive and heat-tolerant lines of cowpea (Ahmed et al. 1993). As stated by Reddy et al. (1996), CO_2 enrichment and increased water and nutrients do not mitigate the adverse effects of the above-optimum temperatures on food and fiber production.

In response to elevated CO_2 and high temperature, some researchers working on the same crop species have reported contradictory results (Table 1.1). These differences might be related to differences in experimental approaches, stage of crop development as well as other environmental factors, which probably have not been taken into consideration during the experiments.

Although elevation of atmospheric CO_2 and air temperature are important components of global climate change, there are other factors, which are correlated with these two factors (Fig. 1.1). For instance, global warming is likely to affect precipitation patterns around the world in such a way that there will be severe drought condition in one region and flooding in another region (IPCC 2007). As an example, we consider the effects of three components of global climate change, such as CO_2, temperature and drought, on canola growth and physiology (Table 1.2).

When drought was not considered as a factor, no differences were found in growth between plants grown under higher temperature at elevated CO_2 and plants grown under lower temperature at ambient CO_2, because elevated CO_2 mitigated the adverse effects of high temperature in the first situation. However, drought stress plus elevated CO_2 and higher temperature decreased growth, as did higher temperature at ambient CO_2 concentration. Also, regardless of CO_2 level, drought stress increased abscisic acid content, but higher temperature inhibited the ability of plants to produce abscisic acid in response to drought (Table 1.3). On the basis of this

Table 1.2 Growth and physiological responses of canola (*Brassica napus*) plants to elevated CO_2 (eCO_2), higher temperature (HT) and the combined effects of eCO_2 and HT, and eCO_2, HT and drought stress (DS), compared to the well-watered plants grown under lower temperature at ambient CO_2 level

Crop response	eCO_2	HT	eCO_2 and HT	eCO_2, HT and DS
Growth				
Height	I*	D	NC	D
Diameter	I	D	NC	D
Leaf number	NC	NC	NC	NC
Leaf area	I	D	NC	D
Growth index				
SLW	I	NC	I	I
LWR	NC	I	I	I
LAR	D	NC	D	D
SRR	NC	NC	NC	D
Dry matter production				
Leaf	I	NC	I	NC
Stem	I	D	I	D
Root	I	NC	NC	NC
Total	I	NC	I	NC
Gas exchange				
CO_2 assimilation	I	NC	NC	NC
Transpiration	D	NC	D	D
Stomatal conductance	D	NC	D	D
Water use efficiency	I	NC	I	I
Chlorophyll fluorescence				
Fv/Fm	NC	NC	NC	D
Y	NC	D	NC	D
Photosynthetic pigment				
Chl *a*	NC	NC	NC	I
Chl *b*	NC	NC	I	I
Carotenoids	D	NC	NC	I

* I, increased; D, decreased; NC, no change.

Adapted from Qaderi et al. (2006).

Table 1.3 Effects of two temperature regimes and two CO_2 levels on abscisic acid concentration (ng g^{-1} dry weight) in well-watered and drought-stressed canola (*Brassica napus*) plants

		Watering regime	
Temperature regime	CO_2 level	Well-watered	Drought-stressed
Lower	Ambient	106.7 ± 23.9 c	1574.7 ± 76.0 a
	Elevated	130.0 ± 15.3 c	1063.0 ± 99.7 b
Higher	Ambient	162.2 ± 78.3 c	71.2 ± 8.0 c
	Elevated	107.7 ± 5.2 c	175.6 ± 24.3 c

Adapted from Qaderi et al. (2006).

study, we could suggest that crop growth models, which only consider elevated CO_2 and high temperature, simply cannot answer the very complicated changes that may occur in global climate which would affect crops in the future (Qaderi et al. 2006).

Climate change is also likely to increase insect infestations and disease outbreaks and to alter the composition of soil nutrients (Lemmen and Warren 2004; Lawlor 2005). Many other abiotic and biotic environmental factors (Fig. 1.1), including ultraviolet-B radiation, ozone concentration, relative humidity and pollutants as well as the composition and activity of soil microorganisms and numerous others, will influence crop responses to elevated CO_2 and temperature (Bazzaz and Sombroek 1996). Thus, some of these factors must be included in the crop growth models.

Most studies have focused on the morphological, physiological and biochemical aspects of crop responses to global climate change (Bazzaz and Sombroek 1996). However, in recent years, there have been increasing interests in studying the molecular aspects of crop responses to climate change (Hoogenboom et al. 2004; White et al. 2004; Taylor et al. 2006).

1.5 Conclusion

One of the questions arising from experiments dealing with CO_2 and temperature is whether crop responses are strongly positive or weakly positive to the combined effects of elevated CO_2 and high temperature. Due to temporal and spatial differences in crop responses to elevated CO_2 and high temperature, it is currently impossible to clearly state the stimulatory effects of elevated CO_2 at high temperature. Therefore, it seems appropriate that we suggest a possible range for crop performance under these environmental conditions (Fig. 1.2).

In order to investigate in-depth the effects of climate change on crops, future studies should consider plant performance under simulated future climate, using all available techniques and tools, including ecological, physiological, biochemical and molecular, in large scale multi-factorial experiments.

It has been predicted that global warming leads to many changes in the environment that may affect the interactive effects of CO_2 and temperature (Tables 1.2 and 1.3). Hence, it is better to design experiments, which include other important factors

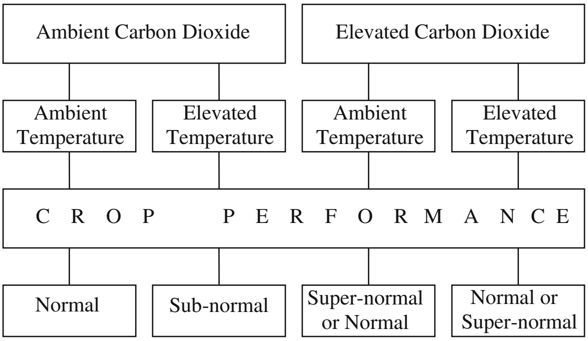

Fig. 1.2 Crop responses to the combined effects of CO_2 and temperature

besides CO_2 and temperature. Such experiments can be conducted in the field, using free-air CO_2 enrichment (FACE) and free-air temperature increase (FATI) (Nijs et al. 1997), which closely represent future high-CO_2 and high-temperature world. Results from the multi-factorial experiments should allow plant biologists to create models, which can predict the effects of global climate change on crops more realistically.

Relatively recent studies that demonstrate the protective aspects of biogenic volatile organic compounds (BVOCs) from plants against global warming are encouraging. For example, Peñuelas and Llusià (2003) have suggested that some of the assimilated carbon that is returned to the atmosphere as BVOCs could protect plants from the negative effects of high temperature.

Although crop yields will inevitably be influenced by climate change in the future, it is possible that through enhanced technology, improved agronomic and management practices, and development and use of stress-tolerant varieties, crop production may remain stable and retain its sustainability under conditions of increased atmospheric CO_2, despite the adverse effects of global warming.

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Chapter 2

Climate Change, Climate Variability and Indian Agriculture: Impacts Vulnerability and Adaptation Strategies

Shakeel A. Khan, Sanjeev Kumar, M.Z. Hussain and N. Kalra

2.1 Introduction

Climate is changing naturally at its own pace, since the beginning of the evolution of earth, 4–5 billion years ago, but presently, it has gained momentum due to inadvertent anthropogenic disturbances. These changes may culminate in adverse impact on human health and the biosphere on which we depend. The multi-faceted interactions among the humans, microbes and the rest of the biosphere, have started reflecting an increase in the concentration of greenhouse gases (GHGs) i.e. CO_2, CH_4 and N_2O, causing warming across the globe along with other cascading consequences in the form of shift in rainfall pattern, melting of ice, rise in sea level etc. The above multifarious interactions among atmospheric composition, climate change and human, plant and animal health need to be scrutinized and probable solutions to the undesirable changes may be sought.

Vulnerability is the degree to which a system is susceptible to, or unable to cope with adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the character, magnitude, and rate of climate change and variation to which a system is exposed as well as the system's sensitivity and adaptive capacity. Vulnerability to climate change varies across regions, sectors, and social groups. Understanding the regional and local dimensions of vulnerability is essential to develop appropriate and targeted adaptation efforts. At the same time, such efforts must recognise that climate change impacts will not be felt in isolation, but in the context of multiple stresses. In particular, the dramatic economic and social changes associated with globalisation themselves present new risks as well as opportunities.

Research on the impact of climate change and vulnerability on agriculture is a high priority in India as the impact, if it follows the predictions, is expected to be widespread and severe. Developing the ability to confidently estimate the impacts

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of climate change on agriculture is critically important. If ever achieved, it could provide the global information needed to help farmers develop their own long-range response to climate change. Fortunately, we are very near to having such a capability, and it may take 5–7 years to substantially improve the resolution and accuracy of the climate model and evaluate the implications for agriculture.

Changes in the atmospheric chemistry have increased during last few decades due to the heightened anthropogenic activities. Global negotiation have been under way for sometime to reduce the emissions of greenhouses gases to 1990 level, but the success of these endeavors are less certain today due to increasing reluctance of the major contributors to change. Considering the business as usual scenario, CO_2 is projected to increase at the rate of 1.8 ppm per year, reaching 397–416 ppm by 2010 and 605–755 by 2070 (Watson et al. 1998). This along with changes in other greenhouses gases is likely to result in a temperature increase of earth's surface and atmosphere.

2.2 Indian Agriculture and Climate Change

Agriculture sector alone represents 23 per cent of India's Gross National Product (GNP), plays a crucial role in the country's development and shall continue to occupy an important place in the national economy. It sustains the livelihood of nearly 70% of the population. It seems obvious that any significant change in climate on a global scale will impact local agriculture, and therefore affect the world's food supply. Considerable studies have been carried out to investigate how farming might be affected in the different regions. Several uncertainties limit the accuracy of current projections. One relates to the degree of temperature increase and its geographic distribution. Another pertains to the concomitant changes likely to occur in the precipitation patterns that determine the water supply to the crops, and the evaporative demand imposed on the crops in carbon dioxide enriched atmosphere. The problems of predicting the future course of agriculture in the changing world are compounded by the fundamental complexity of natural agricultural systems, and socio-economic systems governing the world food supply and demand. Many climatologists predict a significant global warning in the coming decades due to rising atmospheric carbon dioxide and other green house gases. As a consequence, major changes in the hydrological regimes have been also forecast to occur. Changes in the temperature, solar radiation, and precipitation will have an effect on crop productivity and livestock agriculture. Climate change will also have an economic impact on agriculture, including changes in farm profitability, prices, supply, demand, trade and regional comparative advantages. The magnitude and geographical distribution of such climate induced changes may affect our ability to expand the food production area as required to feed the burgeoning population of more than 10,000 million people projected for the middle of the next century.

Agriculture is sensitive to short-term changes in weather and to seasonal, annual and longer-term variations in climate. For the long-term changes, agriculture is able to tolerate moderate variations in the climatic mean. Changes beyond these bands of tolerance may require shifts in cultivars and crops, new technologies and infrastructure or ultimately conversion to different land uses. Crop yield is the culmination of a diversified range of factors. The variations in the meteorological parameters are more of transitory in nature and have paramount influence on the agricultural systems, although other parameters, like soil characteristic, seed genetics, pest and disease and agronomic practices also do impact crop yields. Among these factors, pest and diseases cause a significant loss to world food production under different climatic conditions. Development and distribution of pest and diseases are governed by temperature patterns, rainfall or humidity and seasonal length to a great extent. Especially, winter temperatures are important for the survival of pest and studies have shown that increase in temperature accelerates the development of pests in general. Pest-crop interaction will be also directly affected by the rising CO_2 levels through the alteration of host plant attributes, such as C/N ratios and secondary plant nutrient chemistry. In terms of crop production, these fluctuations must be taken into the account while planning agricultural operations. The climate elements which affect the plant growth and development, hence the agriculture as a whole, are carbon dioxide concentration, temperature, radiation, precipitation and humidity.

Analysis of the food grains production/productivity data for the last few decades reveals a tremendous increase in yield, but it appears that negative impact of vagaries of monsoon has been large throughout the period. In this context, a number of questions need to be addressed as to determine the nature of variability of important weather events, particularly the rainfall received in a season/year as well its distribution within the season. These observations need to be coupled to management practices, which are tailored to the climate variability of the region, such as optimal time of sowing, level of pesticides and fertilizer application.

The mean temperature in India is projected to increase by 0.1–0.3°C in kharif and 0.3–0.7°C during rabi by 2010 and by 0.4–2.0°C during kharif and to 1.1–4.5°C in rabi by 2070. Similarly, mean rainfall is projected not to change by 2010, but to increase by up to 10% during kharif and rabi by 2070. At the same time, there is an increased possibility of climate extremes, such as the timing of onset of monsoon, intensities and frequencies of drought and floods.

The rise in the concentration of green houses gases was caused primarily by human and industrial activities. The increased agricultural activities and organic waste management are presumed to be contributing to the building up of both methane and nitrous oxide in the atmosphere. However, agriculture in general and Indian agriculture in particular is not contributing significantly to global climatic change, as GHG emissions from agriculture indicate. India's total contribution to global methane emission from all sources is only 18.5 Tg per year. Agriculture (largely rice paddies and ruminant animal production) is a major source of CH_4 emission and contributes 68% to it. The continuously flooded rice fields emit methane, because anoxic conditions favor methanogenesis. Since India and China are the major rice producing countries, US-EPA attributed 37.8 Tg Methane/year to the Indian rice paddies. Based on this estimate, an international opinion was made that Asia and in particular, India and China are contributing significantly to global warming and they should do something to prevent this phenomenon. Sinha et al. (1998) estimated that global annual methane emission from rice cultivation is less than 13 Tg. IPCC (1996) has now revised the estimates of global methane emission from rice to 60 Tg/year. These estimates are still very high and can be further brought down. The contribution of Indian paddies to global CH_4 budget was estimated to be only 4.2 Tg/year (Bhattacharya and Mitra 1998). The main reasons of low methane emissions from rice fields in India are that the soils of major rice growing areas have very low organic carbon are also and not continuously flooded.

Atmospheric concentration of N_2O is increasing at a rate of $0.22\pm0.02\%$ per year (Machida et al. 1995; Battle et al. 1996; Mosier et al. 1998). The emission of N_2O is of serious concern, because of its long atmospheric lifetime of 166 ± 16 years (Prinn et al. 1990). But despite its lower concentration and less rapid rise, N_2O is becoming an important GHG, because of its longer lifetime and greater global warming potential than CO_2 (300 times more than that of CO_2 molecule). About 5% of total greenhouse effect can be ascribed to N_2O and it is also responsible for the destruction of stratospheric ozone (Rodhe 1990). Estimates of total nitrous oxides from Indian agriculture are very low due to low soil fertility and lower amounts of fertilizers used in agriculture as compared to the western countries. In India, CO_2 fixation becomes more important, because we use almost 190 million hectare of land for farming. The estimated dry biomass production from agriculture in India is almost 800 million tons every year. This is equivalent to the fixation of 320 Tg of C or 1000 Tg of CO_2 per annum. Only a part is retained over time due to low body weight of human beings and other consumers and the rest is released to the atmosphere.

2.3 The Impact of Climate Change and Climatic Variability on Agriculture Productivity

2.3.1 Crop Productivity

Increase in atmospheric carbon dioxide has a fertilization effect on crops with C_3 photosynthetic pathway and thus, promotes their growth and productivity. On the other hand, an increase in temperature, depending upon the current ambient temperature, can reduce crop duration, increase crop respiration, alter photosynthate partitioning to economic products, effect the survival and distributions of pest populations thus developing new equilibrium between crops and pests, hasten nutrient mineralisation in soils, decrease fertilizer use efficiency, and increase evapotranspiration. Indirectly, there may be considerable effects on land use pattern due to availability of irrigation water, frequency and intensity of inter- and intra-seasonal droughts and floods, and availability of energy. All of these can have tremendous impact on agricultural production and hence, food security of any region.

Wheat growth simulator (WTGROWS), developed at IARI, New Delhi, has been extensively tested for different agro-environments (Aggarwal and Kalra 1994). In past, it has been successfully used for the resource management, forecasting of wheat yields and climate variability related studies. Using WTGROWS, a strong linear decline in wheat yield was noticed with the increase in January temperature. For every degree increase in mean temperature, grain yield decreased by 428 kg/ha. Inter-seasonal climatic variability analysis carried out through yield response of wheat indicated that impact of the variability was lowest for Kota and highest for Solapur. Inter-seasonal climatic variability has been characterized through growth and yield response under different production environments, which clearly indicate the use of crop model as an indicator of climatic variability/change.

The change in rice yields at improved level of management with change in temperature and CO_2 is plotted in Fig. 2.1. Increase of 1°C temperature without any increase in CO_2 resulted in 5, 8, 5 and 7% decrease in grain yield in north, west, east and southern regions, respectively. Increase of 2°C temperature resulted in 10–16% reduction in yield in different regions, while a 4°C rise led to 21–30% reduction. Sinha and Swaminathan (1991) reported that a 2°C increase in mean air

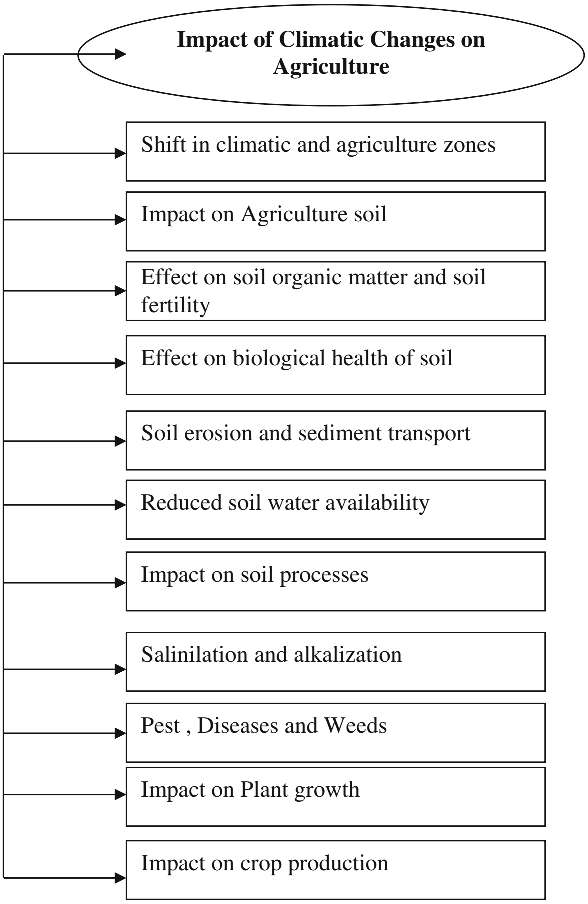

Fig. 2.1 Points that must be considered while doing the study of impacts of climatic changes on agriculture

temperature could decrease rice yield by about 0.75 t/ha in the high yield areas and by about 0.06 t/ha in the low yield coastal regions. Further, a 0.5°C increase in winter temperature would reduce wheat crop duration by seven days and reduce yield by 0.45 t/ha. An increase in winter temperature of 0.5°C would thereby translate into a 10% reduction in wheat production in the high yield states of Punjab, Haryana and Uttar Pradesh. The reduction was lower in eastern India compared to all other regions (Fig. 2.1). Mean grain yields of control crops in eastern region were 7.9 t/ha as compared to 8.7–9.9 t/ha in other regions. This was because of relatively higher temperatures in east (32.2/25.3°C) both during grain formation and filling phase, accompanied by lower radiation. As a result, these crops had fewer grains and shorter grain filling duration. Although temperatures were high in northern India as well (33.8/25.0°C), the region also had more radiation, which resulted in higher grain yields.

The impact of interactions between carbon dioxide and temperature can also be seen in Fig. 2.2. At 350 ppm in north India, there was a change of −5, −12, −21,

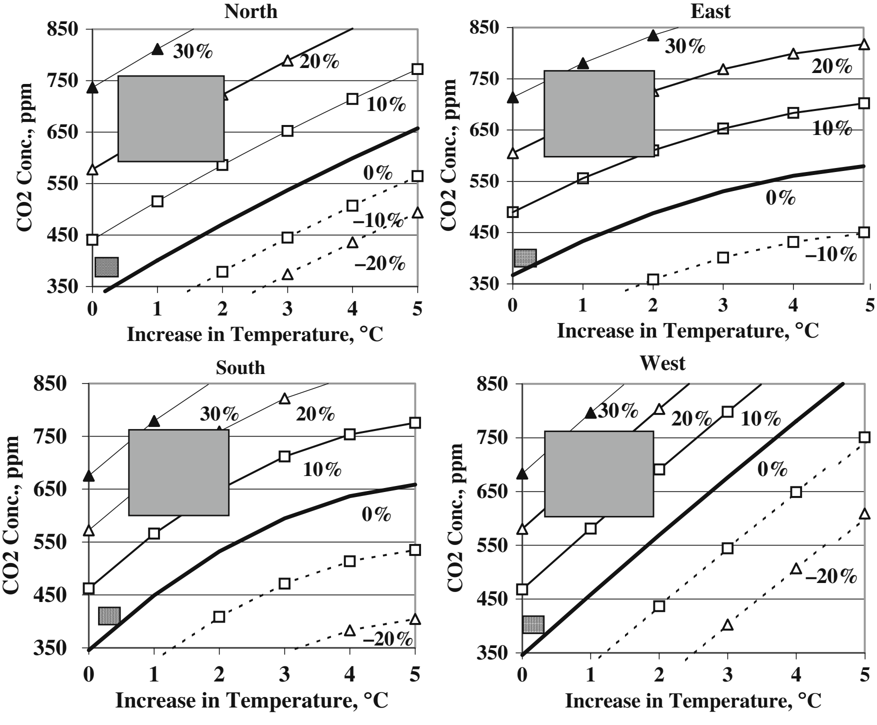

Fig. 2.2 Effect of increase in temperature and CO_2 on simulated grain yields of irrigated rice with improved N management (allowing no N stress) in different regions of India. Lines refer to the equal change in grain yield (% change, labeled) at different values of CO_2 and increase in temperature. Large, shaded box refers to bias in impact assessment due to uncertainties in IPCC scenario of 2070 and the small, hatched box refers to the bias due to uncertainties in the scenario of 2010

−25 and −31% in grain yield with increase of 1, 2, 3, 4, 5°C temperatures, respectively. In the same region, and at the same temperatures but at 550 ppm, these yield changes were 12, 7, 1, −5 and −11%, respectively. Similar interaction could be noted for other regions as well. Thus, in eastern and northern regions, the beneficial effect of 450, 550 and 650 ppm CO_2 was nullified by an increase of 1.2–1.7, 3.2–3.5 and 4.8–5.0°C, respectively (Table 2.1). In southern and western regions, positive CO_2 effects were nullified at temperatures lower than these. It can be concluded that in improved management conditions, the regions, such as southern and western parts of India, which currently have relatively lower temperatures, are likely to show less increase in rice yields under climate change compared to northern and eastern regions.

The effect of sowing at different dates on the yield of wheat was simulated for different locations. Areas having higher potential yields of wheat had greater reduction in yield per day delay in sowing from the optimal date. A few locations (north eastern parts) showed a small yield reduction with delayed sowing. With the temperature rise, the adjustments in the date of sowing to have the similar weather conditions can be ensured, but it can imbalance the cropping system schedule, which is also important for developing countries where intensive cultivation is practiced on small and marginal lands.

Aggarwal and Kalra (1994), by using WTGROWS, demonstrated the shift of iso-yield lines of wheat in India with 425 ppm of CO_2 concentration and 2°C rise in temperature. The rise in carbon dioxide concentration of the atmosphere effectively influences the productivity of crop plants. For these studies, open top chamber facilities have been developed at various Institutes in India. In these chambers, the coupled weather and canopy environment also change along with CO_2, and thus the differences in growth and yield of crops become complex function of these parameters. To work out the impact of carbon dioxide and temperature only, Free Air Carbon Dioxide Enrichment (FACE) facilities have been established at the Indian Agricultural Research Institute, New Delhi in collaboration with National Physical

Table 2.1 Temperature increases (°C) that will cancel out the positive effect of CO_2 in different regions at two levels of management

	CO_2 concentration		
Management	450 ppm	550 ppm	650 ppm
North India			
Improved	1.7	3.2	>5.0
Current	1.9	2.7	4.8
East India			
Improved	1.2	3.5	>5.0
Current	2.0	4.4	>5.0
West India			
Improved	0.9	1.8	2.8
Current	1.0	2.1	3.4
South India			
Improved	1.0	2.3	4.4
Current	0.9	2.0	3.4

Laboratory. Some of the research findings with these facilities are indicated in this section. Enhanced carbon dioxide concentration effects the carbon dioxide assimilation and their partitioning within the source leaf and transport to the sink in mungbean and wheat. Carbon dioxide elevation partially compensates for the negative effect of moisture stress in *Brassica* plants and may possibly help to grow in the drier habitat than they are currently grown. *Brassica* spp. responded differently to elevated carbon dioxide levels. All the yield components in rice viz. panicle number (effective tillers), filled grains per panicle and grain weight responded positively to enhanced carbon dioxide levels. Increased photo-assimilate supply possibly increased the maturity percentage of the seeds. The studies with rice, wheat and mustard are currently being used to refine the existing crop growth models. Presently, the FACE facility has been used to study crop response to CO_2 on growth and yield. The efforts are being made to include also the rising temperature effects (Uprety 1998). This kind of study needs to be also extended to some of the traditional crops of the country, like chickpea, pigeonpea, groundnut and potato.

Gadgil (1995) and Gadgil et al. (1999a, b) used PNUTGRO model to determine the sowing window for rainfed groundnut. Variation in the model yield with sowing date showed that broad sowing window of 22nd June–17th August is the optimum for minimizing the risk of failure. It was also shown that incidence of locally triggered pests/diseases viz. *leaf miner* and late leaf spot (t*ikka*) is low when sowing is postponed to after mid-July and thus does not involve much risk. It was also seen that pod filling stage was critical for moisture availability.

Lal et al. (1999) projected 50% increased yield for soybean for a doubling of CO_2 in Central India. However, a 3°C rise in surface air temperature almost cancels out the positive effects of doubling of carbon dioxide concentration. A decline in daily rainfall amount by 10% restricts the grain yield to about 32%.

Hundal and Kaur (1996) examined the climate change impact on productivity of wheat, rice, maize and groundnut crop in Punjab. If all other climate variables were to remain constant, temperature increase of 1, 2 and 3°C from present day condition, would reduce the grain yield of wheat by 8.1, 18.7 and 25.7%, rice by 5.4, 7.4 and 25.1%, maize by 10.4, 14.6 and 21.4% and seed yield in groundnut by 8.7, 23.2 and 36.2%, respectively.

Lal et al. (1998) examined the vulnerability of wheat and rice crops in northwest India to climate change through sensitivity experiments with CERES model and found that under elevated CO_2 levels, yields of rice and wheat increased significantly (15 and 28% for a doubling of CO_2). However, a 3°C (2°C) rise in temperature cancelled out the positive effect of elevated CO_2 on wheat (rice). The combined effect of enhanced CO_2 and imposed thermal stress on the wheat (rice) crop is 21% (4%) increase in yield for the irrigation schedule presently practiced in the region. While the adverse impact of likely water shortage on wheat crops would be minimized to a certain extent under elevated CO_2 levels, it would largely be maintained for the rice crops resulting in net decline in the rice yields.

Mandal (1998), Chatterjee (1998) and Sahoo (1999) calibrated and validated the CERES-maize, CERES-sorghum and WOFOST models for the Indian environment and subsequently used them to study the impact of climate change (CO_2 levels: 350

and 700 ppm and temperature rise from 1 to 4°C with 1°C increment) on phenology, growth and yield of different cultivars. Chatterjee (1998) observed that an increase in temperature consistently decreased maize and sorghum yields from the present day conditions. Increase in temperature by 1 and 2°C, the sorghum potential yields decreased by 7–12%, on an average. An increase in 50 ppm CO_2 increases yields by only 0.5%. The beneficial effect of 700 ppm CO_2 was nullified by an increase of only 0.9°C in temperature.

However, Mandal (1998) observed that an increase in temperature up to 2°C did not influence potential and irrigated yields of chickpea as well as above ground biomass significantly. Pre-anthesis and total crop duration got reduced with the temperature rise. Nitrogen uptake and total water use (as evapo-transpiration) were not significantly different upto 2°C rise. The elevated CO_2 increased grain yield under potential, irrigated and rainfed conditions. There was a linear increase in grain yield, as the CO_2 concentration increased from 350 to 700 ppm. Potential grain yield of pigeonpea decreased over the control when the temperature was increased by 1°C (using WOFOST).

Sahoo (1999) carried out simulation studies of maize for climate change under irrigated and rainfed conditions. Rise in temperature decreased the yield under both the conditions. At CO_2 level of 350 ppm, grain yield decreased continuously with temperature rise till 4°C. This was possibly due to reduction in days to 50% silking and physiological maturity. At CO_2 level of 700 ppm, grain yield increased by about 9%. The temperature rise effect in reduction of yield was noted in several maize cultivars. Effect of elevated carbon dioxide concentration on growth and yield of maize was established, but less pronounced when compared with crops, like wheat, chickpea and mustard crops. The beneficial effect of 700 ppm CO_2 was nullified by an increase of only 0.6°C in temperature. Further increase in temperature always resulted in lower yields than control.

The sensitivity experiments of the CERES-rice model to CO_2 concentration changes, as conducted by (Saseendran et al. 1999), indicated that over the Kerala State, an increase in CO_2 concentration led to yield increase due to its fertilization effect and also enhanced the water use efficiency. The temperature sensitivity experiments have shown that for a positive change in temperature up to 5°C, there is a continuous decline in the yield. For every one degree increment, the decline in the yield is about 6%. Also, in another experiment, it was noticed that the physiological effect of ambient CO_2 at 2°C in temperature was compensated for the yield losses at 425 ppm CO_2 concentration.

Estimates of impact of climate change on crop production could be biased depending upon the uncertainties in climate change scenarios, region of study, crop models used for impact assessment and the level of management. Aggarwal and Mall (2002) studied the impact of climate change on grain yields of irrigated rice with two popular crop simulation models – Ceres-Rice and ORYZA1N at different levels of N management. The climate change scenarios used were 0.1°C increase in temperature and 416 ppm CO_2 (2010 scenario) and 0.4°C temperature and 755 CO_2 (2070 scenario) as the optimistic scenario, whereas increase of 0.3°C temperature and 397 ppm CO_2 (2010 scenario) and 2.0°C temperature and 605 ppm CO_2 (2070 scenario) as the pessimistic scenarios of climate change, as adopted from studies of Watson et al. (1998). The results showed that the direct effect of climate change on rice crops in different agro-climatic regions in India would always be positive irrespective of the various uncertainties. Depending upon the scenario, rice yields increased between 1.0 and 16.8% in pessimistic scenarios of climate change depending upon the level of management and model used. These increases were between 3.5 and 33.8% in optimistic scenarios. These conclusions are highly dependent on the specific thresholds of phenology and photosynthesis to change in temperature used in the models. Caution is needed in using the impact assessment results made with the average simulated grain yields and mean changes in climatic parameters.

Screening of cultivars for tolerance to sterility under enhanced temperatures during post anthesis phase for the major crops needs to be evaluated in the phytotron (control chambers), for choosing the appropriate cultivars for sustained productivity under climate change. Quality aspects for important crops, like wheat (aestivum and durum), basmati rice and mustard under the climate change, need to be addressed. There is also a need to develop a selection criterion for screening of the cultivars for adaptation to drought and temperature stresses.

Adaptation of crops to gradual change in the climatic conditions needs to be included in the existing crop growth models, as it is not well understood. Moreover, the suitable agronomic and resource management options may nullify the ill effects of climate change on growth and yield of crops.

2.3.2 Soil Productivity

The most important process is the accelerated decomposition of organic matter, which releases the nutrients in short run, but may reduce the fertility in the long run. Soil temperature influences the rates at which organic matter decomposes, nutrients are released and taken up, and plant metabolic processes proceed. Chemical reactions, that affect soil minerals and organic matter, are strongly influenced by higher soil and water temperature. Soil productivity and nutrient cycling are, therefore, influenced by the amount and activity of soil microorganisms. Soil microorganisms fulfill two major functions, i.e. they act as agents of nutrient element transportation as well as store carbon and mineral nutrients (mainly N, P and S) in their own living biomass, acting as a liable reservoir for plant available nutrients with a fast turnover. The doubling of CO_2 increases plant biomass production, soil water use efficiency by the plants, and C/N ratios of plants. The changes in the C/N ratios of plant residues returned to the soil, have impact on soil microbial processes and affect the production of trace gases NOx and N_2O.

Results of the All India Co-ordinated Long-term Fertility Trials indicate that regions, having higher organic carbon content (>0.6%) in the beginning, showed a declining trend, whereas the regions with lower organic carbon content remained more or less static or slight increase in the organic carbon content was noticed in around 25 years. In general, Indian agricultural soils are low in organic carbon content, and for achieving higher agricultural production, we have to depend upon the fertilizers. The hypothesis of increased organic carbon degradation with temperature rise has to be linked with the crop intensity factor, which is significantly higher for India, where proportion of the small and marginal land holdings is increasing due to rapid growth in population with time.

The interaction of nitrogen, irrigation and seasonal climatic variability, particularly at low input of irrigation, has several implications. Under adequate moisture supply situation, like for Punjab and Haryana, the yield benefits are obtained up to higher nitrogen application, whereas in the regions of limited to moderate water supply situations, the increasing trends in yield are noted up to relatively lower values of nitrogen. At low levels of water availability, it is difficult to decide optimal levels of N fertilizer for maximizing yield returns in view of uncertainty of N response, which is strongly related to a good post monsoon rainfall received during crop growing period (Kalra and Aggarwal 1996).

Das and Kalra (1995) evaluated the fertilizer and resource management for enhancing crop productivity under inter-annual variations in weather conditions. The results revealed sensitivity of crop yields to climatic variability and the need of inputs management in relation to climatic variability. Simulation models for judging the soil nutrient availability and subsequently relating to growth and yield of crops are available, but needs to be refined and thoroughly tested for the climate change event.

Analysis of the food grains production data for the last few decades reveals a tremendous increase in yield due to technological advancement, but it appears that impact of vagaries of monsoon has been large throughout the period. The annual food production showed an increasing trend, and the deviations around the technology trend line were significantly related to seasonal rainfall. But no definite trend is noticed in case of rabi season food production with the winter season’s rainfall, as majority of the food production in this season comes from the irrigated areas.

Changes in rainfall due to global climate change may affect the surface moisture availability, which becomes important for germination and crop stand establishment in the rainfed areas. Modifications in the surface and ground water availabilities with the rainfall change, are difficult to be observed when the land use and land cover are so rapidly changing.

Farmers have several agronomic management options to face the situation of water scarcity, through choice of crops, cultivars, adoption of suitable irrigation, nutrient and pesticides application schedules.

Water production functions, which relate to water availability and its use with crop yields, help in identifying critical growth stages at which the limited amount of water can be applied to get the maximum benefits (Kalra and Aggarwal 1994).

Soils dominate the cycling of many atmospheric trace gases because of the highest abundance and diversity of microbes in them. Earlier, equilibrium used to exist between the sources and sinks of GHGs, but a shift in this equilibrium has started becoming evident as a consequence of human induced activities. In order to comprehend the shift of source – sink equilibrium, one needs to understand the processes involved in generating the net flux (a function of production processes, consumption processes and gas transport) at the soil atmosphere interface.

Microbes have emerged as the major contributor as well as consumer of GHGs as they are the main intermediaries of C turnover in soil. They are also considered as sole agents for soil humus formation, cycling of nutrients, soil tilth and structure and also perform myriad of other functions. What will happen to the soil fertility in the event of global climate change needs to be addressed through soil organic matter (SOM)? The assessment of soil health/quality/fertility through changes in SOM is difficult, and therefore, other soil parameters are being used as proxy indicators. For example, soil microbial biomass (the living part of organic matter) due to dynamic character has been shown to quickly respond to changes and perturbations, often before the measurable changes occur in organic C and N, thus acting as an indicator of long term changes in SOM content (Powlson and Brookes 1987). However, the measurement of microbial biomass (C_{mic}) alone will not serve the purpose, because they are generally influenced by climatic variables. Hence, for real measurement of the impact of soil processes, one needs to consider proportion of total organic C or N within the microbial biomass i.e. microbial quotient. Under the equilibrium conditions, C_{org} of agricultural soils contains 2.3–4% C_{mic}. Soils exhibiting a C_{mic} to C_{org} ratio higher or lower than these values appearing in the equilibrium line would be either accumulating or losing C, respectively (Anderson and Domsch 1986). Different climatic conditions, in particular precipitation/evaporation, influence the equilibrium C_{mic} to C_{org} ratio (Insam 1990) and a very high correlation was found in which 73% variation could be explained with the quadratic function, and thereby one can predict the soil fertility in terms of accumulation or losses of C.

Under the changed scenario of atmospheric composition due to global warming, the tropical region, such as India, with small organic C reserves, will show net efflux of CO_2, because rates of soil respiration increase exponentially with temperature. Thus, CO_2 effluxes from tropical system should increase markedly with small change in temperature without any increase in inputs from the above ground communities, thereby leading to rapid losses over a short period of a few decades and later on, it will sustain the balance because of the shortage of substrate for decomposition as well as adaptation of microbial communities towards the climatic change. The alterations in microbial community structure and their physiology can be interpreted in terms of differences in phospho-lipid fatty acids (PLFA) – fingerprinting. In general, PLFA profiles had decreasing unsaturation, greater chain length and larger number of cyclopropyl fatty acids at higher temperatures.

It has also been suggested that climate change could increase rates of soil erosion, further hampering food production. Increases in rainfall will accelerate the rates of soil loss, reducing farm productivity even more. A further negative consequence of accelerated erosion will be increased sedimentation in streams and reservoirs. This will shorten the life span of dams, which helps to prevent floods and provide both electricity and water for irrigation. Another way, in which erosion could accelerate, is through a decrease in rainfall, which could lead to dry spells and increased risk of wind erosion (Parry et al. 1999). If erosion rates go unchecked, continued soil impoverishment would eventually force farmers to abandon their lands. Thus, erosion is among the major threats to food production in a warmer climate. But, these qualitative assessments have not been studied in depth, where the rapid changes in land use patterns may totally reverse our thinking.

Other land degradation problems, such as water logging, soil salinity and sodicity development, are emerging due to rapid land use pattern and land cover changes. The impact of climate change on these aspects needs to be looked into for sustaining the agricultural production.

2.3.3 Insects and Pests

Incidence of pest and diseases is most severe in tropical regions due to favorable climate/weather conditions, multiple cropping and availability of alternate pests throughout the year. Therefore, in the south Asia, pests and diseases deleteriously affecting the crop yields are prevalent. Climatetors are the causative agents in determining the population fluctuations of pests. They influence plant disease establishment, progression and severity. In fact, a clear understanding of population dynamics, as influenced by abiotic and biotic parameters of environment, is of much help in pest forecasting and to formulate control measures.

Indicators of climate change can be a few of the crop species, rhythm/migratory behavior of specific insects/birds, etc. The global warming may affect growth and development of all organisms including insect-pests themselves. Among all the abiotic factors, temperature is the most important one affecting insect distribution and abundance in time and space, since these are cold-blooded animals. The insects cannot regulate their body temperature and thereby, ambient temperature influences their survival, growth, development and reproduction.

The swarms of locust produced in the Middle East usually fly eastward into Pakistan and India during summer season and they lay eggs during monsoon period. The swarms as a result of this breeding, return during autumn to the area of winter rainfall, flying to all parts of India and influencing *kharif* crops (Rao and Rao 1996). Changes in rainfall, temperature and wind speed may influence the migratory behaviour of locust.

Diseases are often hurdles in increasing rice productivity. The rice blast, caused by *Pyricularia grisea*, is most prominent disease across the eco-systems. In the past, rice blast, brown spot and stem rot, were the serious diseases. Consequent to the adoption of high yielding varieties and associated agronomic practices during 1970's, diseases like bacterial leaf blight, sheath blight, sheath rot, tungro virus (transmitted by *Nephotettix* spp.) and bacterial leaf streak, have gained importance over the traditionally known diseases, especially stem rot and brown spot. False smut and discolouration of rice grain, caused by several fungi, have been of minor significance with occasional concern in certain regions only. While analyzing the effect of climatic variability and change on disease status, the interaction of land use and land cover change should also be taken into consideration.

Climate and weather selectively induce specific diseases to develop. The monocyclic diseases, such as stem rot, sheath rot and false smut, are less influenced by the ambient weather conditions. Epidemics of monocyclic diseases are relatively rare in the sense of an explosive increase in their population. In contrast, the polycyclic diseases, such as blast, brown spot, bacterial leaf blight and rice tungro virus that invade the aerial parts of the plants, are subjected to constant interaction with weather. They easily attain epidemic proportions to cause heavy losses (Abrol and Gadgil 1999).

Forecasting of aphids (*Lipaphis erysine* Kalt) on mustard crop, grown during winter season in northern part of India based on the movement of western disturbances, has been established (Ramana Rao et al. 1994). Western disturbances bring in cold and humid air from the Mediterranean region, resulting in cloudy and favourable weather conditions for occurrence of aphids on mustard crop. It was observed that there was a sharp increase in the population of aphids when the mean daily temperature ranged from 10 to 14°C, with relative humidity of 67–85% and cloudiness greater than 5 octas.

For every insect species, there is a range of temperature within which it remains active from egg to adult stage. Lower value of this range is called threshold of development or developmental zero. Within favourable range, there is an optimum temperature where most of the individuals of a species complete their development. Exposure to temperature on either side exerts an adverse impact on the insect by slowing down the speed of development (Pradhan 1946).

The studies have shown that insects remain active within temperature range from 15 to 32°C (Phadke and Ghai 1994). In case of red cotton bug, at constant temperature of 20, 25 and 30°C, the average duration of life cycle was found to be 61.3, 38.3 and 37.6 days, respectively, while at 12.5 and 35°C, the pest did not show any development (Bhatia and Kaul 1966).

A maximum temperature ranging from 19 to 24°C with a mean of 12–15°C for mustard aphid, *Lipaphis erysimi*; maximum temperature between 26.9 and 28.2°C with a relative humidity of 80.6–82.1% for rice stink bug; temperature from 20 to 28°C for rice green leafhopper, temperature from 24.8 to 28.6°C for brown plant hopper; mean temperature around 27.5–28.5°C for aphids, thrips and leaf weevil on green gram and maximum temperature from 23 to 27.8°C for gram pod borer, have been found most congenial for their development (Phadke and Ghai 1994).

With the increase in temperature, the rate of development of insects may also increase, if temperature still lies within the optimal range for the pests. As a consequence, they could complete more number of generations for inflicting more loss to our crops. Crop-pest interaction needs to be evaluated in relation to climate change in order to assess the crop losses.

Development of diseases and pests is strongly dependent upon the temperature and humidity. Any change in them, depending upon their base value, they can significantly alter the scenario, which ultimately may result in yield loss. Any small change in temperature can result in changed virulence as well as appearance of new pests in a region. Likewise, crop-weed competition may be affected, depending upon their growth behaviour.

The following scenarios can be visualized regarding impact of climate change on pest dynamics in agriculture.

- With an increase in concentration of carbon dioxide, the nutritional status of crop will change, and the net effect on agricultural production will depend upon interaction between pests and crops.
- Gradual climate warming will lead to changes in the composition of pest fauna in different areas. The high population growth rate of many species will ensure changes in pest distribution.
- If the rise in winter temperature takes place, the duration of hibernation of pests may decrease, thus increasing their activity.
- Uncongenial areas for pests due to low temperature at present may become suitable due to rise in temperature.

However, we should not forget that insects could adapt to slow changes in the environment and with increase in temperature, their favorable range of temperature may also shift.

2.4 Socio-Economic Aspects

Socio-economic linkage is relatively complex, and needs to be linked through the bio-physical modifications associated with the climate change. Land use and land cover change in our country is changing rapidly due to several driving forces. Socio-economic aspects can be dealt in two ways, one working out the cost-benefit analysis for various climate change scenarios by using econometric-process models (Antle and Capalbo 2001) and the other, generating the socio-economic scenario of future which links with the cropping system model for further impact analysis.

World Bank report (1998) analyzed climate change effects on Indian agriculture, through annual net revenues, by using Ricardian method (Mendelsohn et al. 1994). The three methodologies, as adopted in the study, found Indian Agriculture sensitive to warming. The analyses further showed year-to-year climate sensitivity to the system's response. The studies revealed that net revenues fall precipitously with warmer April's, but also sensitive to warmer January and July. Crop revenues increased with October temperatures. Net revenues were also sensitive to precipitation, but the effects were smaller and off-setting. A warming scenario of $+2.0^{\circ}$C rises in mean temperature and a $+7\%$ increase in mean precipitation levels will create reduction in the net revenues, as revealed from the three approaches. The impact is differential on spatial and temporal scales. But the study seemed to be weak for linking with the biophysical aspects. Even then, this kind of study is a beginning of future plans of initiating the work in this regard.

2.5 Mitigation Options of Green House Gases Emission

The possible strategies for mitigating methane emission from rice cultivation can be made by altering water management, particularly promoting mid-season aeration by short-term drainage. Improving organic matter management by promoting aerobic degradation through composting or incorporating into soil during off-season drained period, is another promising technique. Organic amendments to flooded soils increase methane production and emission. However, application of fermented manure, like biogas slurry, reduces the emission (Debnath et al. 1996). In addition, nitrification inhibitors have been shown to inhibit methane emission. Another mitigation option may be selection of low CH_4 emitting rice cultivars, as cultivars grown in similar conditions show pronounced variations in methane emission (Mitra 2000). Screening of rice cultivars with few unproductive tillers, small root system, high root oxidative activity and high harvest index are ideal for mitigating methane emission from rice fields.

Combined with a package of technologies, methane emission can best be reduced by (a) the practice of mid-season drainage instead of continuous flooding, (b) direct crop establishment like dry seeded rice and (c) use of low C: N organic manure and biogas slurry.

Appropriate crop management practices, which lead to increase N use efficiency and yield, hold the key to reduce nitrous oxide emission. Application of nitrate (NO_3–N) fertilizers e.g. calcium ammonium nitrate (CAN), in crops with aerobic conditions and ammonium (NH_4–N) fertilizers e.g., ammonium sulphate, urea, in wetland crops also help reducing the nitrous oxide emission (Pathak and Nedwell 2001). Curtailing the nitrification process by the use of nitrification inhibitor may further decrease the N_2O emission from soil. There are some plant-derived organics, such as neem oil and neem cake, which can also act as nitrification inhibitors. These are being experimented in fields to reduce the emission of nitrous oxide and increase the fertilizer use efficiency. Other biocidal inhibitors, such as karanja seed extract, have been found to retard nitrification by 60–70% (Majumdar et al. 2000). The efficacy of various mitigation technologies, however, needs to be tested in farmers' fields. Moreover, such technologies need to be also assessed for non-target effects and economic feasibility.

2.6 Vulnerability and Adaptation Strategies

There must be a clear understanding of vulnerable populations and regions, based on an assessment of the capacities to cope with climate variability and change. We are conscious that coping and adaptation strategies are not equally available to all affected populations. At the same time, it is important also to develop formal measures of vulnerability and their application to planning adaptation measures and strategies. The inter-disciplinary work involved requires various Ministries, Agencies and Expert Institutions to pool their resources, knowledge and information. We need to know much more about the factors influencing vulnerability and the aspects related to planning for adaptation. Our understanding in the area of vulnerability and adaptation tools needs to be mature and be refined so as to enhance their applicability.

India is particularly vulnerable to likely increase in the incidence of extreme events. The impacts of climate change could hinder development and progress in eradicating poverty and potentially aggravating social and environmental conditions. In the context of the current debate about climate change, it is necessary to show that the developing countries, like India, are taking considerable actions in terms of policies, programmes and projects. Technology transfer can speed up the modernization process and additional funds can accelerate government initiatives in energy conservation. However, policies for poverty alleviation must be on high priority.

The hierarchy of damage considerations as discussed above – hunger, regional economic, farmer/farm sector, and yield vulnerability, helps to focus on adaptive strategies that reduce vulnerability. How can we avoid yield failures? If yields fail, what other crops can be grown? If farming becomes uneconomic, what can be alternate land use options to increase the profits to the farmers?

Historically, farming systems have adapted to changing economic conditions, technology and resource availabilities and have kept pace with a growing population. While the technological potential to adapt may exist, the socio-economic capability to adapt differs for different types of agricultural systems. An evergreen revolution is the pathway to sustainable advances in productivity per units of land, water and time without associated ecological or social harm. One of the weaknesses is mismatch between production and post-harvest technologies and between production and market demand, and the consequent need for the Government of India to undertake "trade relief" operations like cyclone, flood and drought relief. We can face the internal threats through integrated attention to regulation, education and social mobilization through Panchayati Raj institutions. Also, there is a need to restructure research strategies in a manner that strategic, anticipatory and participatory (i.e. with farm families) research, all receive adequate attention.

The Rural Knowledge Centers should provide computer aided and internet connected information services, so that farm families have timely and relevant meteorological, management and marketing information. Another area, which needs an urgent attention, is the restructuring of the State Land Use Boards in a manner that they are in a position to offer proactive advice to farm families on land use and cropping systems, based on likely monsoon behaviour, ecological efficiency and trends in prices and markets. Assured and remunerative marketing opportunities hold the key to sustaining farmers' interest in producing more.

Immediately, an action is needed to defend the productivity gains we have already made and to extend the same to the areas which have been bypassed by the farm revolution, particularly dry farming areas, and to make new gains through sustainable intensification, market – based farming systems diversification, and value addition to primary produce through agro-processing and agri-business.

The income and on-farm and off-farm employment potential of farming can be improved only through integrated farming systems, based on crop-livestock-fish-trees combinations. Multiple livelihood opportunities are essential both as an insurance mechanism and for a reasonable total "take-home" income. India's strength lies in a farming systems approach to the use of natural resources. This is also the pathway to ecological farming. Such research is best done in farmer's fields through a participatory approach. Conservation of bio-resources, particularly medicinal plants and agro biodiversity in dry farming areas, and their conversion into economic products through biotechnology, will help to end the situation where "poor people inhabit a rich country".

2.7 Conclusion

The climate change, as realized through trends of temperature rise and increased CO_2 concentration, is a major concern. In the recent past, the number of studies for assessing its impact on agriculture has increased. Crop growth models have been modified and tested for various important crops of this region under different climate change scenarios. But most of the results happen to be region specific and with certain assumptions. Accuracy in assessing the magnitude of the climate change on higher spatial and temporal resolution scale is the prime requirement for accurate estimates of the impact. The extent of inter- and intra-annual variability in climate happens to be large in this region, and the crops respond differentially to these changes. Understanding of this differential behavior can aid in working out the impact of climate change. The vast genetic diversity in crops provides a platform to identify suitable thermal and drought tolerant cultivars for sustained productivity in the changed climate. Identification of suitable agronomic management practices can be a potential solution to optimize agricultural production in the changed climate. To have an overall assessment of soil health with the climate change, the possible alterations in soil physical, chemical and biological characters need to be looked into by also including land use and land cover change driving forces. Intensive cultivation in our country has already started showing signs of yield stagnation in some parts of north-west India, raising the alarm of sustaining the yields by adoption of suitable agronomic management options. This concern has now to be viewed along with the climate change and its variability. Increased frequency of droughts and floods in this region, as anticipated in the climate change scenarios, caution us to identify suitable "no regrets and no risks" management options to face the situation. Crop simulation technique offers an opportunity to link the climate change with the other socio-economic and bio-physical aspects. These models can effectively work out the impact and also suggest suitable mitigation options to sustain the agricultural productivity. But one has been cautious in extrapolating the results to a larger region, as most of the exercises are done with certain assumptions, otherwise the results can be misleading. The crop-pest-weather interaction studies, conducted in the past, need to be thoroughly investigated for developing a sub-routine to link with the crop growth models to give the realistic estimates. Socio-economic aspects of climate change are relatively weak, and future scenarios are to be generated for various agro-ecological regions for subsequently linking with other relational layers to work out the impact.

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Chapter 3
Simulation Studies to Characterize the Impact of Climate Change on Crop Production and to Identify Strategies for Adaptation and Mitigation

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3.1 Introduction

The Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC 2007) emphatically shows that the Earth's climate is changing in a manner unprecedented in the past 400,000 years. By the end of 2100, the mean planet-wide surface temperatures will rise by 1.4–5.8°C, precipitation will decrease in the sub-tropics, and extreme events will become more frequent (IPCC 2007). However, changes in climate are already being observed – the last 60 years were the warmest in the last 1000 years and changes in precipitation patterns have brought greater incidence of floods or drought globally. These predicted changes are largely driven by increasing atmospheric concentrations of greenhouse gases, such as CO_2, CH_4 and N_2O. These changes will affect the agro-climatic conditions for food production systems worldwide. Just as the production ecosystems are influenced by these changes, the agricultural activities also contribute to these changes by unsustainable utilization of natural resources, altered biogeochemical cycling of nutrients, and emission of greenhouse gases.

IPCC, in its third assessment report, itself has demonstrated that the global climate changed significantly during the 20th Century, and it may continue to change more precipitously in the coming centuries, irrespective of whether attempts at mitigation through implementation of the Kyoto Protocol to UNFCCC (United Nations Framework Convention on Climate Change) are successful (IPCC 2001). The magnitude and direction of change in the various climate elements will differ from one major region to the other; these changes can be beneficial in certain regions and detrimental elsewhere. But, the less developed countries and regions are likely to

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experience the worst of the consequences of climate change, partly because of negative changes in water availability in the tropical regions and partly because the communities concerned are poorly equipped to adapt. The major sector, that will be exposed to the potential negative changes in climate, is food production. While trying to roll back climate change through the implementation of the Kyoto Protocol, the formulation of strategies for living with changing global climate becomes imperative. Such strategies will require a better understanding of the observed impacts of climate variability and the potential impacts of predicted changes.

Besides concerns about the changing climate, the demand for higher level of control over environment in which plants grow is, otherwise, also increasing due to intensive agriculture. These controls can range from better strategies of soil management to "closed" environments, where most, if not all, the atmospheric and soil variables can be adjusted. It becomes essential to have the plant growth and development models, to make a basis for planning and managing crop production. The crop modeling can also be useful as a means to help the scientists define research priorities. Using the models to estimate the importance and the effect of certain parameters, a researcher can observe which factors should be more studied in future research to increase the understanding of the system.

3.2 Simulation and Modeling in Crop Sciences

To simulate means to imitate, to reproduce, or to appear similar and the art of simulating is as old as man. Since the origin of the civilization, man had to struggle for his survival using, even if unconsciously, simulations of real future processes. Simulation is an analogy with the reality, being common in many areas. Model is a word that admits several connotations, among which the following can be mentioned: (i) the representation of some entity, usually in smaller size than the original; (ii) a simple description of a system, used to explain it or to perform calculations (Swain et al. 2006, 2007). Based on the above definitions, the models can be a prototype, a simplified representation, as well as an abstraction of a reality (system). In crop sciences, simulation and modeling are important to forecast the results of management of a certain system or of a certain environmental condition (Aggarwal et al. 2006a, b).

3.3 Classification of Models

Models can be classified into different types: conceptual, physical or mathematical (Acock and Acock 1991). Every hypothesis, that is tested, has a conceptual model supporting it. The physical down-scaled representations (physical models) of the system have been used by engineers for a long time. But, they are rarely used to represent biological systems. However, a plant in an experimental plot or container can be considered as a physical model of the crop in the field. When the behavior of a system is described mathematically, through equations, then the representation of the system is a mathematical model. The mathematical model represents quantitatively assumed hypotheses about the real system, allowing one to deduce its consequences. They have gradually become more popular, yet more sophisticated, because advanced level of computing and logical operations has become more accessible with computers.

Mathematical models can be classified into a number of classes, but the two main ones are (i) empirical, sometimes called correlative or statistical models describing relationships among variables without referring to the correlated processes and (ii) mechanistic models, at the level of processes or simulators, which are also called explanatory models, trying to represent cause-effect relationships among the variables. The major objective of mathematical modeling is, evidently, to produce a tool that can be used to test hypotheses, to generate alternative hypotheses, to suggest experiments, to refute them, and furthermore, to predict the behavior of the system in unknown situations. The mathematical models follow exactly the basic rules of the scientific method which are: (i) observation of a system; (ii) formulation of a hypothesis in the attempt of explaining the observations; (iii) prediction of the system behavior through simulation; and (iv) experimentation to test the validity of the hypothesis (validation of the model). But, there is no such thing as a right or wrong model, but models provide variable degrees of suitability for a certain circumstance. There is no universal model that provides a solution to all problems, however, models should continue to be developed and adapted to several particular situations. In the plant growth and development, an empirical model can describe the plant behavior, based on direct observations at the plant level, while a mechanistic model of vegetative growth describes the plant performance based on the knowledge of the processes that are taking place.

3.4 Agricultural Sciences: Use of Mathematical Models

The science of agriculture depends on research activities for (i) the acquisition of knowledge; (ii) the ordering of knowledge and the development of understanding on that knowledge, and (iii) the application of the knowledge and/or understanding to the solution of practical problems (Rimmington and Charles-Edwards 1987). The mathematical models can be used in different ways within each one of these three activities. Basically a simplified description of a system, a mathematical model can help us to better understand the operation of a real system and the interactions of its main components. Thus, they are excellent forecast mechanisms. The important uses of mathematical models in agricultural sciences can be (i) analysis of observed responses in plant growth as a function of certain factors, to increase our understanding of the crop growth and to provide direction in our research; (ii) simulation of plant growth by models consisting of many interacting components and levels, as an aid for teaching and learning; and (iii) forecast of the plants response to certain climatic or management condition, as a tool for management and decision-making.

Besides its scientific importance, the simulation of plant yield has practical application in the management of cropping systems, in the formation of stocks, in the commercialization, in the making of agricultural policies and zoning, and in many other branches of agricultural activity. Before the model is applied for resource management, its accuracy needs to be tested within a given range of variables. Only then, it is wise to use the model to simulate the effects of different management techniques or environmental variations on the crop performance. The model should necessarily be used within its tested boundaries (Rimmington and Charles-Edwards 1987).

3.5 Projected Climate Change Scenarios over Indian Sub-continent

Climate change is no longer a distant scientific prognosis, but is becoming a reality. Using the UKMO GCM model, Bhaskaran et al. (1995) predicted a total precipitation increase of approximately 20% and an increase in winter or *rabi* crop season temperature by 1–4°C with increased CO_2 concentration. The specific humidity increases by 19%, indicating that the increased monsoon rainfalls were largely due to increased water content of the atmosphere. Greater number of heavy rainfall days during the summer monsoon or *kharif* period, and an increased interannual variability were predicted. Recently, Lal et al. (2001) estimated larger uncertainty associated with projected *rabi* rainfall than *kharif* rainfall in 2050s. The standard deviation of future projections of area-averaged monsoon rainfall centered around 2050s was not significantly different relative to the present-day atmosphere, implying thereby that the year-to-year variability in mean rainfall during the monsoon season might not significantly change in the future. More intense rainfall spells were, however, projected over the land regions of the Indian sub-continent in the future, thus increasing the probability of extreme rainfall events in a warmer atmosphere.

Rupakumar and Ashrit (2001) projected an increase of 13% in monsoon or *kharif* season rainfall in India using ECHAM4 model, while HadCM2 model suggested a reduction in *kharif* rainfall by 6% due to the greenhouse gas simulation. Both GCMs suggested an increase in annual mean temperature by more than 1°C (1.3°C in ECHAM4 and 1.7°C in HadCM2). Rupakumar et al. (2003) concluded that the coupled models indicated general warming and enhanced rainfall conditions over India towards the later half of the 21st century, in a GHG increase scenario; however, there was some disagreement among the models on rainfall changes, with respect to decreases in rainfall in some states (viz. Jammu and Kashmir, Himachal Pradesh, Bihar, Gujarat and Rajasthan etc.). May (2002) predicted an intensification of rainfall in Indian region during monsoon season as a consequence of the anticipated increases in the greenhouse gas concentrations.

The increase in the regional rainfall was found to be related to an intensification of the atmospheric moisture transport into the Indian region. Stephenson et al. (2001) observed a weakening of the large scale aspects of the Indian summer monsoon. Rupakumar et al. (2003) concluded that under future scenarios of increased greenhouse gas concentrations (GHG), there would be marked increases in both rainfall and temperature into the 21° century, which will become more conspicuous after 2040s in India. Over the region south of 25°N (south of cities, such as Udaipur, Khajuraho and Varanasi), the maximum temperature would increase by 2–4°C during 2050s. In the northern region, the increase in maximum temperature might exceed 4°C, which might exceed over the southern peninsula, northeast India and some parts of Punjab, Haryana and Bihar.

3.6 Simulating the Impact of Climate Change on Agriculture

Estimating the effect of changing climate on crop production in India is more difficult due to a variety of cropping systems and levels of technologies used. The use of crop growth models holds a promise in which these effects can be studied. Despite the need for simplifying a large number of assumptions, these models allow a better understanding of the complex interaction between the main environmental variables influencing crop yields. There have been a number of studies in India, which are aimed at understanding the nature and magnitude of yield gains or losses of crops at selected sites under elevated atmospheric CO_2 and associated climatic change (Abrol et al. 1991; Sinha and Swaminathan 1991; Aggarwal and Sinha 1993; Aggarwal and Kalra 1994; Gangadhar Rao and Sinha 1994; Mathauda and Mavi 1994; Gangadhar Rao et al. 1995; Mohandass et al. 1995; Lal et al. 1998, 1999; Francis 1999; Saseendran et al. 2000; Mall and Aggarwal 2002; Aggarwal and Mall 2002; Aggarwal 2003; Attri and Rathore 2003; Mall et al. 2004; Krishnan et al. 2007). There is also report of a decline of 600–650 grains m^{-2} in wheat crop with every 1°C increase in mean temperatures above 17–17.7°C during the terminal spikelet initiation to anthesis.

Table 3.1 lists the various simulation studies done on the impact of climate change on yield of different crops. The integrated impact of a rise in temperature and CO_2 concentration on yield of crops might be negative (Sinha and Swaminathan 1991). They estimated that a 2°C increase in mean air temperature could decrease rice yield by about $0.75\,t\,ha^{-1}$ in the high yield areas and by about $0.06\,t\,ha^{-1}$ in the low yield coastal regions. Further, a 0.5°C increase in winter temperature would reduce wheat crop duration by seven days and reduce yield by $0.45\,t\,ha^{-1}$. An increase in winter temperature of 0.5°C would thereby translate into a 10% reduction in wheat production in the high yield states of Northern India. In another report, Achanta (1993) simulated irrigated yields for Pantnagar district under doubled CO_2 and increased temperature and concluded that the impact on rice production would be positive in the absence of nutrient and water limitations. Aggarwal and Sinha (1993) reported that, at 425 ppm CO_2 concentration and no rise in temperature, wheat grain yield at all levels of production (i.e. potential, irrigated and rainfed) increased significantly.

Gangadhar Rao and Sinha (1994) showed that wheat yields decreased due to the adverse effects of temperature during grain filling and maturity stages of growth.

Table 3.1 Simulated impact of climate change on selected crops in India

Crop	Simulation model used	Region	Effect	References
Rice		All India	Positive	Sinha and Swaminathan (1991)
Rice		North India	Positive	Achanta (1993)
Rice	ORYZA1	All India	Positive	Mohandass et al. (1995)
Rice	CERES-rice	Punjab	Negative	Hundal and Kaur (1996)
Rice	CERES-rice	North-West	Positive / Negative	Lal et al. (1998)
Rice	CERES-rice	South India	Positive / Negative	Saseendran et al. (2000)
Rice	CERES-rice and ORYZA1N	All India	Positive / Negative	Aggarwal et al. (1997)
Rice	CERES-rice and ORYZA1N	All India	Positive / Negative	Aggarwal and Mall (2002)
Rice		North India	Positive / Negative	Aggarwal (2003)
Rice	ORYZA1 INFOCROP-rice	Eastern India	Positive / Negative	Krishnan et al. (2007)
Wheat		All India	Negative	Sinha and Swaminathan (1991)
Wheat		All India	Positive / Negative	Aggarwal and Sinha (1993)
Wheat		All India	Negative	Gangadhar Rao and Sinha (1994)
Wheat	WTGROWS	All India	Positive / Negative	Aggarwal and Kalra (1994)
Wheat	CERES-wheat	Punjab	Positive / Negative	Hundal and Kaur (1996)
Wheat	CERES-wheat	North West	Positive / Negative	Lal et al. (1998)
Wheat		North India	Positive / Negative	Aggarwal (2003)
Sorghum	CERES-sorghum	All India	Positive / Negative	Gangadhar Rao et al. (1995)
Sorghum	CERES-sorghum	All India	Negative	Chatterjee (1998)
Soybean	CROPGRO-soybean	Central India	Positive / Negative	Lal et al. (1999)
Soybean	CROPGRO-soybean	All India	Positive / Negative	Mall et al. (2004)
Maize	CERES-maize	Punjab	Negative	Hundal and Kaur (1996)
Maize	CERES-maize	North India	Positive / Negative	Sahoo (1999)
Groundnut	PNUTGRO	Punjab	Negative	Hundal and Kaur (1996)
Chickpea	CROPGRO-chickpea	All India	Negative	Mandal (1998)

Aggarwal and Kalra (1994), evaluated the WTGROWS crop simulation model to estimate the effect of climate change on productivity of wheat in India, and simulated normally sown crops at three levels of production (potential, irrigated and rainfed) with the assumption of CO_2 level at 425 ppm and temperature rise options of 0, 1 and 2°C. At the 425 ppm CO_2 concentration and no rise in temperature, grain yield at all levels of production increased significantly at all places. One degree celsius rise in mean temperature had no significant effect on the potential yields. Irrigated yields, however, showed a small increase in most places where current yields were greater than 3.5 t ha^{-1}. In the central and peninsular India, where current irrigated yields were between 2 and 4 t ha^{-1}, the response varied from significant decreases to significant increases. The rainfed yields, however, showed a significant increase. But, an increase of 2°C in temperature reduced potential yields at most places. In sub-tropical (above 23°N) environments, there was a small decrease in potential yields (1.5–5.8%), but in tropical locations, the decrease was 17–18%. In the same study, the mean simulated yield of wheat for current and changed climate scenario (2°C rise and CO_2 level of 425 CO_2 ppm) in different latitudinal ranges when evaluated, showed that the irrigated yields slightly increased for latitudes greater than 27°N, but reduced at all other places. The decrease in yield was much higher in lower latitude. Several locations, particularly where current rainfed yields were greater than 2 t ha^{-1}, showed a very significant increase in the rainfed yields with climate change. Depending upon the magnitude of temperature increase, crop duration, particularly the period up to anthesis, was reduced. Unless accompanied with suitable research and policy interventions, the reduction in productivity under changed climate might reduce wheat production options in central India (Aggarwal and Kalra 1994; Aggarwal 2000).

The impact of climate change on the productivity of sorghum [*Sorghum bicolor* (L.) Moench] in three diverse growing areas in India (i.e., Hyderabad, Akola and Solapur) was studied by Gangadhar Rao et al. (1995), using the CERES-sorghum simulation model (Ritchie and Alagarswamy 1989) under climate change scenarios generated by the three GCMs, namely Goddard Institute of Space Studies (GISS), Geophysical Fluid Dynamics Laboratory (GFDL), and United Kingdom Meteorology Office (UKMO). The simulated results indicated a decrease in yield and biomass of rainy season sorghum at Hyderabad and Akola under all climate change scenarios. The post rainy season sorghum, grown at Solapur on stored soil water, showed a marginal increase in yield. The positive effects of increased CO_2, if any, were masked by the adverse effects of predicted increase in temperature, resulting in shortened crop growing seasons. For rice crop production in India, Mohandass et al. (1995) used ORYZA1 model developed by Kropff et al. (1994) to simulate under current and future climate scenarios and showed an increase in rice production under the GCMs scenarios used. This was mainly due to the fertilizing effect of the increased CO_2 level more than any detrimental effects of increased temperatures. Large decreases in yields were predicted for second season crops at many of the locations due to high temperatures prevailing, but its overall effect on the rice production was small due to relatively low proportion of total rice produced. Uprety et al. (1996) concluded that with the type of climate in the northern belt of Indian subcontinent, *viz.*, variation in temperatures and CO_2 concentration, the production of Brassica crop (an oilseed crop) is likely to increase and to be shifted to relatively drier regions. Hundal and Kaur (1996) examined the climate change impact on productivity of wheat, rice, maize and groundnut crops in Punjab, using CERES-wheat (Godwin et al. 1989), CERES-rice (Singh et al. 1993), CERES-maize and "PNUTGRO" crop simulation models. If all other climate variables were to remain constant, temperature increase of 1, 2 and 3°C from present day condition, would reduce the grain yield of wheat by 8.1, 18.7 and 25.7%, rice by 5.4, 7.4 and 25.1%, maize by 10.4, 14.6 and 21.4% and seed yield in groundnut by 8.7, 23.2 and 36.2%, respectively.

Lal et al. (1998) examined the vulnerability of wheat and rice crops in northwest India to climate change through sensitivity experiments with CERES-wheat and CERES-rice models and found that under elevated CO_2 levels, yields of rice and wheat increased significantly (15 and 28% for a doubling of CO_2). However, a 3°C and 2°C rise in temperature annulled the positive effect of elevated CO_2 on wheat and rice, respectively. The combined effects of enhanced CO_2 and imposed thermal stress on the wheat and rice crop were 21 and 4% increase, respectively, in yield for the irrigation schedule presently practiced in the region.

Chatterjee (1998) used CERES-sorghum model and observed that an increase in temperature consistently decreased the sorghum yields from the present day conditions. Increase in temperature by 1 and 2°C in sorghum decreased the grain yields by 7–12%, on an average. A further increase in temperature drastically reduced the yields by 18–24%, on an average. The magnitude of decrease in yield with increase in temperature was, in general, proportional to the increase in temperature in most years, indicating that there was no large interaction effect between yearly climatic variation and increase in temperature. Similarly, the small beneficial effect of still higher CO_2 concentrations was nullified by further increase in temperature.

Mandal (1998) used the CROPGRO-chickpea model and observed that an increase in temperature up to 2°C did not influence potential yield as well as above ground biomass of chickpea significantly. Lal et al. (1999) projected 50% increased yield for soybean for a doubling of CO_2 in Central India by using CROPGRO-soybean model. However, a 3°C rise in the surface air temperature almost nullified the positive effects of doubling of carbon dioxide concentration and reduced the total duration of the crop (and hence productivity) by inducing early flowering and shortening the grain fill period. Soybean crops in Central India are found to be more vulnerable to increase in maximum temperature than in minimum temperature. Acute water stress due to prolonged dry spells during monsoon season could be a critical factor for the soybean productivity even under the positive effects of elevated CO_2 in the future.

Sahoo (1999) used CERES-maize crop model and carried out simulation for irrigated and rainfed conditions. Rise in temperature decreased the maize yield in both the environments. At CO_2 level of 350 ppm, grain yield decreased continuously with temperature rise till 4°C and the yield was decreased by about 30% over the present day condition. Effect of elevated carbon dioxide concentration on growth and yield of maize was established, but less pronounced when compared with crops, like wheat, chickpea and mustard crop. The beneficial effect of 700 ppm CO_2 was nullified by an increase of only 0.6°C in temperature. Further increase in temperature always resulted in lower yields than control. For one of the IPCC scenario (an increase of 1.8°C temperature for India and 425 ppm CO_2 by the year 2030), potential maize yields would be severely effected (about 18%).

The sensitivity experiments of the CERES-rice model to CO_2 concentration changes conducted by Saseendran et al. (2000) indicated that an increase in CO_2 concentration would lead to yield increase due to its fertilization effect and also enhance the water use efficiency over the Kerala state. The temperature sensitivity experiments had shown that for a positive change in temperature up to 5°C, there was a continuous decline in the yield. For every one-degree increment, the decline in yield was about 6%. Nevertheless, the physiological effect of ambient CO_2 at 425-ppm concentration compensated for the yield losses due to increase in temperature up to 2°C.

The overall impacts of the climate change scenario for a 2°C rise in temperature and a 7% increase in precipitation, were negative and about 8.4% loss of the total farm level net-revenue for India, when the functional relationship between farm level net revenue and climate variables, introduced through linear, quadratic, and interaction terms, were estimated. Increases in temperature resulted in significant negative impacts, while higher precipitation considered under the scenario increased the net-revenue. Haryana, Punjab, and western Uttar Pradesh, which grow predominantly wheat in the winter season, experience the most negative effects, along with the coastal districts of Tamil Nadu. On the other hand, the eastern districts of West Bengal and parts of Bihar seem to benefit from the changes in future. When the impact of various climate change scenarios had been assessed on grain yields of rice with two popular crop simulation models – CERES-rice and ORYZA1N (Aggarwal et al. 1997) at different levels of management, Aggarwal and Mall (2002) showed that rice yields at current level of management (referred to application of 150 kg N ha^{-1} in 3 split doses and frequent irrigations, a common practice in irrigated rice growing areas in several parts of the country) changed with change in temperature and CO_2. Increase of 1–2°C temperature without any increase in CO_2 resulted in a 3–17% decrease in grain yield in different regions. In general, as the temperatures increased, rice yields in eastern and western India were less affected, moderately affected in north and severely affected in southern India. Although a doubling of CO_2 resulted in 12–21% increases in yield in different regions, the beneficial effect of 450 ppm CO_2 was nullified by an increase of 1.9–2.0°C in northern and eastern regions and by 0.9–1.0°C in southern and western regions. In the improved level of management (analogues to potential production environment), the beneficial effect of 450 ppm CO_2 was nullified by an increase of 1.2–1.7°C in northern and eastern regions and by 0.9–1.0°C in southern and western regions. In another study, Aggarwal (2003) showed that the irrigated wheat and rice yields in north India would not be significantly affected due to direct effect until 2050. When the temperature increases become very large by 2070, the crops will show huge reduction in yields.

Impact of elevated CO_2 and temperature on rice yield in eastern India was simulated by Krishnan et al. (2007), using the ORYZA1 and the INFOCROP-rice models. The crop and weather data from ten different sites viz., Bhubaneswar, Chinsurah, Cuttack, Faizabad, Jabalpur, Jorhat, Kalyani, Pusa, Raipur and Ranchi, which differed significantly in their geographical and climatological factors, were used in these two models. For every 1°C increase in temperature, ORYZA1 and INFOCROP-rice models predicted average yield changes of −7.20 and −6.66%, respectively, at the current level of CO_2 (380 ppm). But, increases in the CO_2 concentration up to 700 ppm led to the average yield increases of about 30.73% by ORYZA1 and 56.37% by INFOCROP-rice, respectively. When temperature was increased by about +4°C above the ambient level, the differences in the responses by the two models became remarkably small. For the GDFL, GISS, and UKMO scenarios, the ORYZA1 predicted the yield changes of −7.63, −9.38 and −15.86%, respectively while the INFOCROP did at −9.02, −11.30 and −21.35% for the corresponding scenarios. There were considerable differences in the yield predictions for individual sites, with declining trend for Cuttack and Bhubaneswar, but an increasing trend for Jorhat. These differences in yield predictions were mainly attributed to the sterility of rice spikelets at higher temperatures (Krishnan and Rao 2005).

Developing the climate change scenarios for the selected regions of the Indian sub-continent using three GCMs namely, Goddard Institute of Space Studies Model (GISS-2, Russell and Rind 1999), Geophysical Fluid Dynamics Laboratory Model (GFDL-R30, Knutson et al. 1999) and United Kingdom Meteorological Office – Hadley Climate Prediction Centre Model (UKMO – HadCM3, Mitchell et al. 1998), Mall et al. (2004) used the CROPGRO-soybean model to simulate the impact of climate change on soybean production in India. The probable changes in surface air temperature during the growing season were estimated at the selected sites in the region, following standard rationalization techniques suggested by IPCC (Mearns et al. 2001). Probable changes in precipitation, cloudiness and solar radiation under the climate changes scenarios were not taken into consideration, in view of the significant uncertainties associated with non-linear, abrupt and threshold rainfall events projected by GCMs over the Indian subcontinent. All the GCM projected climate change scenarios (at the time of doubling of CO_2 concentrations) predicted decreased yields for almost all locations. Mean decline in yields across different scenarios ranged from 14% in Pune (West India) to 23% in Gwaliar (Central India). Decline in soybean yield was found to be less in west and south India as compared to other parts of the country. The mean yield was found to be significantly affected under the UKMO model generated climate scenarios for both current and doubled CO_2 atmosphere. In general, the direct impacts of climate changes would be small on *kharif* crops, but *kharif* agriculture would become vulnerable due to increased incidence of weather extremes, such as change in rainy days, rainfall intensity, duration and frequency of drought and floods, diurnal asymmetry of temperature, change in humidity, and pest incidence and virulence. The *rabi* crop production might become more vulnerable due to larger increase in temperature, asymmetry of day and night temperature and higher uncertainties in rainfall. Apparently, the impacts of the climate change on Indian agriculture would be small in the near future, but in long run, the Indian agriculture may be seriously affected depending upon season, level of management, and magnitude of climate change.

3.7 Advantages of Crop Simulation Studies

Highly sophisticated computer models are fast and cheap in application for estimating the impacts of climate change on crop growth. Nevertheless, the models are still based on results from experiments and may carry their restrictions also to the models. To assess the degree of sustainability of particular agricultural production system in relation to environmental factors as well as to develop adoption measures, there is a need to understand quantitatively the processes determining crop production and how these are influenced by climate characteristics, environmental conditions and management practices. Field studies and long-term experimentations are one way to get this required information. Since the field studies, replicated across locations and years, are laborious and resource consuming, it takes considerable time to generate outputs for use in decision making. Additionally, they are site-specific in nature and variability in environmental conditions makes them difficult to duplicate in other places. On the other hand, crop models offer a cheaper and quicker complimentary approach and can easily evaluate a number of alternative strategies and risks in agricultural decision making. The simulation models can be used to indicate future trends and prescribe appropriate actions, such as suitable crops, best soil and water management practices, and changes in agronomic practices that maximize profit and minimize negative environmental impacts.

3.8 Adaptation Strategies for Crop Production Systems under Changing Climate

Adaptation is an adjustment made within the crop production systems, in order to live successfully with changing climate (Matthews and Wassmann 2003). The probable adaptive responses need not to be new and they can include farm level practices, such as change of planting dates, altered water conservation practices, change to early maturing varieties to mitigate shortened growing season, change to drought tolerant crop varieties, and change to high yielding crop varieties to take advantage of unusually favorable weather. Other adaptation strategies, which can be considered, are application of irrigation and adoption of multiple cropping to take advantage of longer growing seasons. The policy makers require an assessment of the benefits derivable from the adoption of the various adaptation measures. The computation of such benefits would require knowledge of the pre- and the post-adoption yields in addition to the costs of the adaptation itself, in order to make the choice among potential adaptation options. The success in the assessment of impacts and adaptations does not depend on the accuracy of yield predictions as much as it depends on the extent to which the differences between pre impact (adaptation) and post impact (adaptation) yields are reflected. What is more needed for the assessment of impacts of climate variability is the difference between pre- and post-impact productivity and production. The performance of model could be adjudged satisfactory once the model truthfully indicated the differences between pre- and post-adoption yields, not necessarily the actual productivity or production. The adaptive responses could be facilitated by increased knowledge of weather patterns and climate-related variability through the use of climate forecast information.

To abate the shortening effect of temperatures on crop cycles, changed sowing dates and late-maturing genotypes of different crops could be used. But the winter cereals, whose cycle length is linked with cold temperature requirements (vernalization), will not able to avoid the climatic risk during warmer winters. Similarly, the late-maturing crops would face climatic risks in the early summer. The sowing dates of winter crops cannot be postponed to early fall because of much higher probability of experiencing low temperatures at sensitive stages and the cost of fungal disease control during the early fall. For summer crops, using earlier sowing dates or longer-maturing varieties would counteract the detrimental effects of climate change in all cases, as was demonstrated for sunflowers throughout Europe by Claire (1996), for spring wheat in Finland by Saarikko and Carter (1996), and for maize in Spain by Claire (1996). In the northern European regions, selecting adequate sowing dates could help to synchronize full canopy development and maximum radiation availability on maize-type crops (Delécolle et al. 1996), which would enhance final production. Many of the cropping mechanisms, such as shifting planting dates, choosing varieties with different growth duration, or changing crop rotations may result in lower yields. In the Indo-Gangetic plains, delayed planting is already one of the major causes of reduction in crop yields of rice and wheat. The rice-wheat cropping system is the economic backbone of this region and only a small gradual decrease in productivity in either rice or wheat crops will drastically imperil food security. Most studies project decreased yields in non-irrigated wheat and rice, and a loss in farm-level net revenue between 9 and 25% for a temperature increase of 2–3.5°C (Aggarwal and Mall 2002). Although the important mitigation and adaptation strategies required to cope with anticipated climate change impacts generally include adjustment in sowing dates, breeding of plants that are more resilient to variability of climate, and improvement in agronomic practices (Attri and Rathore 2003). The identification of suitable response strategies is key to sustainable agriculture. Likewise, Mall et al. (2004) suggested that delaying the sowing dates would be favorable for increased soybean yields at all the locations in India. Sowing in the second season would also be able to mitigate the detrimental effects of future increases in surface temperature due to global warming at some locations. However, the proposed shift in soybean production from the current main season to a second season may necessitate additional planning and change in management practices. Changing the sowing dates is a no-cost decision that can be taken at the farmer level, but a large shift in sowing dates probably would affect the agrotechnological management of other crops to be grown during the remaining part of the year. Hence, there is a need to identify district or agroclimatic regions vulnerable to climate change and to find out suitable adaptation practices to be followed in order to sustain the productivity of these regions to some extent. It is most likely that at least in the short run, the effects of climatic variability are much larger than the projected impact of global climatic change. Therefore, evolving strategies for managing climatic variability in agricultural production will take care of adaptation required for climatic change (Aggarwal 2003).

3.9 Adjustments in Sowing Date of Rice Crop

Sowing date adjustments are a simple and powerful tool for mitigating the effects of a potential global warming (Baker and Allen 1993). Krishnan et al. (2007) demonstrated the potential outcomes by adjusting the sowing time of rice at two sites (Cuttack and Jorhat) and simulating the crop growth under different climate change scenarios (Fig. 3.1). Under the GCMs scenarios, temperature at the time of flowering for the main season was found to be high, and there were considerable variations when simulated for different climate change scenarios under different sowing dates. Among the different sowing dates tested, the sowing on July 15th at Cuttack led to the yield changes of +6.6, +4.1 and −9.8%, respectively under the GFDL, GISS and UKMO model scenarios. Interestingly, the sowing on July 1st at Jorhat resulted in yield increases at +27.1, +24.3 and +13.4% respectively, for the corresponding scenarios. Any further delay in sowing at both the sites, which had different dates for the maximum response, was not beneficial in terms of crop yield.

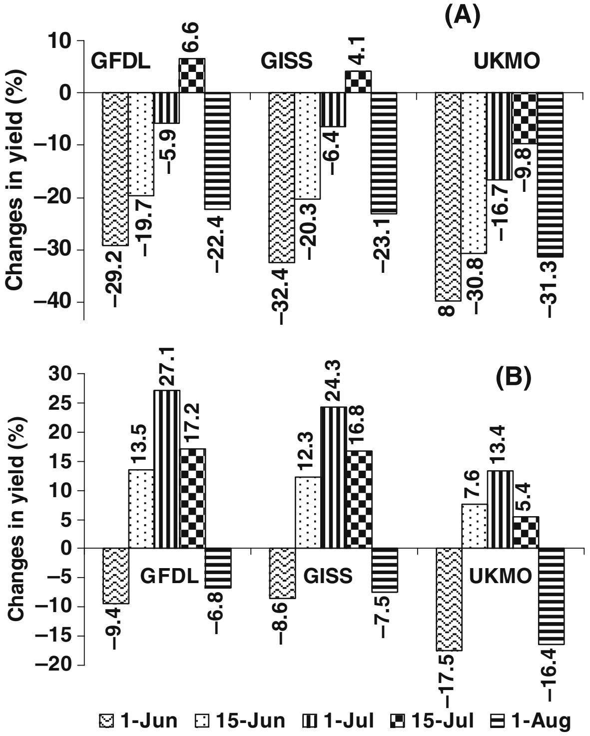

Fig. 3.1 Adaptation to climate change by adjusting the sowing dates under different GCMs scenarios of the rice variety IR 36 grown during the *kharif* season at Cuttack (**A**) and at Jorhat (**B**) using ORYZA1 model (Source: Krishnan et al. (2007))

3.10 Increased Tolerance of Rice Spikelet Fertility to Temperature

Hypothesis that high temperature induces spikelet injury was evaluated by Krishnan et al. (2007) by enhancing the tolerance level of the rice cultivar IR 36 in the crop model. The equation used in the ORYZA1 model to describe the response of spikelet fertility to temperature is

$$\delta = 100/[1 + e^{0.853(\mathrm{Tmax}-\mathrm{Tmp})}]$$

Where δ is the fertility percentage (%), Tmax is the average daily maximum temperature during the flowering period (°C), and Tmp is the average daily maximum temperature (°C) at which 50% of the spikelets are fertile. For the *indica* variety, Tmp had a value of 36.5. To simulate the possible effect of an increase in tolerance of spikelet to high temperatures, it was assumed that this response was shifted by 2°C by increasing the value of Tmp to 38.5°C. This adaptation in the spikelet trait was examined in Cuttack site. With the available weather data for this site, and with a constant sowing date of June 15th, a comparative study using the ORYZA1 model was made for the current climate and other GCM scenarios as obtained by the GFDL, GISS and UKMO (Fig. 3.2). Under the GCMs scenarios, temperature at the time of flowering for the main season was already high. Without any temperature tolerance of the variety by not adjusting the value of Tmp, large decreases in yield due to spikelet sterility were predicted. But, with the adaptation of variety by improved temperature tolerance of the spikelet, the yield increased higher than that of the current scenario level, at about +10.7, +13.6 and −8.4 respectively, under the GFDL, GISS and UKMO model scenarios (Fig. 3.2).

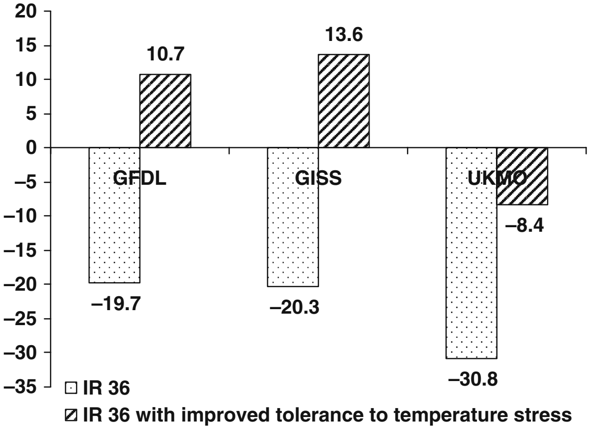

Fig. 3.2 Adaptation to climate change by improving the temperature tolerance of the rice cultivar IR 36 under the different GCMs scenarios grown during the *kharif* season at Cuttack using ORYZA1 model (Source: Krishnan et al. (2007))

3.11 Mitigation Efforts: Use of Simulation Studies

The global climate change is not a new phenomenon and every new anthropogenic driving force can affect the rate of the climate change. The changes, which might be observed over a geologic time period, could happen over a shorter time span, since the start of Industrial Revolution. Apparently, human activities are causing rapid climate change. The concentrations of key anthropogenic greenhouse gases, such as carbon dioxide (CO_2), methane, nitrous oxide and tropospheric ozone, have reached their highest levels, primarily due to the combustion of fossil fuels, agriculture, and land-use changes (Korner and Bazzaz. 1996; Rosenzweig and Hillel 1998). These greenhouse gases stay in the atmosphere for a long time, as the atmospheric life time for these chemicals vary (5–200 years for CO_2, 12 years for CH_4 and 114 years for N_2O). If the current greenhouse gas emission rates continue into the future, agriculture and crop production will face enormous pressure from the stresses caused by these heat trapping gases.

Rice paddy fields are one of the major sources of atmospheric CH_4 and N_2O. There is a need for careful evaluation of the source strength of this ecosystem, and of the influence of soil, water and crop management practices on both grain yield and greenhouse gas fluxes. A number of models have been developed in recent years to predict the rate of CH_4 emission from rice fields, each model having its own strategy or philosophy. Some models tried to use the least number of input parameters and more empirical equations to capture basic pattern of gas fluxes, so that these models could be easily used at the regional or global scale. Several models, such as DNDC (Li et al. 2004), Expert-N, CASA, CENTURY, NLOOS, MERES, MEM and DAYCENT have been developed. Recently, a process-based model INFOCROP (Aggarwal et al. 2006a, b) has been developed for scaling-up gas emission estimates from tropical agriculture. These models can be used to demonstrate a number of mitigation options for reducing methane emissions from rice soil, based on the significant influence of soil texture and pH, moderate influence of organic C content, little effect on short-term seasonal simulations due to the variation in the quantity of aboveground biomass returning to the soil, increasing the length of mid-season aeration, and addition of sulphate fertilizer reducing CH_4 emissions from rice soil.

The climate models indicate that greater warming can occur in the next century; all land areas will warm more rapidly than the global average, particularly at high northern latitudes in the winter season. The projected climate change will have beneficial and adverse effects on both environmental and socio-economic systems, but the larger and more abrupt changes in climate will cause more adverse effects on crop production and thus affecting the regional food security. Hence, there is a continual need to identify and develop mitigation options to minimize the rate of adverse climate changes.

3.12 Crop Models in Decision Making

The major challenge in crop modeling is to develop user friendly and economically viable technology that is readily adoptable by the decision makers as well as farmers. More interactive communication between the model developers and all the stakeholders including extension staff, consultants, farmers, researchers and policy decision makers is required for improving the transferability of models from laboratories to various modes of applications. The crop modeling can be best used as an aid for on-farm decisions. The crop growth simulator, when incorporated with important physiological process and appropriately addressed various physical process, such as water uptake, sunlight etc into one package, would correctly predict growth and yield under varying climate change conditions. For practical use, the farmers require less complicated decision support systems. Hence, any model based on reasoning system, which simplifies the information input and provides a user friendly output format for crop management decisions, such as when to irrigate and how much fertilizer to apply, the extent of land to lease, etc., is very useful to the farmers. But, an extensive consultation with farmers, while testing and validation of the model before it is released for public use, is necessary (Whisler et al. 1986). Every positive feed back from farmers during various stages of model building can help to its successful adoption in various locations and wider use by farmers.

3.13 Natural Resource Management Using Crop Models

The crop simulation models are increasingly available as a part of Decision Supporting Systems (DSS). These DSS can be used to extrapolate results for strategic decision making tasks, such as regional planning, policy analysis and poverty alleviation. The crop systems models have played a useful role in agricultural planning, more commercially in developed countries like USA, France and Australia, and in developing countries as a part of internationally funded projects in the last two decades. Regional or national planning involves analysis of information that covers different crop production systems for making decisions like best land use to meet the specific development goals. One of the successful approaches is coupling of crop simulation models with GIS containing land and water characteristics of a region. In India, Selvarajan et al. (1997) used the ORYZA1 and WTGROW models for analyzing trade offs between water use, farm income, and adoption risks at the district level. Jansen (2001) developed a methodology called SOLUS (Sustainable Options for Land Use) in which crop simulation model MACROS was coupled with a Geographic Information Systems (GIS) and a Linear Program Model (LP) to define crop options and associated management practices in Costa Rica. Using the same approach, Schipper et al. (2001) evaluated the policy issues, such as taxing chemicals to reduce environmental contamination and maintenance of forests through subsidies. The SARP and SYSNET projects of IRRI, involved scientists from India, Malaysia, Vietnam and the Philippines to develop and evaluate methodologies and tools for land use analysis and apply them at regional levels to support agricultural and environmental policy making. The land units of India were defined by agro-ecological zones based on soil and weather characteristics and WTGROW and CERES-RICE models were used to explore possible combinations of integrated farming to achieve the goals of maximizing food, minimizing water use and controlling environmental degradation and soil salinity (Aggarwal 2000).

The researchers and the decision makers have begun to apply the results of crop simulation models to strategic policy analysis. The integration of crop simulation models with GIS and expert systems facilitated it as a useful tool for investment decisions. Beinroth et al. (1998) described the development and use of AEGIS (Agricultural and Environmental GIS) for application with DSSAT type models in land use analysis, as a part of rural development in Columbia. The outputs from crop models were used as inputs to other models of second order effects. Other examples in India (Singh and Thornton 1992) and Putero Rico (Hanson et al. 1999) are also reported. Parry et al. (1985) used CERES models to study the first-order effects of climate on cereal yields. Then, the farm level profitability was investigated as a second order effect of the climate induced yield change by balancing the gross return per unit of production. The implications of changes in crop yields and production for agricultural policy were examined both at the national and international levels. The crop models in combination with spatial analysis tools have the potential to develop rural poverty reduction strategies and evaluate changes in government support programs in different agro-climatic zones. The use of crop simulation models in this way could help to reduce poverty in Kenya (McCown et al. 1994). Due to increasing population pressure, nutrient depletion, soil degradation, low crop yields, and income reduction, were described as a spiraling "poverty trap." The use of a crop simulation model, complimented by a small set of on-farm trials, showed that a use of small amount of fertilizer was an efficient strategy to break the poverty cycle. As the farmers in that area never used fertilizer for crop production, extensive field research conducted past several years did not consider this as an option.

There are many potential uses of crop simulation models to support strategic policy decisions at the regional as well as national level (Anbumozhi et al. 2003). The policy makers and aid agencies like World Bank can greatly benefit from the use of crop simulation models in evaluating the type of interventions, including the conservation of the natural resource base. Some specific areas where the systems analysis will help in this endeavor are (i) to assess the changes in natural resource base because of new policies; (ii) to evaluate advantages and disadvantages of different policy packages, such as changing cropping pattern, shifting production basins, etc; (iii) to analyze farmer responses to policy changes; and (iv) to design new policies based on sustainable rural development by determining acceptable level of trade-off between development and natural resource depletion. Any well informed decision for natural resource management has the potential to reduce rural poverty and for this purpose, crop simulation models will be of high relevance and useful for developmental assistance programs.

3.14 Integration of Climate Prediction and Agricultural Models

The interest in integrating crop simulation models with dynamic seasonal climate forecast models is expanding in response to a perceived opportunity to add value to seasonal climate forecasts for agriculture. Integrated modeling may help to address some obstacles to the effective agricultural use of climate information. Firstly, the modeling can address the mismatch between farmers' needs and available operational forecasts. The probabilistic crop yield forecasts are directly relevant to farmers' livelihood decisions and, at a different scale, to early warning and market applications. Secondly, the credible evidence of livelihood benefits, using integrated climate-crop-economic modeling in a value-of information framework, may assist in the challenge of obtaining institutional, financial and political support; and targeting for greatest benefit. Thirdly, the integrated modelling can reduce the risk and learning time associated with adaptation and adoption, and related uncertainty on the part of advisors and advocates. It can provide insights to advisors and enhance site-specific interpretation of recommendations when driven by spatial data. The model-based "discussion support systems" contribute to learning and farmer–researcher dialogue. The integrated climate–crop modelling may play a genuine, but limited role in efforts to support climate risk management in agriculture, but only if they are used appropriately, with understanding of their capabilities and limitations, and with cautious evaluation of model predictions and of the insights that arise from model-based decision analysis.

3.15 Pertinent Issues to be Considered while Using Simulation Models

While interpreting results from the scenarios predicted by the GCMs, some considerations are necessary. The most significant limitations are their poor resolution, inadequate coupling of atmospheric and oceanic processes, poor simulation of cloud processes and inadequate representation of the biosphere and its feedbacks. The poor resolution is likely to be significant in north-eastern parts of India where the relief is varied and local climate may be quite different from the average across the area used by a GCM. Most GCMs have difficulty in even describing the current climate adequately (Bachelet et al. 1995). The current GCMs are able to predict neither the changes in the variability of the weather nor the frequency of catastrophic events, such as hurricanes, floods or even the intensity of monsoons, all of which can be important in determining crop yields as the average climatic data. The GCMs can, at best, be used to suggest the likely direction and rate of change of future climates.

According to Long et al. (2005), fertilization effect of CO_2 has probably been overestimated. The omission of O_3 effects from the most models could have led to 20% overestimation of crop production in the Northern Hemisphere. The database of chamber studies is the mechanistic basis for crop yield models. Hence, these models overestimate the yield gain due to elevated CO_2 compared to those observed under fully open-air condition (FACE) experiments in the field. The current FACE experiments are, however, not adequate enough to reparameterize the existing models (Long et al. 2005). In a recent study, Bannyayan et al. (2005) evaluated ORYZA 2000 (Bouman and Van Laar 2006) against the observed growth and yield of rice in a 3-year field experiment in Japan where rice plants were subjected to the elevated CO_2 in FACE under varying N fertilizer rates. The simulation results showed that the model overestimated the increases in green leaf area indices due to the elevated CO_2 concentration, but the enhancement of total biomass was only a minor overestimation. While the model was successful in simulating the increase in rice yield due to the CO_2 enrichment, it failed to reproduce the observed interaction with N in the rice yield response to elevated CO_2 concentration. The lack of complete understanding of the effects and the potential interactions of environment variables on plant processes preludes the definitive predictions of the effects of global climate change. Despite the limitations imposed by the assumptions made in both the GCM and the crop simulation models, the models provide significant progress in our understanding of how future climates are likely to affect crop production. Nevertheless, the use of simulation models to predict the likely effects of climate change on crop production is an evolving process.

3.16 Future Research Priorities Using Simulation Studies

Assessment on agriculture and policy response to manage climate change impacts will not be complete unless the biophysical, environmental and socio-economic sectors of agro-ecosystems are studied together. The global integrated impact assessment models provide such a framework, but they are inadequate for regional policy planning. Often, these are not validated at that scale due to their inherent inter-and intra-sectoral conflicts. There is an urgent need to develop the integrated assessment simulation models in which cropping systems; water use and socio-economic parameters are brought together for assessing the impact of climate change in diverse regions of the country. The collaboration with several stakeholders including policy makers, agricultural and environmental scientist, climatologist, economist, administrators, industry and farmers organization, is very much essential. In future studies, unless the uncertainties and limitations discussed above are considered in the crop simulation modeling and climate change scenarios, the assessment of climate change on agriculture cannot provide sound basis for regional policy planning.

3.17 Conclusion

The crop simulation models offer many opportunities and can enhance natural resource management decisions in several ways. At the field level, these models can investigate the long-term changes in the environmental quality of air, soil and water, and yield stability. At the farm level, the applications of the models can include selection of new cropping systems that adapt to micro-climate change, socio-economic viability, and analysis of yield gaps between experimental stations and field production. At the regional level, these models allow the aggregation of crop production responses to various environmental changes.

Many uncertainties exist in modeling studies, partly due to the quality of the predictions by the models, from the use of limited sites for which historical weather are available, due to the quality of the crop simulation models, especially when applied under the rain-fed conditions (Bachelet et al. 1995), and due to the quality of the climate models used to predict future weather scenarios. These uncertainties may be reduced only when a large number of scenarios for different locations are compared and evaluated. Improvement in models has been a continuous process and more scientific understanding is needed to deal with sensitivity of crop production to dynamic changes taking place in the natural resource base. The global, regional and local information sharing can be highly complementary and the information generated from such efforts will serve as a sound basis to make refined models, to develop policy interventions, and to attain food security at the regional as well as the national level.

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Chapter 4
Response of Rice (*Oryza sativa* L.) to Increasing Temperature and Atmospheric CO$_2$

S.V.K. Jagadish and Madan Pal

4.1 Introduction

Rice (*O. sativa* and *O. glaberrima*) is one of the world most important cereal food crops, particularly in Asia and increasingly so in Africa and Latin America. Rice provides a substantial portion of the dietary requirements of nearly 1.6 billion people, with another 400 million relying on rice for quarter to half of their diet (Swaminathan 1984). Rice is cultivated as far north as Manchuria in China (39° 53′N) and far south as New South Wales in Australia (28° 81′S) (Khush 2005), either as an upland (aerobic) or wetland (irrigated, rainfed and deepwater) crop. Upland rice cultivation covers 17 M ha, while wetland rice is cultivated on 131 M ha, contributing about 30 and 70%, respectively, of the total rice production in the world (Dubey 2001). Rice occupies 23% of the total cultivated area under cereals in the world, of which 89% is in Asia (FAO 2003). Hence, Asia produces 523 MT of rice (91% of the world production) (Dubey 2001), on which nearly half of the world's population depend for food and livelihood (Carriger and Vallee 2007). Since the world population is increasing at 1.17% annually, an annual increase in rice production by 0.6–0.9% is required until 2050 (Carriger and Vallee 2007) to meet the anticipated demand. Previously, this demand was met by extending cultivation into marginal lands aided by advancement in irrigation facilities. In future, the reduced availability of water due to ground water depletion and competition for natural resources will render marginal lands unproductive (Young 1998). Hence, more rice will have to be cultivated on less land, with less water and labour (Khush 2005). On a global scale, nearly 60% of the rice is managed under triple cropping, 15% under double cropping and 25% cropped once a year (Matthews et al. 1991). Reduced land availability and intensive cultivation pattern have resulted in an increase in area grown under unfavourable climatic conditions e.g., hot summer seasons.

Anthropogenic activities are major contributors for increasing atmospheric CO$_2$ concentration, from approximately 280 μmol of CO$_2$ mol^{-1} of air to a current level

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of 379 μmol mol^{-1} (IPCC 2007). Annual average increase of 1.9 μmol mol^{-1} of CO_2 emission is recorded from the past 12 years and rapid economic growth of the developing countries is expected to increase CO_2 concentration to 570 μmol mol^{-1} by 2050 (IPCC 2007). Moreover, concomitant increase in other radiatively active gases, such as methane (CH_4), nitrous oxide (NO) and chloro-fluro-carbons (CFCs) will increase the pace of global warming. Considering the above, global surface mean air temperature increase by 2.0–4.5°C by 2100 (with an increased variability over this mean) is predicted by general circulation models (IPCC 2007). Rice, in the tropics, is grown at average temperatures of 28/22°C (Prasad et al. 2006), but with increasing global temperature and shift in cultivation patterns, a sustained increase in rice production will have to be obtained in a much warmer climate in the future. Both temperature and CO_2 are major variables responsible for global climate change, affecting food and fodder production. Therefore, the response of rice to increasing temperature and elevated CO_2 concentration is discussed in this chapter.

4.2 High Temperature and Vegetative Growth of Rice

Rice, like other crops, has an optimum temperature for growth and developmental processes (Table 4.1). Increase or decrease in optimum temperature will result in an altered physiological activity or may lead into a different developmental pathway (Downton and Slatyer 1972). A temperature gradient chamber study with increase in mean and maximum temperature by 2.0–3.6°C and 4.0–7.0°C, respectively, resulted in increased plant height and an earlier maximum tillering (Oh-e et al. 2007). High temperature (ambient +5°C) maintained throughout the crop growth period decreased the leaf photosynthetic rate by 14% (Prasad et al. 2006) and Oh-e et al. (2007), reflecting a decline of 11.2–35.6% in photosynthesis from heading to middle ripening stage. A decrease in spikelet fertility and seed yield was attributed to decline in pollen production and pollen numbers on the stigma, but not due to decrease in photosynthesis (Prasad et al. 2006). Increase in leaf temperature from 20 to 30°C differentiated *O. sativa* sub-spp, resulting in an increase of 18–14% and 100–150% in photosynthesis and respiration, respectively, with the increase being higher in japonica spp. than the indica spp. (Weng and Chen 1987). High temperature (36/30°C) led to an abnormal loss in chlorophyll (Thimann 1987), accelerating senescence in wheat plants (Harding et al. 1990).

Table 4.1 Optimum temperature for different growth and development stages of rice

Trait	Optimum temperature	References
Seed germination	25–35°C	Ueno and Miyoshi (2005)
Rate of leaf emergence	26°C	Ellis et al. (1993)
Days to heading	27–30°C	Horie (1994)
Rate of flowering	30°C	Nakagawa et al. (2005)
Seed-set	29°C	Ziska and Manalo (1996)

Extracted from Jagadish et al. (2007) and Jagadish (2007).

Vegetative stage heat tolerance has been studied by using cellular membrane thermo-stability (CMT) to record the electrolyte leakage, as a measure of heat tolerance (Blum and Ebercon 1981; Reynolds et al. 1994; Fokar et al. 1998). In cowpea (*Vigna unguiculata* L.), a positive association between CMT and heat tolerance at flowering has been found (Ismail and Hall 1999). However, Prasad et al. (2006) found a poor correlation ($r^2 = 0.02$) between heat tolerance during vegetative stage as measured by CMT and reproductive stage tolerance measured by spikelet fertility in 14 rice genotypes. A similar relation has also been found in peanuts (Kakani et al. 2002; Craufurd et al. 2003). Therefore, a measure of heat tolerance by CMT at the vegetative stage and response to heat at reproductive stage in rice could be different and hence cannot be extrapolated.

4.3 High Temperature and Reproductive Development of Rice

Temperature below or above optimum will slow down the developmental rate and increase the time to reach heading stage (Horie 1994). Reproductive stage in rice is more sensitive to heat than the vegetative stage (Yoshida et al. 1981). An independent extreme heat episode during vegetative stage has no influence on reproductive stage (Porter and Semenov 2005).

Rice genotypes can avoid high temperatures during anthesis, by heading during the cooler periods of the season (macro-escape) or by anthesing during cooler hours of early morning (micro-escape). An advancement of peak anthesis towards early hours of the morning (Prasad et al. 2006), is a genuine attempt to escape high temperatures during later hours of the day. Among the two cultivated rice spp. *O. glaberrima* has the ability to anthese during early morning (immediately after dawn) and escape high temperatures during the later hours of the day (Yoshida et al. 1981; Prasad et al. 2006). On the other hand, *O. sativa* spp. flower later during the day (Prasad et al. 2006) experiencing high day temperatures which results in the spikelet sterility. The early morning flowering advantage of *O. glaberrima* has been exploited in interspecific crosses between *O. glaberrima* and *O. sativa* to advance peak flowering towards early hours (Prasad et al. 2006). Alternatively, genotypes anthesing under high temperature and maintaining seed-set exhibit true heat tolerance (*O. sativa* sub-spp. indica cv. N22; Yoshida et al. 1981; Prasad et al. 2006). Anthesis/flowering is the most sensitive process during reproductive stage to high temperature (Nakagawa et al. 2002; Satake and Yoshida 1978), followed by microsporogenesis (Yoshida et al. 1981). High temperatures (35°C) during anthesis and microsporogenesis resulted in 71 and 34% decline in the spikelet fertility, respectively (Yoshida et al. 1981). Heat stress during anthesis leads to an irreversible effect, as there is no increase in panicle dry weight with subsequent improvement in the environment (Oh-e et al. 2007).

Anthesis is highly sensitive to heat as spikelet tissue temperature $>33.7°\text{C}$ for $\leq$ 1 h was sufficient to render a spikelet sterile (Jagadish et al. 2007). Spikelet opening even an hour before showed no effect of the subsequent heat exposure, while those spikelets anthesing within an hour after high temperature exposure were affected (Satake and Yoshida 1978), but not after one hour. The preceding high temperature affected either the anther or pollen in spikelet opening within an hour after high temperature (Matsui et al. 2000). Reciprocal studies with manual shedding of pollen from control plants on to the stigma exposed to high temperature and vice versa showed that the ability of the pistil to be fertilized remained unaffected even over a period of 5 days at 41°C (Yoshida et al. 1981). Similarly, wheat spikelet fertility was increased from 30 to 80% by pollinating heat stressed pistil with unstressed pollen (Saini and Aspinall 1982). Hence, the male reproductive organ is mainly responsible for spikelet sterility under high temperature and hence, to be targeted for increasing tolerance to warmer climates. The male gametophyte was found highly sensitive to high temperature in majority of the cultivated crops (Table 4.2) and the mechanism leading to sterility could be similar to rice under high temperatures.

High temperatures during anthesis could result in spikelet sterility due to the sensitive physiological processes (anther dehiscence, pollination, pollen germination on the stigma, pollen tube growth or the early events of fertilization) being affected. A rice spikelet, on an average, open for 45 min, exposing the sensitive reproductive organs to ambient temperature and relative humidity. Anther dehiscence is the most susceptible process during anthesis under high temperature (Matsui et al. 1999). Earlier, anther dehiscence was perceived to be the result of a simple desiccatory process, which if true, would be promoted by high ambient air temperature.

Table 4.2 Effect of high temperature on reproductive processes in various cultivated crops

Stage	Temperature	Summary of the effects	Crop – references
Microspor-ogeneis	30°C	Anthers with no pollen or less starch accumulation, collapsed, shrivelled, clumped microspores	*Triticum aestivum* (Saini and Aspinall 1982)
	38/30°C		*Glycine max* (Koti et al. 2005)
	33°C		*Capsicum annum* (Erickson and Markhart 2002)
Anthesis	32/27°C	Pollen count and/or pollen viability and/or pollen germination and tube growth reduced	*Phaseolus vulgaris* (Porch and Jahn 2001)
	32/26°C		*Lycopersicon esculentum* (Peet et al. 1998)
	38°C		*Arachis hypogaea* (Craufurd et al. 2003)
	30/20°C		*T. aestivum* (Saini and Aspinall 1982)
	33/30°C		*Vigna unguiculata* (Ahmed et al. 1992)
	38/32°C		*Zea mays* (Herrero and Johnson 1980)

Extracted from Jagadish et al. (2007) and Jagadish (2007).

However, a higher temperature with an increase in vapour pressure deficit will increase evaporation, thereby depriving the crucial moisture needed for pollen grain swelling which is inevitable for anther dehiscence. An increase in relative humidity from 45 to 75% also resulted in reduced pollen count on the stigma due to abnormal anther dehiscence (Matsui et al. 1997b). Genotypic differences in anther characteristics between susceptible and tolerant rice genotypes do exist (Table 4.3). Artificial spikelet opening triggered rapid pollen swelling helping in anther dehiscence and subsequent pollen shedding from apical and basal pores (Matsui et al. 1999). The anther basal pore length is considered to have a significant contribution towards pollination under high temperature (Matsui and Kagata 2003) because of its close proximity to the stigmatic surface. The importance of the apical pore under high temperature is not known, but can assist in pollination.

Dehiscence of the anther leading to pollen deposition on the stigma is called as pollination. It has been accepted that the critical number of pollen grains (Matsui and Kagata 2003) and the number of pollen germinating on the stigma (Satake and Yoshida 1978) for a spikelet to be fertile are 20 and 10, respectively. However, 20% of the spikelets having 10 or more germinated pollen grains on the stigma had 0% fertility at 40°C (Matsui et al. 2001). Moreover, Matsui et al. (1997a) recorded 13% spikelet fertility with spikelets having ≤ 5 germinating pollen, while spikelets having >20 germinating pollen on the stigma showed good agreement with fertility. Therefore, genotypic variation in pollen germination under high temperatures exists and a fixed number of pollen count leading to the assumption of 50% of them germinating and equating them to fertility can be misleading. Developmental processes beyond pollen germination are sensitive to heat and have been also shown in other crops (*Arachis hypogea*: Kakani et al. 2002, 2005; *Glycine max*: Salem et al. 2007).

Table 4.3 Anther characteristics influencing dehiscence in tolerant and susceptible rice genotypes

Tolerant genotypes	Susceptible genotypes	References
Longer anthers	Comparatively shorter anthers	Matsui et al. (2001)
Two cell layers (degrading or degraded tapetum and endothecium cells) separate the locule from the lacuna, allowing for easy anther dehiscence	Three cell layers (degrading tapetum, endothecium cells and parenchyma cells) separate the locule from the lacuna, hindering anther dehiscence	Matsui and Omasa (2002)
Easy and homogeneous anther dehiscence	Abnormal or no anther dehiscence	Matsui et al. (1997b), Yoshida et al. (1981), Satake and Yoshida (1978)
Anthers dehisce within the spikelet on short filaments, shedding more pollen on the stigma	Anthers do not dehisce or they may dehisce outside the spikelet on loose sagging filaments, with less pollen shed on the stigma	Satake and Yoshida (1978)

Extracted from Jagadish et al. (2007) and Jagadish (2007).

After pollination, it takes about 30 min for the pollen tube to reach the embryo sac and fertilization will be completed in 1.5–4 h (Cho 1956). Rice pollen is extremely sensitive to temperature and relative humidity (Matsui et al. 1997b) and looses its viability within 10 min of shedding (Song et al. 2001). Critical high temperature for pollen germination ranges from 41 to 45°C (Enomoto et al. 1956). The reduced pollen germination percentage at 39.5°C was attributed to reduced viability prior to shedding (Matsui et al. 1997b). Both pollen germination and pollen tube elongation are inhibited by high temperature, but there exists genotypic differences for these traits (Enomoto et al. 1956; Yamada 1964; Satake and Yoshida 1978). Poor pollination at 37.5°C and pollen germination at 40°C are considered to be the likely reasons for the spikelet sterility (Matsui et al. 2001). A decline of about 7% in rice yields with per 1°C increase in temperature has been reported in India. This reduction is mainly attributed to spikelet sterility at high temperatures (Krishnan et al. 2007).

High temperature of 39°C (±0.5°C) exhibited significant reduction in pollen activity, pollen germination and floret fertility. However, the tolerant cv. Shanyou 63 showed a much slower reduction in the rates compared to the susceptible cv. Teyou559 (Tang et al. 2008). Furthermore, they quantified the growth hormones in the anthers and found a decrease in IAA (Indole Acetic Acid), GA_3 (Gibberellic acid), free proline and soluble proteins, but a significant increase in ABA (Absisic acid) content. Nine japonica spp. were screened between 35 and 40°C and a 3°C difference in critical temperature was found causing 50% spikelet sterility between the tolerant genotype (40°C; Akitakomachi) compared to the susceptible genotype (37°C; Hinohikari) (Matsui et al. 2001). Working on three rice genotypes, Yoshida et al. (1981) identified cv. N22 to be highly tolerant when exposed to high temperature at flowering. Subsequent reports have identified the heat tolerant trait during flowering in N22 (Prasad et al. 2006; Jagadish et al. 2008). Rice genotypes were screened for spikelet fertility and yield in different species, ecotypes and cultivars under ambient and high temperature (ambient + 5°C) conditions for the whole crop growth period (Prasad et al. 2006). They concluded that heat tolerance exists in both sub-spp. of *O. sativa* and it cannot be generalized that indica sub-spp. are more tolerant than japonicas.

Matsui and his associates exposed the plants to high temperature beginning 10:00 h, while Prasad et al. (2006) recorded the beginning of flowering as early as 09:00 h. Moreover, the quantitative effect or acclimation to high temperature occurring on different days of the flowering period is not known. To minimize these problems in studying the response of rice to heat sensitive anthesis stage, a protocol has been documented to identify the spikelets explicitly exposed to high temperature (Jagadish et al. 2007). Considering the sensitivity of rice during anthesis, they exposed the plants to high temperatures (35 and 38°C) for a single day and identified spikelets receiving 1, 2, 4 or 6 h of high temperature by adopting the marking protocol using acrylic paint. One concern with this approach was the manual interruptions with the spikelets ability to cool its microclimate through transpiration resulting in reduced tissue temperature. This was overcome by marking the spikelets after the heat treatment under control conditions giving all spikelets equal opportunity to efficiently utilize their ability to cool under high temperatures. A difference of

0.4, 1.3 and 1.8°C in spikelet tissue temperatures were recorded using copper constantan thermocouples at ambient temperatures of 30, 35 and 38°C, respectively in temperature controlled growth cabinets (Jagadish et al. 2007). Moreover, the flowering pattern of both indicas and japonicas is known (Yoshida et al. 1981; Prasad et al. 2006) and hence exposing the plants to 6 h (09:00–15:00 h) of high temperature included >95% spikelets flowering during the day and each spikelet would be exposed to at least an hour of high temperature sufficient to test the spikelet for tolerance. From this study, it was evident that high temperature (>33.7°C tissue temperature) for <1 h also affected spikelet fertility. This was reflected in the short duration heat treatments of 1 and 2 h, where spikelets opening 30 min before high temperature exposure would experience next 30 min high temperature apparently enough to reduce spikelet fertility to a certain level (Fig. 4.1), indicating the extreme sensitivity of rice spikelets at anthesis. This marking protocol could also be used to study true cold tolerance during anthesis since the mechanisms leading to sterility at low temperature (Gunawardena et al. 2003) are similar to high temperature (Yoshida et al. 1981).

Identifying heat avoiding genotypes (early morning flowering) could help solve the problem of increasing day temperatures on rice production. However, day/night temperatures and number of warmer nights are predicted to increase in the future (IPCC 2007). Recently, it has been reported that minimum temperature (night) has increased by 1.13°C and the maximum temperature (day) by 0.35°C as evident from 25 years weather data from the International Rice Research Institute, Philippines (Peng et al. 2004). In their study, they showed decreasing yield with increase in minimum temperature. Although the time of occurrence of anthesis and its sensitivity are well known, the degree of heat sensitive processes during panicle initiation

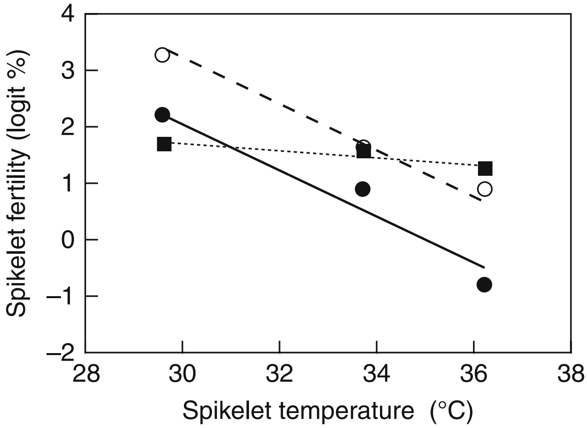

Fig. 4.1 Spikelet fertility in rice cv. IR64 exposed to temperatures of 29.6, 33.7 and 36.2°C for 2 h (*solid circle* and *line*), and seed-set of spikelets that opened at 29.6°C either 1 h before (*open circles* and *hashed line*) or 1 h after (*solid square* and *dotted line*) the 2 h treatment. *Fitted lines*: ○$y = 15.43 - 0.408x$; •$y = 14.30 - 0.408x$; ■$y = 3.56 - 0.062x$. Adapted from Jagadish et al. (2007) and Jagadish (2007)

and microsporogenesis are unknown. Therefore, identifying heat avoiding genotypes will not solve the problem completely and true heat tolerance identification is crucial to mitigate future climatic changes.

Although genotypic difference in critical heat thresholds in rice is documented (Yoshida et al. 1981), the interaction between high temperature and duration of exposure was studied recently (Jagadish 2007). Present crop models have the genotypic difference in thresholds incorporated in them but the possibility of an interaction between temperature and duration of exposure is assumed to be non-significant. Response of rice to high temperature has been modeled using daily mean temperature (Horie et al. 1995; Kropff et al. 1995) or including number of days with maximum temperature >34°C (Challinor et al. 2007) and more recently using daily minimum and maximum temperature (Krishnan et al. 2007). Although the above studies included sub-daily temperatures, response to anthesis is extremely sensitive to sub-daily time course of temperature and flowering models with hourly temperature changes are needed, which can be incorporated into crop models for better prediction. Furthermore, quantification of high temperature impact on future crop yields based on predictions is still in its infancy (Challinor et al. 2007) due to less experimental data available.

Following the marking protocol, an interaction between high temperature and duration of exposure in a heat sensitive genotype (Azucena) but not in a moderately tolerant IR64 was identified (Fig. 4.2). The interactive effect could be included into crop models by adopting the cumulative temperature response above a threshold temperature of 33°C (Nakagawa et al. 2002). A similar response was seen in ground nut and has been quantified by cumulative temperature approach (Vara Prasad et al. 1999). Accumulated temperature or thermal time above a threshold can be calculated by $\mathrm{TT} = (T - 33^\circ\mathrm{C}) \times t$, where T is the day temperature and t, duration of the treatment (Fig. 4.3).

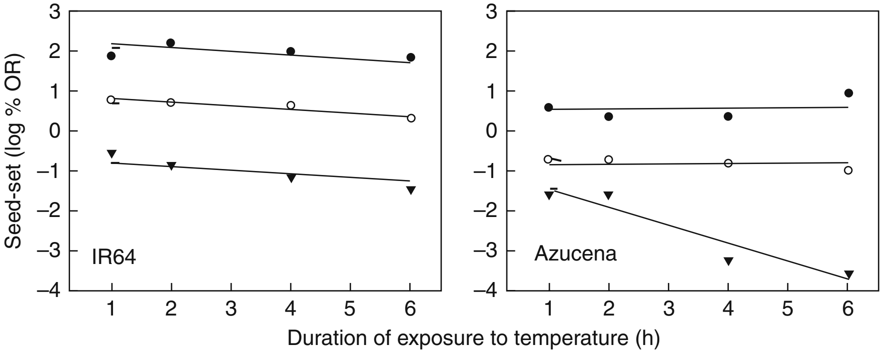

Fig. 4.2 Spikelet fertility of spikelets of IR64 and Azucena exposed to temperatures of 29.6° (●), 33.7° (○) and 36.2°C (▼) for 1, 2, 4 and 6 h. *Fitted lines* are: IR64 ● y=2.29–0.0945x; ○ y=0.92–0.0945x; ▼ y=–0.69–0.0945x; Azucena ● y=0.526+0.0116x; ○ y=–0.849–0.0116x; ▼ y=–1.029–0.4518x. Adapted from Jagadish et al. (2007) and Jagadish (2007)

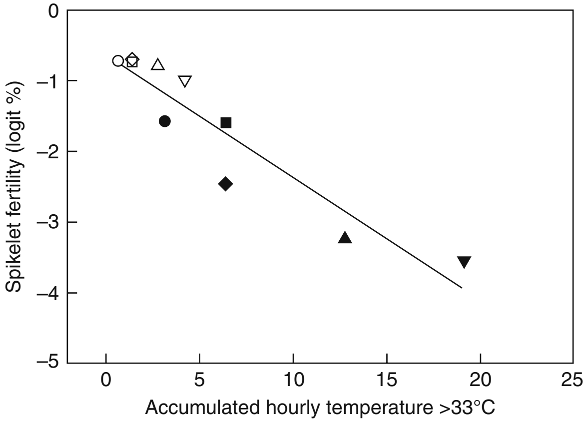

Fig. 4.3 Relation between spikelet fertility and accumulated hourly temperature > 33°C in Azucena. Key: *open symbols* = 33.7°C; *closed symbols* = 36.2°C; 2003 (◆, ◇); 2004 – duration: ○, 1 h; □, 2 h; Δ, 4 h; ∇, 6 h. *Fitted line*: $y=-6.50-1.67x$, $r^2=0.88$. Adapted from Jagadish et al. (2007) and Jagadish (2007)

Almost all previous studies on the response of rice or crops in general were conducted in temperature controlled chambers or temperature gradient tunnels until recently T-FACE (Temperature – Free Air Controlled Enhancement) was introduced. Initially heating systems were devised to heat ecosystems (e.g. soil) to increase soil temperature without increasing air temperature (Shaver et al. 2000; Shen and Harte 2000). To study the effects of global warming in warm open field trials, efficient infrared heater arrays have been tried (Kimball et al. 2008). Positioning the heaters at 45° from the initially tried horizontal position and arranging them in hexagonal array could uniformly warm an area of 3 m^2. To begin with, the infrared heaters were maintained at constant power mode, resulting in excess heating during calm night time and less heating under variable day time condition (Harte and Shaw 1995; Harte et al. 1995). But Nijs et al. (1996) and Kimball (2005) devised controllers to maintain controlled warming system to simulate diurnal pattern of temperature increase during day and night. In future, a much more cost effective heating system could be available for in situ studies related to climate change.

4.4 Effect of Elevated CO$_2$ on Vegetative Growth and Photosynthesis in Rice

CO$_2$ could well be an evil in disguise, as elevated CO$_2$, apart from enhancing photosynthesis in most crop plants (C$_3$), has the ability to trap infra red radiations from earth surface and redirect them back to increase global mean temperature. Rice, being a C$_3$ plant, responds positively to increase in atmospheric CO$_2$. Several long and short-term CO$_2$ enrichment studies show that doubling the current ambient CO$_2$ increased leaf photosynthetic rate by 30–70% (Ghilidyal and Natu 2000). The magnitude of this increase depends on rice cultivar, growth stage and environment (Lin et al. 1997). Rice responses to CO$_2$ are quite different for plants grown under isolated conditions (e.g. in pots) compared to those under field conditions (Nakagawa et al. 1994).

Horie (1993) reported large differences in CO_2 enriched treatments on biomass production in pot experiments compared to field like environments. Horie et al. (2000) conducted a similar analysis and revealed that CO_2 effects on biomass production for rice grown in pots showed a strong temperature dependency, while under field-like conditions, temperature had a minor effect. The pot-grown plants displayed a strong CO_2 – temperature interaction because these plants continue to produce tillers for a longer time than the field-grown rice, especially at higher temperatures. This is attributed to reduced mutual shading in potted plants, while plants grown under field-like conditions develop a closed canopy and so mutual shading suppresses positive feedback effects of increased photosynthesis on growth.

Since one of the major effects of elevated CO_2 on net photosynthesis is suppression of photorespiration, it could be expected that optimum temperature for photosynthesis could shift upwards. The effect of CO_2 enrichment on rice leaf area development is small (Nakagawa et al. 1993; Kim et al. 1996a; Ziska et al. 1997), but the canopy photosynthetic response is mainly attributed to responses at the unit leaf area level. The response of leaf photosynthetic rate to intercellular CO_2 was similar for rice plants grown at different CO_2 and temperature regimes under field-like conditions (Lin et al. 1997). These reports indicate that photosynthetic acclimation to elevated CO_2 is not likely to occur in rice grown under field conditions within double the current CO_2 concentration. However, photosynthetic acclimation to enriched CO_2 could occur either at much higher CO_2 concentration (above $1000\,\mu l\,mol^{-1}$) or from reduced sink size caused by restricted root growth (Arp 1991) or by high temperature induced spikelet sterility (Lin et al. 1997).

Rice cv. IR-30 was subjected to a wide range of CO_2 [sub-ambient (160 and $250\,\mu l\,mol^{-1}$), ambient ($330\,\mu l\,mol^{-1}$) and super-ambient (500, 660 and $900\,\mu l\,mol^{-1}$)] and tested for acclimation by comparing canopy photosynthesis at short and long-term with different levels of CO_2 concentrations (Baker et al. 1990a). When all long-term CO_2 treatments were compared to common short-term CO_2 of $160\,\mu l\,mol^{-1}$, canopy photosynthesis was decreased by 44% across the long-term CO_2 treatments from 160 to $900\,\mu l\,mol^{-1}$. This decline in photosynthesis was accompanied by a 32% decrease in Rubisco (Ribulose 1,5 Biphosphate Carboxylase) relative to other soluble protein with a 66% decrease in Rubisco activity (Rowland-Bamford et al. 1991). Similarly, Vu et al. (1998) found reductions in leaf Rubisco content ranging from 6 to 22% for the CO_2 enriched treatments compared with ambient controls. Baker et al. (1990b) tested rice cv. IR-72 to short-term CO_2 ranging from 160 to $1000\,\mu l\,mol^{-1}$ and found no photosynthetic down-regulation. Thus, while photosynthetic acclimation responses in terms of enzyme down-regulation may be detected at the single leaf biochemical level, these effects may or may not result in a detectable loss of canopy photosynthetic capacity when measured using gas-exchange techniques. Respiration leads to loss of carbon fixed during photosynthesis. Increase in respiration with higher carbohydrates at elevated CO_2 will be seen due to strong relationship between the two (Farrar 1985). The maintenance respiration increases with increase in growth and the positive relation between photosynthesis and respiration is nearly non-existent at senescence (Poorter et al. 1988). Therefore, the enrichment in canopy photosynthesis and respiration differs with plant growth and hence CO_2 budgetting throughout crop cycle is essential (Sasaki et al. 2005).

Elevated CO_2 increases phenological developmental rate and hence reduces the number of days to heading (Baker et al. 1990a; Nakagawa et al. 1993; Kim et al. 1996a). Baker et al. (1990a) found that elevated CO_2 did not significantly affect the interval between similar growth stages of successive leaves (phyllochron) on the main-culm of rice, but the leaf number was reduced due to accelerated phenological development. Reports on the effects of CO_2 on tiller production appear contradictory. Baker et al. (1990a, 1992b) found relatively minor effects of CO_2 on tiller production across CO_2 range of 330–660 $\mu l\,mol^{-1}$, with a density of 235 plants m^{-2}. In contrast, Kim et al. (1996a) and Ziska et al. (1997) found a marked increase in tiller number caused by a doubling of CO_2 at plant densities of 50 and 75 m^{-2}, respectively. This discrepancy is possibly related to the differences in plant densities. A high degree of mutual shading among plants at higher density would have increased competition for light, suppressing the development of tiller primordia (Baker et al. 1990a, 1992b). Therefore, under planting densities usually practised in Asian rice cultures, elevated CO_2 may substantially promote tiller production (Ziska et al. 1997).

To overcome limitations of controlled chamber studies, FACE (Free Air CO_2 Enrichment) technology was developed (McLeod and Long 1999; Hendrey 1993; Long et al. 2006) for in situ study of elevated CO_2 on crop plants. Rice grown under FACE (Ambient + 200 μ mol of CO_2 mol^{-1} of air) accumulated more biomass during the vegetative stage which increased partitioning of photosynthates to the ear. They concluded that with no significant increase in canopy photosynthesis during grain filling stage, the advantage obtained by CO_2 enhancement during vegetative stage is lost during grain filling period (Sasaki et al. 2005). Long et al. (2006) questioned the possible effects of elevated CO_2 in non FACE experiments and concluded that the advantage was roughly half compared to in situ FACE experiments and suggested a downward revisions of the estimated world food supply by the end of this century. However, another group (Tubiello et al. 2007) attributed the conclusions of Long et al. (2006) to technical inconsistencies and lack of statistical significance and concluded that the response of elevated CO_2 was similar across FACE and non FACE experiments as revealed from previous comprehensive modelling and experimental analysis.

4.5 Effect of Elevated CO_2 on Reproductive Development and Yield in Rice

Increased biomass production due to elevated CO_2 has the potential to increase yield, provided flowering and grain-filling are not disrupted by environmental stresses, such as drought or high temperature. In rice, increase in grain yield can be associated with various components like tiller number per ground area, increased panicle weight at maturity, seed fill and individual grain weight. Two rice cultivars (cv N22 and IR72) at elevated CO_2 recorded an increase in grain yield at ambient temperature and the increase was associated to increase in tiller number and panicle weight, but cv NPT-2 had no effect on either tiller number or panicle weight. This shows that tiller formation may be a factor in optimizing the response of rice to increasing CO_2 concentration (Moya et al. 1998). Elevated CO_2 markedly increased rice spikelet number per unit area over a wide range of air temperatures, through increase in the number of productive tillers per unit area and spikelets per tiller (Imai et al. 1985; Kim et al. 1996b). In rice, the spikelet number per unit area is generally proportional to plant nitrogen content (Hasegawa et al. 1994) at the spikelet initiation stage. The fact that elevated CO_2 increased the spikelet number despite a reduction in plant N content suggests that elevated CO_2 promotes better nitrogen use efficiency.

Moya et al. (1998) reported significant enhancement in total plant biomass of rice (cv. IR 72) with increased CO_2 concentration, but grain yield responded to a lesser extent than biomass. Ziska et al. (1997) obtained a yield increase of 27% with CO_2 enrichment in cv. IR72 grown at ambient temperatures in both wet and dry seasons at the International Rice Research Institute, Philippines. The percentage yield increase due to a doubling of CO_2 for a japonica cv. Akihikari, grown at ambient temperatures in Kyoto, Japan, ranged from 20 to 40% over two consecutive years (Kim et al. 1996b). While there is considerable variation among these reports in the relative yield response to doubled CO_2, it appears that a 30% enhancement in yield may be a reasonable estimate for rice exposed to long-term doubled CO_2 concentration under field conditions with moderate temperatures.

4.6 Interaction Between Increasing Temperature and Elevated CO_2 on Rice

Previously, an increase in atmospheric CO_2 resulting in temperature increase by 2.5°C was recorded (Gutowski et al. 1998; Cohen 1990) and the same has been convincingly reported (increase by 2.0–4.5°C) in the present IPCC report (IPCC 2007). Hence, examining the interaction of CO_2 and temperature is the need of the hour, as their concomitant occurrence is most likely in future climates. Several studies have examined the impact of increasing carbon dioxide concentration and ambient air temperature on rice growth and yield in controlled environment or field conditions (Baker et al. 1992b; Baker and Allen 1993; Ziska and Teramura 1992). Elevated CO_2 effect on rice developmental rate leading to decrease in days to heading has been shown to be temperature dependent. Akihikari (*japonica*), headed earlier by 6 and 11% at ambient temperatures of 28 and 30°C, respectively (Kim et al. 1996b).

Response of rice with respect to biomass and yield under CO_2 and/or temperature studies could vary considerably under field condition and in growth chamber studies (Ziska et al. 1996). Studies show much larger stimulation (70%) in growth chambers compared to field experiments (10–30%) (Baker et al. 1990a, 1992b; Horie 1993). Although cv. IAC 165 and N22 were recommended for elevated CO_2 conditions, but none performed convincingly for sustained increase in production under concomitant increase in CO_2 and temperature. Similarly, Moya et al. (1998), after studying long term responses, concluded that cv. N22 recorded highest biomass and yield under elevated CO_2 concentration, but did not maintain the same under combined CO_2 and temperature increase. Matsui et al. (1997a), studying the interaction of CO_2 and temperature at reproductive stage, recorded an increase in canopy temperature due to closing of the stomata at high CO_2 concentrations, resulting in low transpiration cooling.

Photosynthetic response of rice may vary under different regimes of air temperature. Nakagawa et al. (1997) found that higher temperature stimulated single-leaf photosynthesis of rice subjected to long-term CO_2 treatments during the vegetative stages. In contrast, Baker and Allen (1993) reported that rice canopy photosynthesis was relatively unaffected by a range of air temperatures. Biochemically, increase in CO_2 concentration stimulates an increase in Rubisco and photorespiration is partially inhibited reducing carbon loss. Hence, increasing temperature should result in higher net photosynthesis and CO_2 uptake as seen in single leaf (Potvin 1994). The interaction of CO_2 and temperature at both vegetative and reproductive stages has to be further explored to exploit the increasing CO_2 for increasing yields.

Rice grains are a significant sink for the assimilates and removal or restriction of this carbon sink will fail to exploit the elevated CO_2 due to photosynthetic insensitivity (Stitt 1991; Webber et al. 1994). Accordingly, Ziska et al. (1996) recorded a significant increase in root/shoot ratio with elevated CO_2 with increasing temperature and hinted at alternative sinks becoming active recipients with reduced carbon sink capacity of the grains due to spikelet sterility from high temperature exposure. Increasing temperatures from 28/21 to 37/30°C decreased grain yield significantly even under 660 μ mol of CO_2 mol^{-1} of air (Baker et al. 1992a). Ziska et al. (1996) recorded 70 and 22% increase in biomass at elevated CO_2 treatment under 29/21 and 37/29°C, respectively, while grain yield of 17 contrasting cultivars recorded $<1\%$ filled spikelets.

4.7 Conclusion

There is sufficient information available on the beneficial effects of elevated CO_2 during the vegetative stage and the antagonistic effect of high temperature during reproductive stage. However, experimental data on the interactions between CO_2 and temperature at critical growth and developmental stages is limited. Therefore, future studies should be concentrated on interaction studies to harness the beneficial effect of elevated CO_2 and to minimize the deleterious effects of increasing temperature. This can be achieved by developing varieties which exhibit heat escape (flower early in the morning) or breeding varieties having heat tolerance during sensitive stages (panicle initiation, microsporogenesis and anthesis). With increasing purchasing power of people in the developing countries, nutritional aspects will be a priority in coming years and should be given importance in future breeding programme. Robust information collected and knowledge gained from environmentally controlled studies could be extended for in situ trials to study the crop response to elevated CO_2 under the FACE system and increasing temperature under T-FACE system which, in future, could be used for interaction studies between temperature and CO_2.

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Chapter 5
Carbon Sequestration and Greenhouse Gas Fluxes from Cropland Soils – Climate Opportunities and Threats

Pete Falloon, Pete Smith, Richard Betts, Chris D. Jones, Jo Smith, Deborah Hemming and Andy Challinor

5.1 Introduction

Globally, soils contain approximately 1500 Pg (1 Pg $=$ 1 Gt $=$ 10^{15} g) of organic carbon (C) (Batjes 1996), roughly three times the amount of carbon in vegetation and twice the amount in the atmosphere (IPCC 2001). The annual fluxes of carbon dioxide (CO_2) from atmosphere to land (global Net Primary Productivity [NPP]) and land to atmosphere (respiration and fire) are of the order of 60 Pg C yr^{-1} (IPCC 2001). during 1990s, fossil fuel combustion and cement production emitted 6.3 ± 1.3 Pg C yr^{-1} to the atmosphere, while land-use change accounted for 1.6 ± 0.8 Pg C yr^{-1} (Schimel et al. 2001; IPCC 2001). Atmospheric C increased at a rate of 3.2 ± 0.1 Pg C yr^{-1}, the oceans absorbed 2.3 ± 0.8 Pg C yr^{-1} and there was an estimated terrestrial sink of 2.3 ± 1.3 Pg C yr^{-1} (Schimel et al. 2001; IPCC 2001). The amount of carbon stored in soils globally is, therefore, very large compared to gross and net annual fluxes of carbon to and from the terrestrial biosphere, and the pools of carbon in the atmosphere and vegetation. Human intervention, via cultivation and disturbance, has also decreased the soil carbon pools relative to the store typically achieved under native vegetation. Historically, these processes have caused a loss of soil C between 40 and 90 Pg C globally (Paustian et al. 1998; Houghton et al. 1999; Lal 1999). Hence, increasing the size of the global soil carbon pool by even a small proportion has the potential to sequester large amounts of carbon, and thus help mitigate climate change.

The trace gases, like methane (CH_4) and nitrous oxide (N_2O), are also potent greenhouse gases and emitted from, and absorbed by soils. For this reason, soils also have a second role to play – reducing trace gas emissions to the atmosphere – in combating climate change. Nitrous oxide is formed primarily from nitrification

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and denitrification processes and is a by-product of nitrification and an intermediate during denitrification. N_2O fluxes from agricultural soils (0.53 Pg C equivalents yr^{-1}) account for more than 50% of the global anthropogenic N_2O flux (Robertson 2004). The majority of the CH_4 flux from agriculture is associated with livestock systems, and arises from enteric fermentation in ruminants (Robertson 2004). The only significant soil source of methane in croplands is rice cultivation (0.25 Pg C equivalents yr^{-1}), which accounts for 22% of agricultural emissions or 12% of total anthropogenic fluxes (Robertson 2004). There are also significant fluxes of N_2O and methane from natural ecosystems. Since carbon storage and trace gas fluxes from natural ecosystems offer less greenhouse mitigation potential (with some exceptions – e.g. avoided deforestation) and are harder to manage, we focus mainly on the role of cropland soils in this chapter.

Climate change could also alter carbon storage and trace gas fluxes from cropland soils, since changes in temperature, precipitation and atmospheric CO_2 concentration will affect net primary production (NPP), carbon/nitrogen inputs to soil, soil carbon decomposition rates and soil nitrogen cycling. Due to the large size of the soil carbon and nitrogen pools, they have considerable potential to drive large positive climate feedbacks, as increased atmospheric CO_2, N_2O and CH_4 concentrations will enhance climate change (Cox et al. 2000; Friedlingstein et al. 2003, 2006; Jones et al. 2003). Therefore, climate change could also act to reduce or alter soil carbon sequestration and trace gas mitigation actions in croplands, with associated impacts on soil quality and fertility. This chapter deals with:

(a) opportunities for greenhouse gas mitigation in cropland soils,
(b) the possible impacts of climate change on greenhouse gas fluxes from cropland soils,
(c) how climate change could impact future cropland greenhouse gas mitigation potential, and
(d) outline future challenges for scientists and policymakers.

5.2 Opportunities for Greenhouse Gas Mitigation in Cropland Soils

5.2.1 Mechanisms of Greenhouse Gas Mitigation in Cropland Soils

Cropland soils can act as both sources and sinks for carbon dioxide and other greenhouse gases. Whether soils act as a sink or source, but their sink/source strength depends critically on soil management (Janssens et al. 2003). Cropland soils emit carbon dioxide (CO_2; through soil and root respiration/decomposition of soil organic matter), nitrous oxide (N_2O; formed during nitrification and denitrification processes) and may either emit or remove (oxidise) methane (CH_4) from the atmosphere. The greenhouse gas mitigation potential for cropland soils, therefore, depends on reducing these emissions, increasing carbon inputs or decreasing disturbance. This could be possible by alteration in greenhouse gas emissions – by manipulating the factors which control the inputs and outputs of the cropland soil carbon and nitrogen cycle.

CO_2 mitigation involves either reducing the CO_2 efflux from the soil or sequestering carbon in the soil. The major inputs to the soil carbon system are the amount and quality of plants (and any additional) carbon input, which, in croplands, are controlled by NPP (crop type and varieties) and management of organic residues. Losses of soil carbon via decomposition are mainly controlled by soil texture, moisture, temperature and oxygen availability, which (with the exception of texture) can also be modified by land management strategies. For instance, mulching and tillage can alter soil moisture and temperature regimes. Soil carbon sequestration can thus be achieved by either (a) increasing the net flux of carbon from the atmosphere to the terrestrial biosphere by increasing global NPP (thus increasing carbon inputs to the soil), (b) storing a larger proportion of the carbon from NPP in the longer-term carbon pools in the soil (e.g. altering root to shoot ratios), (c) adding additional C containing materials to the soil (such as manures or cereal straw), or (d) slowing decomposition. For soil carbon sinks, the best options are, therefore, to increase C stocks in soils that have been depleted in carbon, i.e. cropland soils and degraded soils (Lal 2004; Smith 2004a), since the capacity for increasing C storage is very high in these soils. It is also important to minimise further losses of soil carbon stocks by more judicious land management, for example avoiding land degradation and the drainage of peatlands (Bellamy et al. 2005).

N_2O mitigation entails reducing N_2O emissions. The main options for N_2O mitigation in cropland soils involve altering organic and inorganic fertiliser applications, agricultural operations (especially tillage and compaction), the use of rotations and water management. The impact of these options is to either (a) reduce inputs of nitrogen (and thus reduce N_2O emissions) or (b) minimise conditions suitable for N_2O production (Smith et al. 2004).

For CH_4, mitigation involves reducing CH_4 emissions from rice paddy soils – (Guo and Zhou 2007) and maximising the methane oxidation potential of other soils. The greatest potential for CH_4 mitigation from agricultural soils is in rice production. Rice crop management to reduce CH_4 emissions includes yield improvement by well-managements, as high yield rice crops have significantly lower CH_4 emissions (where more C is allocated to grain than to the rhizosphere where it can undergo methanogenesis). Other mitigation options in rice include residue management and irrigation scheduling (Robertson 2004).

5.2.2 Greenhouse Gas Mitigation Potential of Cropland Management Options

Cropland greenhouse gas mitigation options and indicative C sequestration rates are given in Table 5.1. Clearly, land management practices involving the greatest increases in C inputs to soils have the greatest C mitigation potential achievable

Table 5.1 Greenhouse gas mitigation potential of cropland management options

Management option	Indicative maximum soil C sequestration rate (t C ha^{-1} yr^{-1})	Likely uncertainty	Additional C savings	Impact on trace gas fluxes	Other impacts	Notes
Mechanical operations						
Zero/reduced tillage	0.38 (0.29)	>50%	0.02 t C ha^{-1} yr^{-1} overall reduced C costs (Increased C cost from extra herbicides, reduced fossil-fuel C costs via less farm machinery work)	Unclear	Reduced soil erosion, improved soil fauna	Interacts with soil physical conditions; Ploughing or deep ploughing may reduce trace gas emissions but decrease carbon sequestration; Conservation (reduced) tillage may be intermediate; no-till may increase nitrous oxide emissions
Reduced compaction	Impact uncertain	Impact uncertain		May reduce N_2O	Reduced surface runoff, improved root penetration	
General land use/management						
Set-aside/ Conservation Reserve Program	<0.38	>>50%		May reduce N_2O	Biodiversity and amenity benefits	Impacts on crop production
Convert to permanent crops	0.62	>>50%		May reduce N_2O		
Convert to deep-rooting crops	0.62	>>50%		May reduce N_2O		
Rotational changes	>0	Very high		May reduce N_2O		Catch crops reduce bare soils (possible link to BNF), Amelioration crops (crop type i.e. deep rooting or shallow rooting)

Table 5.1 (continued)

Management option	Indicative maximum soil C sequestration rate (t C ha^{-1} yr^{-1})	Likely uncertainty	Additional C savings	Impact on trace gas fluxes	Other impacts	Notes
Bioenergy crops	0.62	>>50%	6.0 t C ha^{-1} yr^{-1} in above ground biomass, reduced fossil fuel usage	May increase N_2O	Some biodiversity benefits	May increase nitrous oxide fluxes
Extensification/de-intensification	0.54	>>50%		May reduce N_2O		Extensive cropping may be better than intensive for N_2O fluxes – further research needed for to assess this difference and the effect of the arable/livestock ratio. Also compare to intensive areas with other land use on spare land
Organic farming	0–0.54	>>50%				
Convert to grassland	1.2–1.69(1.92)	>>50%				
Convert to woodland	0.62		2.8 t C ha^{-1} yr^{-1} in above-ground biomass		Biodiversity and amenity benefits	
Management to reduce wind and water erosion	Impact uncertain	Impact uncertain			Reduced surface runoff and water pollution	
Increased field margins	0.46–0.52	>>50%	0.2–2.8 t C ha^{-1} yr^{-1} in above ground biomass		Reduced surface runoff, water pollution, biodiversity and amenity benefits	
Biological N fixation	Impact uncertain	Impact uncertain		May reduce N_2O	Improved soil fertility?	Avoid mono cropping, nitrous oxide flux dependent on legumes/grass mixture ratio and is mostly after ploughing; net effect is unknown

Table 5.1 (continued)

Management option	Indicative maximum soil C sequestration rate (t C ha^{-1} yr^{-1})	Likely uncertainty	Additional C savings	Impact on trace gas fluxes	Other impacts	Notes
Organic fertilisers and amendments						Timing (less critical than inorganic); Application techniques; Storage, processing, and handling (temperature, duration, capacity, cover, etc.); Amount/rate of application (apply at recommended rate to coincide with crop demand)
Improve efficiency of animal manure use	0.38 (1.47)	>>50%		May increase or reduce N_2O	Improved soil fertility, increased N & P leaching?	Nitrous oxide flux dependent on degree of maturity/degradation, whether fermented, moisture content
Improve efficiency of crop residue (cereal straw) use	0.69 (0.21)	>>50%		May increase or reduce N_2O	Improved soil fertility, increased N & P leaching?	Important for C sequestration; Quality and size (C:N ratio, total N), Quantity (weight); Application/incorporation techniques (note the interaction with tillage; priming effect on soil N_2O flux mainly with incorporation) avoid wet conditions
Agricultural use of sewage sludge, liquid organic fertilisers and slurries	0.26	>>50%		May increase or reduce N_2O	Improved soil fertility, increased N & P leaching?	Give higher nitrous oxide fluxes than dried material., avoid spreading with mineral fertiliser

Table 5.1 (continued)

Management option	Indicative maximum soil C sequestration rate (t C ha^{-1} yr^{-1})	Likely uncertainty	Additional C savings	Impact on trace gas fluxes	Other impacts	Notes
Application of other wastes to land (compost, industrial, household, biogas residue)	0.38	>>50%			Improved soil fertility, increased N & P leaching?	
Inorganic fertilizer use	0	Very high		May increase or reduce N_2O	Improved soil fertility, increased N & P leaching?	Suitable type and characteristics (e.g. ammonium providing fertiliser vs. nitrate fertiliser, slow-release, inhibitors); Application techniques (synchronisation/timing e.g. split-application to coincide with crop demand, placement, burial may reduce flux); Amount/rate of application, appropriate fertiliser recommendations required
Irrigation and water management	0	Very high		May increase or reduce N_2O		Appropriate timing and size of machinery required

per unit land area, for example, large inputs of organic amendments or conversion to grassland or woodland. However, there are other impacts of different options which also need to be considered, such as biodiversity and soil erosion benefits. The options presented in Table 5.1 are not all mutually exclusive – options for the management of both productive and surplus cropland can be combined to maximise the C mitigation potential (Smith et al. 2000).

Robertson (2004) emphasises the needs for a systems approach for assessing GHG mitigation potential in agriculture. For example, increasing soil carbon stocks in the soil through reduced tillage can lead to anaerobic zones in some soils and thereby increase N_2O emissions (MacKenzie et al. 1998; Smith et al. 2001; Li et al. 2005; Six et al. 2004). Similarly, management to reduce CH_4 emissions in paddy rice fields might increase N_2O emissions. Trade-offs between the GHGs are complex (Robertson et al. 2000), but to be considered. Wherever possible, we have indicated any likely trace gas trade-offs in Table 5.1, although the magnitude and direction of trace gas trade-offs are often uncertain. For instance, altering inorganic fertiliser applications could either increase or reduce N_2O emissions depending on the timing and amounts applied. The different GHGs also have different impacts – for instance, since CO_2has direct physiological effects on plant transpiration via its influence on stomatal conductance, it has the potential to alter run-off in addition to altering climate directly (Betts et al. 2007a).

5.2.3 Climate Mitigation Opportunities in Global Croplands

Cropland soils can be a large source of carbon dioxide (Janssens et al. 2003). Hence, there is a significant potential to reduce the efflux of carbon from agricultural soils, and to sequester carbon in them. Estimates of the potential for additional soil carbon sequestration vary widely. The most recent global estimate is that of Lal (2004a) 0.9 $\pm$ 0.3 Pg C yr^{-1}, or approximately 11% of the 1990's annual anthropogenic CO_2 flux (fossil fuel combustion, cement production and land use change). Over 50 years, this level of C sequestration would restore a large part of the carbon lost from soils historically. However, soil carbon sequestration rates have a limited duration and cannot be maintained indefinitely – in cool temperate climates, soil carbon changes often reach a new equilibrium 50–100 years, following a land use change (Poulton et al. 2003), after which no additional sequestration may be achieved. Geographically, the greatest potential for GHG mitigation on a unit land area basis is in the Cool Moist and Warm Moist climate zones, while in the Cool and Warm dry zones, the mitigation potential is generally much lower (Smith et al. 2007b).

Estimating mitigation potential is often confounded by the choice of constraints. Some authors quote biological potentials (Metting et al. 1999), while others quote potentials as limited by available land or resources (Smith et al. 2000), and many others consider economic and social constraints (Cannell 2003; Freibauer et al. 2004). Smith (2004a) provided a figure showing how these mitigation potential estimates differ and how the potential is reduced by a number of constraints

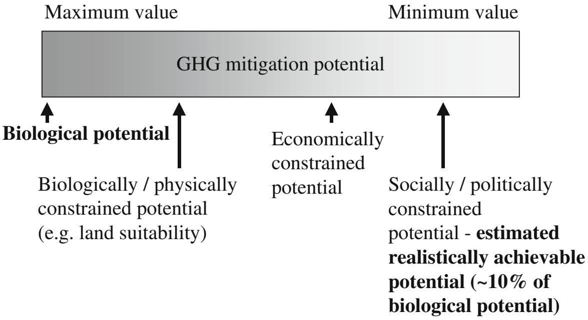

Fig. 5.1 How different constraints reduce the GHG mitigation potential from its theoretical biological maximum to realistically achievable potentials that are much lower (adapted from Smith 2004a)

(Fig. 5.1). An analysis of the estimates presented by Freibauer et al. (2004) and the assumptions used by Cannell (2003) suggest that the realistic sustainable (or conservative) achievable potential of GHG mitigation (taking into account limitations in land use, resources, economics, and social and political factors) may be only about 10–20% of the biological potential. Although this value is derived predominantly from expert judgment, it may be useful in assessing how different estimates of GHG mitigation potential can be compared and how they might realistically contribute to GHG stabilization. This value did not, however, consider how future changes in climate might impact GHG mitigation potential. More recent sectoral and multi-sectoral economic assessments allow the mitigation potential for each activity to be quantified more precisely (Smith et al. 2007a, b). These studies show that the implementation of different measures depends upon the carbon price and measures are implemented to different extents at differing carbon price (Smith et al. 2007a, b).

Smith (2004b) calculated how future carbon emissions and CO_2 stabilization targets might influence the relevance of soil carbon sequestration as a GHG mitigation measure. The IPCC standard reference emission scenarios (SRES) provide estimates of possible emissions under a range of different possible futures (IPCC 2000), which depend upon the degree to which greenhouse gas mitigation policies become global and upon whether environmental or economic concerns take precedence over the next century. In all of these scenarios, the global population will grow, the population will become wealthier and per-capita energy demand will increase over the next century (IPCC 2000), but the extent of these changes differs between scenarios. For each of the scenarios, carbon emission trajectories have been determined (IPCC 2000). Annual carbon emissions ($\mathrm{Pg\,C\,yr^{-1}}$) by 2100 would be A1FI$\sim$30, A1B$\sim$17, A1T$\sim$7, A2$\sim$28, B1$\sim$6, B2$\sim$18.

Emissions trajectories can also be calculated for a range of atmospheric CO_2 stabilization targets (e.g. 450, 550, 650, 750 ppm). For each stabilization target, the allowed carbon emission trajectories, which cannot be exceeded if the target is to be reached, can be calculated. The difference between the allowed emission trajectory for stabilisation at a given target concentration, and the emissions associated with the estimated global energy demand are the carbon emission gaps. For a stabilization target of 550 ppm, carbon emission gaps for each scenario by 2100 ($\mathrm{Pg\,C\,yr^{-1}}$) are

A1FI = 25, A1B = 12, A1T = 2, A2 = 22, B1 = 1, B2 = 13 (IPCC 2001); however, climate-carbon cycle feedbacks could increase these gaps substantially (Jones et al. 2006).

Carbon emission gaps by 2100 could be as high as 25 Pg C yr^{-1}, almost four times the current annual emission of CO_2-carbon to the atmosphere (6.3±1.3 Pg C yr^{-1}). Since the maximum annual global C sequestration potential is 0.9 ± 0.3 Pg C yr^{-1}, this implies that even if these rates could be maintained until 2100, soil carbon sequestration would contribute a maximum of 2–5% towards reducing the carbon emission gap under the highest emission scenarios in the long-term. The limited duration of carbon sequestration options in removing carbon from the atmosphere, also implies that carbon sequestration could play only a minor role in closing the emission gap by 2100. However, if atmospheric CO_2 levels are to be stabilised at concentrations below 450–650 ppm by 2100, drastic reductions in emissions are required over the next 20–30 years (IPCC 2000). During this critical period, all measures to reduce net carbon emissions to the atmosphere would play an important role – there will be no single solution (IPCC 2000) and options, such as reduced emissions from deforestation (Gullison et al. 2007) could also contribute significantly. Since carbon sequestration is likely to be most effective in its first 20 years of implementation, it could form a central role in any portfolio of measures to reduce atmospheric CO_2 concentrations over the next 20–30 years, whilst new energy technologies are developed and implemented (Smith 2004b).

5.2.4 Climate Change Impacts on Greenhouse Gas Fluxes from Cropland Soils

As mentioned in the introduction, climate change alone could alter the future levels of carbon storage and trace gas fluxes from cropland soils, since changes in temperature, precipitation and atmospheric CO_2 concentration will affect net primary production (NPP), carbon/nitrogen inputs to soil, soil carbon decomposition rates and soil nitrogen cycling. We will next examine predicted changes in each of these drivers, and how these could affect GHG fluxes from cropland soils.

5.2.5 Predicted Changes in the Climate Drivers of Cropland Greenhouse Gas Fluxes

Most General Circulation Models (GCMs) agree that much of the globe is likely to experience considerable warming over the next century, with best estimates of the global mean temperature increase by 2100 ranging from 1.4 to 4.0°C (likely range 1.1–6.4°C), dependent on GCM and emissions scenario (IPCC 2007a). Warming is predicted to be greatest over land and high northern latitudes (Fig. 5.2a). However, there is much less agreement regarding which regions will experience increases or decreases in precipitation and soil moisture. Increases in the total annual amount

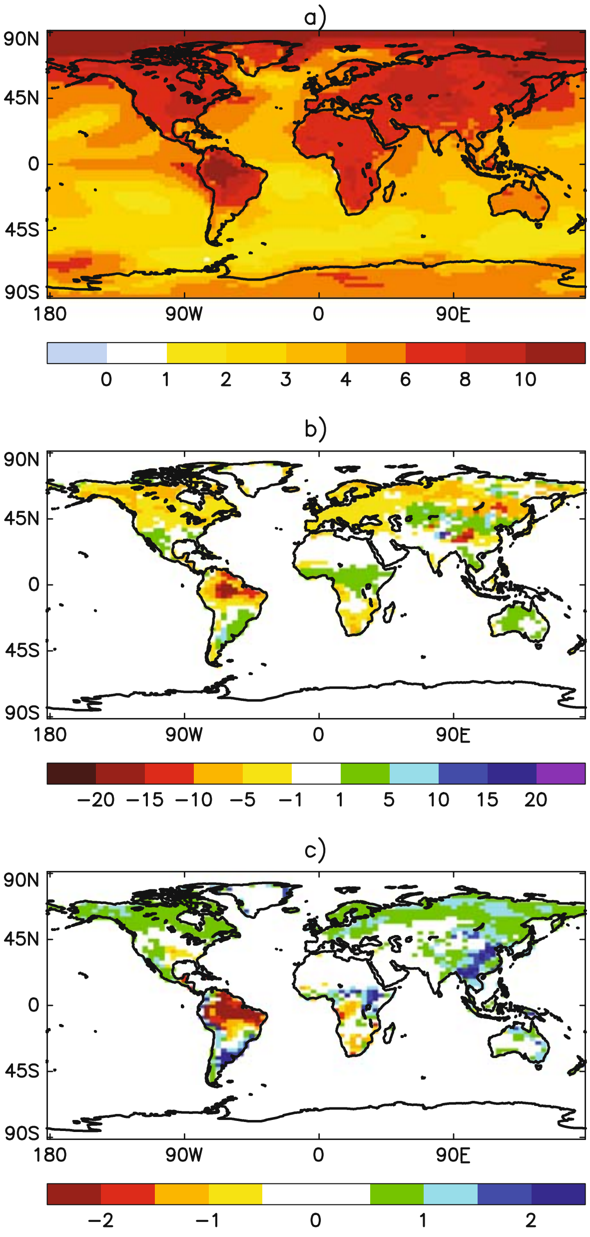

Fig. 5.2 Change in (**a**) temperature (°C), (**b**) top-level (0–10 cm) soil moisture (kg m^{-2}) and litter C inputs (kg C m^{-2}), from 2000 to 2100 – HadCM3LC model (IS92a emissions scenario) (Cox et al. 2000)

of precipitation are predicted in high latitudes, while decreases of up to 20% by 2100 are predicted in most sub-tropical land regions (IPCC 2007a). Changes in the seasonal pattern of precipitation are also likely. In Northern Hemisphere winter (DJF), precipitation increase of over 20% are predicted for Northern Hemisphere high latitudes and Eastern Africa, whilst decreases of over 20% are predicted for the Southern USA, North Africa and the Middle East, by 2100. In Northern Hemisphere summer (JJA), precipitation increases of up to 20% by 2100 are predicted for Northern Hemisphere high latitudes, and decreases of up to 20% for Southern Europe, North Africa, South Africa, Brazil and Central America (IPCC 2007a). The regions where there is least agreement between GCM projections of changes in precipitation are the USA, Russia, South Asia, Australia, tropical Africa in summer, and Southern Africa, Australia, South Asia, mid USA and South America in winter (IPCC 2007a). Figure 5.2 shows predicted changes in temperature, soil moisture and litter C inputs under a coupled climate-carbon cycle simulation (Cox et al. 2000)

These changes in annual totals and seasonal patterns and in the magnitude and frequency of extreme climate events are also predicted, including warmer and fewer cold days and nights, warmer and more frequent hot days and nights, an increased frequency of warm spells/heat waves and heavy precipitation events, and a general increase in the areas affected by droughts. Atmospheric CO_2 concentrations were approximately 280 ppm during the pre-industrial period, and 379 ppm in 2005, while under the IPCC SRES scenarios, they are predicted to rise between 730 and 1020 ppm dependent on scenario and the magnitude of climate-carbon cycle feedbacks (IPCC 2007a). The GCMs generally agree in the sign of soil moisture changes in many regions, although the magnitudes of change are more uncertain. Annual mean decreases of up to 20% in soil moisture are predicted in the subtropics, the Mediterranean region and in high latitudes, where snow cover diminishes (Fig. 5.2b), whilst increases of over 20% in precipitation are predicted in east Africa, Central Asia, and some other regions (IPCC 2007a). Jones and Falloon (2007) discuss the implications of future climate uncertainty in determining future SOC storage.

5.2.6 Predicted Changes in Land Use, Cropland Productivity and Management

Future changes in the amount and quality of plant C and N inputs returned to cropland soils depend on several factors, including changes in large scale land use, crop suitability (crop types grown in particular regions), crop productivity, and crop-specific factors (root to shoot ratios, and litter quality).

In 1990, ~12% of the global land area was in crop and energy use, ~33% in forests and ~55% in other uses including grasslands. The IPCC SRES scenarios suggest a wide range of possible future mixes of global land use, dependent upon the scenario chosen. By 2100, global land use could consist of ~5–25% crop and energy use, ~25–45% forests and ~47–62% other land uses including grasslands (IPCC 2000). Thus, both decreases and increases in the present global land area under crops are possible. Large regional differences in land use changes are anticipated – for instance, the A2 scenario includes widespread deforestation in Amazonia and Africa and some reforestation in Europe, whilst the A1B scenario suggests much less deforestation in Amazonia and Africa, and more widespread afforestation especially in regions of China and Europe (Falloon and Betts 2006).

Whilst climate change could alter the suitability of crops and thereby, the geographic areas occupied by particular cropping systems (Smith et al. 2007b). However, there is a little information on the nature of these changes, not least because economics and agricultural policy are strong drivers, and subject to considerable uncertainty. For instance, increasing temperatures are likely to have a positive effect on crop production in colder regions due to a longer growing season, (Smith et al. 2005), allowing crop production zones to shift northward. However, increases in productivity may not necessarily lead to increases in carbon storage, since climate change could also increase the length of the season when respiration occurs (Harrison et al. 2008). In mid- to high latitude regions, a combination of local temperature increases of 1–3°C, carbon dioxide (CO_2) increase and rainfall changes could have small beneficial impacts on crop yields, although in low-latitudes, similar changes are considered likely to have negative yield impacts for major cereals (Fischlin et al. 2007). Further, warming is projected to have increasingly negative impacts in all regions and global production potential is likely to decline as global average temperature rises above +3°C. Air pollution could also reduce crop yields, since tropospheric ozone has negative effects on biomass productivity (Booker and Fiscus 2005; Liu et al. 2005).

There is a little information concerning climate impacts on the crop-specific factors which influence C and N inputs to soil, and given the complexities discussed above, there is considerable uncertainty in likely responses. For example, the predicted doubling of atmospheric CO_2 concentrations in the next century will alter plant growth rates, plant litter decomposition, drought tolerance, and nitrogen demands (Torbert et al. 2000; Norby et al. 2001; Jensen and Christensen 2004; Henry et al. 2005; van Groenigen et al. 2005; Long et al. 2006). Given that changes in crop breeding and production systems are also likely, it is difficult to speculate how climate change might alter factors, such as root to shoot ratios and litter quality. However, in many regions, improvement in cultivars has a far greater impact on yields in the past few decades than any other factor (Amthor 1998). Where maximum potential yields are not yet attained, this could continue to be the case in the coming decades. With more of the NPP being directed toward harvestable yield, carbon content produced in other components of crop plants has decreased during recent decades. This trend could continue over the coming decades, resulting in lower C and N returns to soil each year.

5.2.7 Climate Change Impacts on Cropland Greenhouse Gas Fluxes

Greenhouse gas fluxes from croplands will be affected by the changes in climate, cropping systems and management drivers discussed above. Increases in temperature alone are likely to accelerate decomposition of soil organic matter, resulting in soil carbon losses (Knorr et al. 2005; Fang et al. 2005; Smith et al. 2005). The global scale assessments made by Cox et al. (2000) and Jones et al. (2003), using a coupled climate-C cycle Global Circulation Model (GCM), HadCM3LC, predicted decreases in soil C stocks across most of the globe by 2100, even in regions where C inputs to soil from vegetation had increased (Jones et al. 2003). Regionally, large decreases in soil carbon for the Amazon region (over $4\,\mathrm{kg\,C\,m^{-2}}$), Southern Africa ($2–4\,\mathrm{kg\,C\,m^{-2}}$) and Eastern USA ($2–4\,\mathrm{kg\,C\,m^{-2}}$) and increases in soil carbon for Siberia, Alaska and Northern Canada and much of Eurasia ($1–3\,\mathrm{kg\,C\,m^{-2}}$) are predicted (Jones et al. 2004; Falloon et al. 2007a). In the C4MIP study of Friedlingstein et al. (2006), all coupled climate-carbon cycle models showed a decrease in soil carbon storage globally due to climate change. Higher temperatures will also increase N mineralisation rates, which may increase N_2O fluxes and nitrogen leaching, especially in N saturated systems.

How changes in precipitation (and hence soil moisture) alone will affect GHG fluxes from cropland soils is more complex than as previously discussed, future predictions of precipitation (and soil moisture) are less certain than future changes in temperature. Globally, overall decreases in soil moisture alone could lead to increased global soil carbon storage because drying will reduce respiration rates (Falloon et al. 2007a). Regional scale studies in the UK also found that future moisture changes alone can increase cropland soil C storage (Falloon 2004). Hence, the predicted annual decreases in soil moisture for the sub-tropics and Mediterranean region are likely to lead to increased soil C storage, whilst the predicted increases in soil moisture for East Africa and Central Asia are predicted to enhance reduced soil C stocks. The impact of seasonal moisture changes is less certain, although the predicted increases in winter precipitation for Northern hemisphere high latitudes and East Africa, and summer precipitation for East Africa and Central Asia, could lead to reduced soil C stocks. The predicted decreases in summer precipitation for Southern USA, North African and the Middle East, and winter precipitation for Southern Europe, South Africa, North Africa, Brazil and Central America, may act to increase soil C stocks. Higher winter rainfall could also increase N_2O production and emission, and permanent waterlogging of wet soils may increase CH_4 emissions. Conversely, the drying, as discussed above, may lead to reduced N_2O and CH_4 emissions. How these seasonal changes balance annually and in the long-term is very complex and will depend upon the relative influence of wetting/drying patterns on GHG fluxes in each season.

There has been little research on the impacts of changes in climate extremes on GHG emissions from cropland soils, although the recent European heatwave of 2003 led to significant carbon fluxes from terrestrial ecosystems (Ciais et al. 2005). Extreme increases in soil temperatures and drought events may also have implications on soil biological activity, reducing the decomposition capability of bacteria, ultimately reducing biomass growth and soil fertility. An increased frequency and magnitude of heavy rainfall events could impact cropland GHG fluxes by increasing soil erosion (and thus losses of soil C to watercourses), or by increasing the occurrence of short periods of warm and wet conditions suitable for N_2O production. The predicted increase in atmospheric CO_2 concentrations is likely to have only small overall impacts on soil carbon storage (Smith 2005). Although soil organic matter is often increased under elevated CO_2 in the short-term (Allard et al. 2004), the long-term soil carbon sink may saturate at elevated CO_2 concentrations, especially when nutrient inputs are low (Gill et al. 2002; van Groenigen et al. 2006).

The uncertainties in future cropping and land use patterns make it difficult to postulate their likely impacts on cropland GHG fluxes. However, impacts of the predicted changes in crop productivity on GHG fluxes can be inferred. Assuming that the fraction of C returned to soil remains unchanged, the small mid-term increases in yield predicted for Mid-High latitudes may lead to small increases in C inputs to soil over the next few decades and so increased soil C storage. In the longer term, the predicted decreases in crop yields would lead to reduced C inputs to soil, and thus reduced soil C storage. However, the general trend to reduce below-ground C allocation in crops is considered likely to continue (Amthor 1998). Thus, overall response will be a net decrease in C inputs to soil, and so in soil C storage, unless crops are specifically bred to allocate more C below ground. A decrease in C and N inputs to soil is expected to reduce cropland N_2O emissions. The impacts of air pollution on crop production could also indirectly affect soil carbon storage. Recent researches suggest that tropospheric ozone could significantly reduce carbon-sequestration rates under elevated CO_2 (Loya et al. 2003; Sitch et al. 2007), due to reduced biomass productivity and altered litter chemistry (Booker and Fiscus 2005; Liu et al. 2005). This would further reduce soil C storage in croplands.

The combined impact of changes in temperature, precipitation, atmospheric CO_2 concentration, crop productivity, land use change and other factors is difficult to predict without holistic studies incorporating all of the above factors. Falloon (2004) investigated the impact of climate change on UK arable soils with the RothCUK soil carbon model (Falloon et al. 2006a), changing either only single climate variable, or all variables simultaneously. This showed that whilst temperature changes acted to decrease, SDC stocks precipitation and evapotranspiration changes acted to increase. This led to an overall decrease in SOC stocks. Moreover, the effects were not simply additive – summing the outputs of runs changing only single climate variable did not produce the same result as runs changing all climate variables at the same time (Fig. 5.3). This non-linearity in the response of soil C storage to different driving factors reinforces the need for a holistic systems modelling approach to assess climate impacts on C sequestration.

A number of studies, such as those of Cox et al. (2000) and Jones et al. (2003), discussed earlier in this chapter, have used coupled climate-carbon cycle GCMs which are able to capture the feedbacks between soils, vegetation and the atmosphere, although these models presently only represent natural ecosystems and not croplands. Further examination of these results (Falloon et al. 2007a) indicated that the regional response of soil C to precipitation differed from the global response – although there was a slight increase in precipitation globally, a decrease in C stocks was predicted. The reason for this may be been that whilst temperature increases under climate change were predicted everywhere, the nature of precipitation changes varied greatly between regions. Thus precipitation may control the sign of regional soil C changes under climate change with wetter conditions resulting in

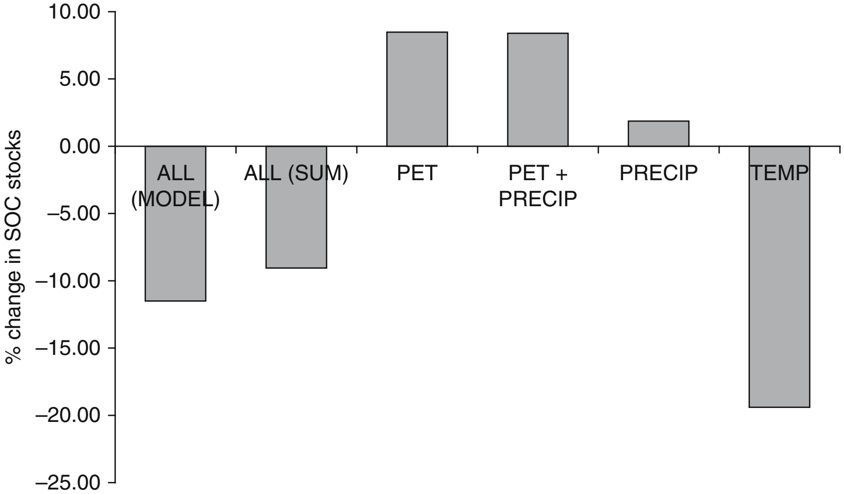

Fig. 5.3 The impact of climate change on UK arable soil C stocks under the SRES A1F1 scenario (HadCM3 model, 2080s) using the RothC soil carbon model. PET, PRECIP, TEMP = changing only potential evapotranspiration, precipitation or temperature; PET + PRECIP – changing both PET and PRECIP; ALL = changing PET, PRECIP and TEMP (MODEL) simultaneously in the model, and summing values from runs changing single climate variables (SUM)

higher soil C stocks and drier conditions in lower soil C stocks, since increased NPP in wetter conditions could override any increase in respiration. In contrast, temperature appeared to control changes in C stocks under climate change globally. Finally, a recent inter-comparison of coupled climate-carbon cycle GCMs found a wide range of positive climate-carbon cycle feedback values, showing response of soil carbon to climate change to be highly uncertain (Friedlingstein et al. 2006).

5.3 Climate Change Threats and Opportunities for Greenhouse Gas Mitigation in Cropland Soils

5.3.1 Impacts of Climate Change on Cropland Greenhouse Gas Mitigation Options and Their Potential

Climate change could potentially act to reduce (or increase) GHG mitigation potential through different constraints as depicted in Fig. 5.1. Changes in temperature, moisture and CO_2 concentrations could act to alter the biological potential and land suitability which, in turn, along with other climate-related factors could alter economically constrained potential. For example, by changing the demand for crops and agricultural products, future changes in environmental policy, such as adaptation and mitigation efforts, could alter the socially/politically constrained potential.

Falloon and Smith (2003a) used the RothCUK soil carbon model to investigate the impact of climate change on the carbon sequestration potential of alternative land-use and land management practices in the UK. This showed that while there was considerable potential for single land management option to contribute towards Kyoto Protocol emissions reductions targets (up to 0.8% of 1990 UK CO_2 emissions, or 6.4% of the UK target), climate change could significantly reduce C sequestration potential of alternative land management options (Fig. 5.4), by up to 57% of the estimate without climate change, depending on land management and climate change scenario.

These simulations also found a non-linearity in the response of soil C storage, climate change and land management changes, since summing the results from runs considering either only climate change or land management changes did not equal the results of runs changing both factors simultaneously. These studies only considered the impact of climate change on soil carbon storage, and did not assess how crop responses to climate change might alter C inputs to soil and hence soil C sequestration. In light of likely future decreases in C inputs to croplands due to reduced yields and changes in harvest index as discussed above, a further reduction in soil C sequestration potential under climate change might be expected at the global scale.

Smith et al. (2005) assessed future changes in European cropland and grassland soil carbon stocks, using the RothC soil carbon model incorporating climate data from four global climate models under different IPCC SRES scenarios, with

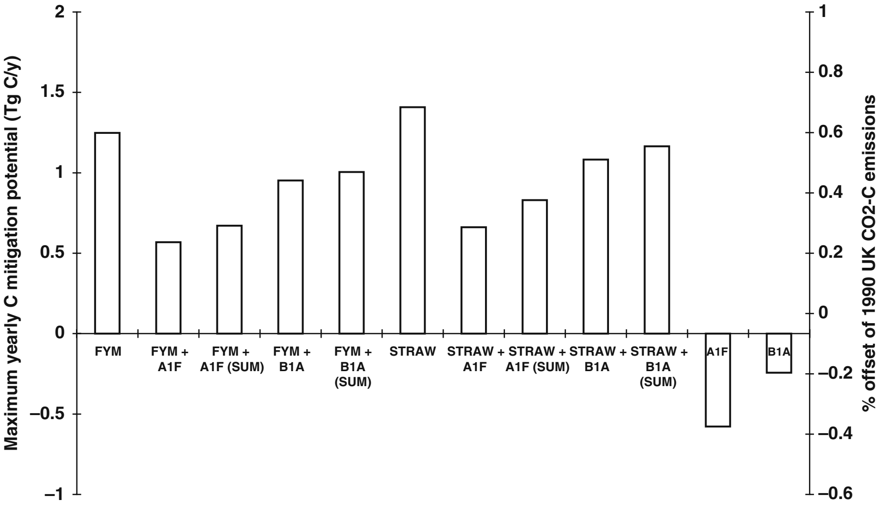

Fig. 5.4 Impact of climate change under the IPCC SRES A1F1 and B1A scenarios (2080s) on C sequestration potential of UK arable soils using the RothCUK model. FYM = additional $10\,t\,ha^{-1}\,yr^{-1}$ farmyard manure, STRAW = additional $5\,t\,ha^{-1}\,yr^{-1}$ cereal straw, A1F and B1A = IPCC SRES HadCM3 A1F1 and B1A climate change scenarios for the 2080s respectively. SUM = calculated by summing values from runs changing only climate or land management

changes in NPP calculated by the Lund–Potsdam–Jena model, and land-use change scenarios derived from the IPCC SRES story lines. While climate effects were predicted to decrease carbon stocks, the predicted increases in NPP across Europe (and hence carbon input) and technological improvements were predicted to slow these losses. Considering all these factors, Smith et al. (2005) found European cropland soils to show a small (1–7 t C ha^{-1}) increase in soil carbon on per unit area basis under future climate. Accounting for the large predicted decrease in cropland area, total European cropland C stocks is expected to decline in all scenarios from 11 Pg in 1990 by 4–6 Pg (39–54%) by 2080.

Smith et al. (2005) assumed that changes in carbon inputs to the soil were proportional to increases in NPP and hence, the influence of NPP on C inputs in their study should be regarded as the maximum possible. The range of possible changes in C returns, due to changes in yield associated with improvements in technology, was calculated using equations for the lowest (winter wheat) and highest (winter oilseed rape) soil carbon return per unit yield increase (Smith et al. 1996). Under the B1 scenario (2050), a 62% change in yield resulted in a maximum and minimum change in carbon return of 19 and 6%, respectively. The predicted SOC stocks under each assumption differed by about 9%, equivalent to an uncertainty of 1 Pg for cropland soils. Uncertainty of the impact of NPP on soil C inputs was 9% (1 Pg) of croplands SOC estimates. Further, an un-quantified uncertainty was introduced in the NPP estimates, as NPP estimates by LPJ were not nitrogen limited, meaning that the CO_2 response could have been overestimated. However, this potential uncertainty fell within the quantified uncertainty range for NPP given above, where the effect could range from 0 to the impact predicted (assuming C returns to the soil increase in proportion to NPP). This study demonstrates considerable regional uncertainties in the response of C inputs, and soil C changes under climate change (Smith et al. 2005).

Felzer et al. (2005) used the TEM terrestrial ecosystem model to assess future ozone and climate impacts on global carbon sequestration, including the impacts of CO_2 fertilization, but excluding land use changes. They predicted overall gains in global cropland carbon storage due to climate change by 2100, although ozone damage to crops could significantly offset these increases, with the largest damages occurring in the Southeast and Midwestern regions of the United States, Eastern Europe, and Eastern China.

5.3.2 Uncertainties in Climate Impacts on Cropland Greenhouse Gas Mitigation Potential

Uncertainties in climate impacts on cropland GHG mitigation potential could arise from uncertainties in (a) climate change, (b) how cropland GHG fluxes respond to climate change, (c) climate impacts on land use and management changes, (d) and other uncertainties/processes. Uncertainties in climate change could also arise from missing or incomplete descriptions of processes in GCMs, uncertainties in future emissions of greenhouse gases, uncertainties in GCM parameters, or differences between models. The latter two can be investigated using multi-member ensemble runs of the same climate model run with different parameter sets (Murphy et al. 2004), and by inter-model comparisons (Friedlingstein et al. 2006). Falloon et al. (2006b) used data from four members of a large multi-member climate model parameter perturbation ensemble with the RothC soil carbon model to assess uncertainties in climate impacts on global soil carbon storage under natural vegetation. Global soil carbon changes under climate change were found to cover both small gains and large losses (mean 91.5 Pg C, range 356.8 Pg C) depending on the climate sensitivity of the ensemble member (Fig. 5.5), with the role of litter inputs (NPP) dominating the response of soil carbon globally and regionally, and temperature and moisture playing a smaller role globally. Smith et al. (2005) found uncertainty of 4.5% in European cropland soil carbon response to climate due to differences in the climate produced by different climate models. Uncertainties in future emissions of greenhouse gases can be investigated using a wide range of emissions scenarios (IPCC 2000). Whilst a range of responses of soil C changes to emissions scenarios of up to 50% were found by Smith et al. (2005) and Falloon (2004), different scenarios are generally considered more likely to affect the magnitude of response rather than the pattern or sign of change (Falloon et al. 2007a).

The main uncertainties in cropland GHG flux responses to climate change relate to how GHG fluxes respond to changes in temperature, moisture and carbon and

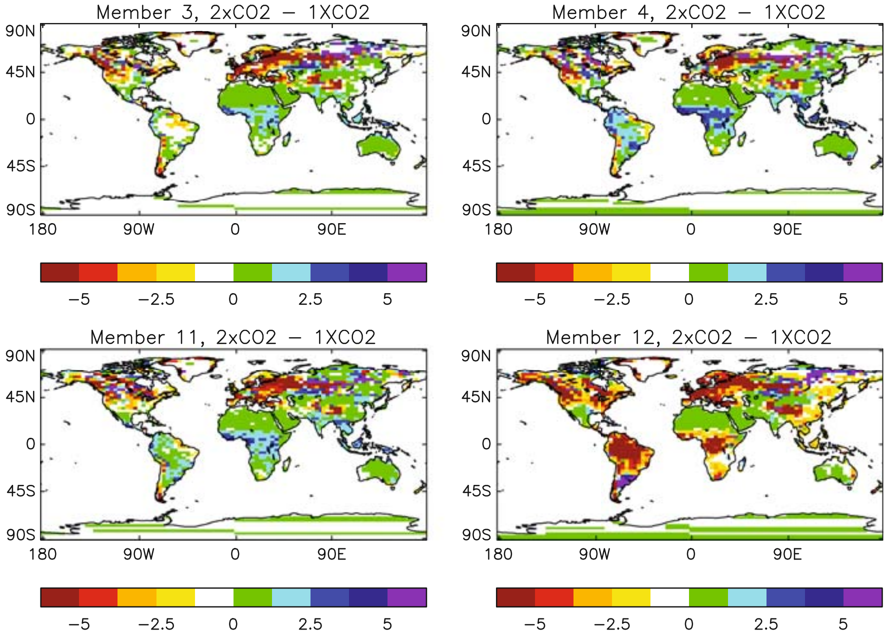

Fig. 5.5 Changes in global soil carbon stocks (kg C m^{-2}) from four different HadCM3 single parameter perturbation ensemble members – difference between 2×CO_2 and 1×CO_2

nitrogen inputs to soils. There is presently no consensus on the temperature sensitivity of soil carbon stocks (Davidson and Janssens 2006) and therefore, considerable and unquantifiable uncertainty. However, Knorr et al. (2005) suggest that these conflicting opinions are compatible with long-term temperature sensitivity of SOC turnover and may be explained by rapid depletion of labile SOC combined with the negligible response of non-labile SOC on experimental time scales. Since non-labile SOC may be more sensitive to temperature than labile SOC, the long-term positive feedback of soil decomposition could be even stronger than predicted by global models (Knorr et al. 2005). While the impact of soil temperature on soil carbon storage has been the subject of considerable debate (Giardina and Ryan 2000; Fang et al. 2005; Knorr et al. 2005; Davidson and Janssens 2006), the influence of soil moisture on large-scale soil carbon stocks has received little attention used twelve soil moisture-respiration functions with the RothC soil carbon model and data from a coupled-climate carbon cycle GCM to investigate the impact of heterotrophic respiration dependent on soil moisture on the global climate-carbon cycle feedback under natural vegetation. Considerable uncertainty in the soil carbon changes was found due to response of soil respiration to soil moisture – the range of global soil carbon changes from 1860 to 2100 was 71.8 Pg C (minimum −54.1 Pg C, maximum 17.7 Pg C), considering temperature, moisture and litter changes, and 60.9 Pg C (minimum −17.8 Pg C, maximum 43.1 Pg C) considering only changes in moisture (Fig. 5.6).

In contrast, the response of soil carbon to changes in C and N inputs is well understood. Although it is difficult to directly measure total C and N inputs to soil, since there are both above and below ground components (Falloon 2001) and there is a strong linear relationship between soil C storage and C inputs (Paustian et al. 1997; Buyanovsky and Wagner 1998), up to a certain ‘saturation limit’. Errors in predicted SOC values, because of uncertainties in the size and quality of C inputs, have been assessed by Falloon et al. (1998) and Falloon (2001). Using data from a global network of long-term experiments and the RothC soil carbon model, Falloon et al. (1998) found that C inputs accounted for 60% of the variance in soil C stocks. SOC values were found to be more sensitive to C input quantity than quality, and C input quality was most important for forest systems with relatively small impacts on cropland SOC estimates (Falloon 2001). Halving or doubling the quantity of C inputs, when simulating a 120-year experiment, resulted in differences in SOC of 41 and +80%, respectively, reducing the fit to measured data by 28–71%, and an error of 10% in C inputs could result in an error of 14.35% in SOC stocks. Altering the quality of C inputs resulted in errors in modelled SOC of 0.2–11.6%, similar to the errors in C input of 5–8% calculated by Jenkinson et al. (1999). Long-term data sets of changes in SOC are often used to evaluate SOC models and estimate C inputs. However, most long-term data sets have only mean SOC values and no estimate of the error about the mean. Falloon and Smith (2003b) showed that when using data sets that do not include estimates of the error about the mean, it is not possible to reduce the error between modelled and measured SOC below 6.5–8.5% even with site specific calibration; equivalent errors for model runs using regional default C input values were 12–34%.

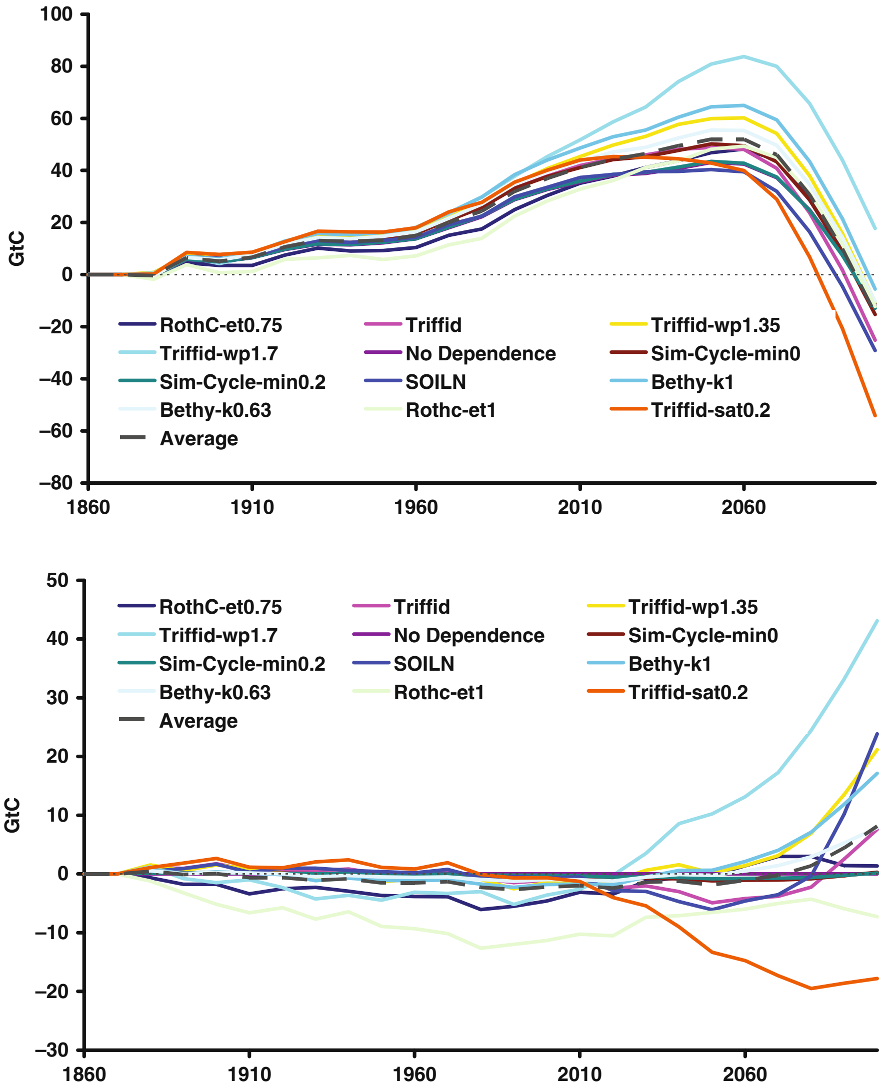

Fig. 5.6 Changes in global total soil carbon from 1860 values using the RothC model and different soil moisture-respiration functions driven by HadCM3LC outputs changing (**a**) all forcings (soil temperature, moisture and plant carbon inputs) and (**b**) soil moisture only

There is considerable uncertainty in future changes in C and N inputs to soils, which depend on both how crops respond to climate change and on changes in land use, crop technology and management. Betts et al. (2006) examined the uncertainties in natural vegetation responses to climate change, diagnosing Net Primary Productivity from the QUMP ensemble. Although a global mean increase in NPP under doubled-CO_2 climate change was predicted, the climate-related uncertainty

(5–95 percentile range $> 0.6\,\mathrm{kg\,C\,m^{-2}\,yr^{-1}}$) was larger than the mean change in many regions. NPP in most regions was considered likely to increase, although decreases were simulated in a small number of cases. This was due to the impact of elevated CO_2 on NPP – the impact of climate on NPP is uncertain (Friedlingstein et al. 2006). In Amazonia, NPP decreases in parts of the basin in all simulations with HadCM3 were found, with the geographical extent of the NPP varying very widely between simulations. Challinor et al. (2005) investigated uncertainty in crop responses over India to doubled CO_2 using the same four ensemble members used by Falloon et al. (2006b), described earlier in this chapter. In all four ensemble members, the impact of as doubled CO_2 climate on the mean and standard deviation of yield was marked. Differences in simulated yields across ensemble members varied geographically, while differences in the inter-annual mean, and especially the standard deviation, varied in magnitude and sign across grid cells. Variations were greater in the doubled-CO_2 climate than in the present-day climate. Uncertainty in crop model parameters also led to marked differences in crop yield. However, the impact of crop parameter perturbation was more spatially systematic than that of climate parameter perturbation. Challinor et al. (2005) concluded that the major causes of uncertainty in the simulation of crop yield under doubled CO_2 included uncertainty in temperature (particularly how crop duration responds to increases in temperature), and in the CO_2 fertilization effect. As discussed above, the relationship between changes in NPP and C inputs under climate change is complex, since it involves both biological and technological aspects, and hence, a further source of uncertainty exits.

Uncertainties in climate impacts on future land use and management could have significant impacts on cropland GHG fluxes. Since future changes in land use, management and crop technology are dependent on patterns of future development, it is extremely difficult to quantify the likelihood of any particular scenario. Despite this, in the European scale studies of Smith et al. (2005), uncertainty associated with the land-use and technology scenarios was not quantified, but worst-case quantified uncertainties were 22.5% for croplands, equivalent to a potential errors in C sequestration of 2.5 Pg SOC, or 42–63% of the predicted SOC stock change. In UK-scale recent studies (1990–2000), land use impacts on soil carbon, uncertainty in input soil, climate and land use databases were considered likely to be <25%, although a tentative overall uncertainty range of 50–100% was considered reasonable (Falloon et al. 2006a). Climate change could also limit the feasibility of different land management options (Falloon et al. 2007a). For instance, if the predicted dieback of the Amazon forest (Cox et al. 2000, 2004; Betts et al. 2004; Cowling et al. 2004; Scholze et al. 2006) is correct, then clearly land management options favouring forestry will be severely limited in the Amazon region. Considerable reductions in forest cover are also expected due to the expansion of logging schemes and clearance of land for agricultural expansion (Laurance et al. 2004).

A remaining source of uncertainty is the biophysical impact of land use changes on the climate itself. Plans and strategies for adaptation to climate change require specific and local details of climate change. Although this can be provided by regional climate models, these are typically only used for down-scaling of radiatively-forced global climate change. Significant works (Lean and Warrilow 1989; Betts 2001; Feddema et al. 2001; Betts et al. 2007b) demonstrate that the biophysical effects of land use change are also of large importance. Land use change, particularly deforestation or reforestation, can exert significant impacts on local climates by influencing surface albedo and evaporation. Deforestation in temperate regions leads to a cooling through increased surface albedo (Betts 2001), while continued tropical deforestation is expected to lead to a warming and drying of local climate (Lean and Warrilow 1989). Biogeophysical effects of the Amazon forest dieback are important locally, acting to further reduce rainfall (Betts et al. 2004). Scenarios of greenhouse gas emissions implicitly assume changes in land use, but the direct effects of these changes are often not considered in climate change projections. Feddema et al. (2001) showed that the projected climate change, in some regions, can be significantly affected by the assumed land cover change associated with the emissions scenario. For example, the SRES B1 scenario implies reforestation in mid-latitudes and relatively little tropical deforestation, whereas the A2 scenario implies less mid-latitude reforestation, but extensive tropical deforestation. These differences in projected land cover lead to significant variations in the predicted climate change at regional scales.

Regional climate change studies used for impacts assessments should, therefore, consider uncertainties associated with land cover changes and their biophysical effects. This is particularly important for assessments of agricultural impacts, since consistency between land use and the overlying climate will be crucial (Betts 2005). For instance, Falloon and Betts (2006) showed that land use changes alone could have significant impacts on regional river flows, which would impact the feasibility of GHG mitigation options involving irrigation. Raddatz (2007) also found that agriculture has an impact upon near surface weather elements and regional hydrological cycles, through the physiological and physical properties of the land cover. By changing the availability of energy and water vapour mass for moist deep convection at local and regional scales, and creating latent heat flux discontinuities, agriculture may induce mesoscale circulations that initiate moist deep convection. By altering the level of stored soil moisture, agriculture can also may influence the level of seasonal convective activity within a region (Raddatz 2007).

Irrigation itself can also increase the surface moisture flux and hence reduce the Bowen ratio, exerting a cooling influence on local near-surface temperatures (Betts 2007). Boucher et al. (2004) introduced present-day patterns of irrigation into the Laboratoire de Meteorologie Dynamique (LMD) GCM and found a simulated surface cooling of up to 0.8 K in some regions. Schaeffer et al. (2006) suggested that the use of biofuel may exert a double effect on reducing temperature rise by mitigating CO_2 emissions, while maintaining a relatively high surface albedo. Biofuel crops tend to be short in stature and hold less foliage than forests and so the surface albedo of an area of biofuel plantations is higher than a forested landscape. In contrast, since tropical deforestation exerts a warming effect through reduced evapotranspiration (Betts 2007), reforestation (or avoided deforestation) in tropical regions could exert a double cooling effect through carbon sequestration and increased evaporation and cloud cover. Jackson et al. (2005) showed that water resources are directly impacted by forestry activities designed for carbon sequestration. Climate change adaptation plans may, therefore, be inappropriate if based on projections which ignore land use change.

5.4 Conclusions

There is considerable potential for climate mitigation via cropland soil carbon sequestration and GHG reductions. Care should be taken for the choice of alternative land management strategies since there may be additional negative or positive impacts including both socio-economic and environmental aspects. The mitigation potential of cropland management options, which is actually achievable in practice, given economic, political, social, land suitability and other constraints may only be around 10–20% of the biological potential. Alternative cropland management strategies could meet only 2–5% of emissions gaps. However, many such options are most effective during the first 20 years following implementation, so they have a key role to play in any portfolio of emissions reduction measures over the next 20–50 years, while new energy technologies are developed and implemented.

Globally over next 100 years, climate models project temperature increases of 1.4–4.0°C, and significant changes in regional precipitation patterns, whilst CO_2 concentrations could reach 2–5 times of present day values. Changes in global land use patterns could occur, with increases and decreases in the present cropland, both possible. While some increases in crop productivity are expected in the short to mid term in Northern mid-high latitudes, globally decreases are likely in the longer term in the absence of adaptation. If present trends towards more harvestable product continue, globally crops are likely to return less C and N to soils due to changes in the harvest index.

GHG fluxes from croplands will be altered by these changes in controlling factors. Temperature increases will generally increase GHG fluxes; the influence of moisture changes will depend on regional patterns of change – for instance, drying alone could increase soil C sequestration. Changes in CO_2 concentrations alone are expected to have only small impacts on cropland GHG fluxes. Globally, long-term reduced crop productivity and changing harvest index are likely to reduce C and N inputs to soil, thus reducing soil carbon storage and GHG fluxes in the absence of adaptation measures. The combined impact of these factors is not simply additive, and most holistic systems studies have been conducted for natural ecosystems, not croplands. These studies generally show reductions in global soil carbon storage.

Climate change is likely to reduce cropland soil carbon sequestration and GHG mitigation potential in many regions in the long-term in the absence of mitigation, perhaps with the exception of Northern Hemisphere mid-high latitudes in the short-mid term. Increased ozone concentrations are likely to further reduce GHG mitigation potential, particularly in some of the major global cropping regions. Changes in future cropland GHG mitigation potential are, however, strongly dependent on changes in agricultural technology and crop breeding. Uncertainties in climate change impacts on cropland GHG mitigation potential are large, and derived from several sources including climate predictions, the extent of climate impacts, and changes in future cropland distribution and management. The biophysical impacts of changes in land use and management on climate can be significant, and hence need to be considered with other factors while determining suitable GHG mitigation options for croplands. Current GHG mitigation practices should be, therefore, reassessed to account for both biogeochemical and the biogeophysical forcing, acknowledging the significant opportunities and risks that occur in the complex interactions between agriculture and the environment (Desjardins et al. 2007), as in the preliminary approach of Seguin et al. (2007) using the LPJ model. Research needs include improved representation of croplands in GCMs (Osborne et al. 2007; Bondeau et al. 2007). With fully-coupled crop-climate models such as these, and with adequate account of uncertainty, a robust understanding of soil-atmosphere-crop interactions will emerge. There is a lack of research into the impacts of changes in extreme events on GHG fluxes from croplands, and a need for more detailed information on regional aspects of crop suitability and productivity, technology changes, crop breeding impacts and land use changes is greatly realized. More detailed regional information is particularly critical, if some of the projected regional differences are large in climate, and socio-economic factors. For instance, during the next three decades, Asia will remain the largest global food consumer (consumption could rise from 40 to 55% of the global total between 2000 and 2015) and the largest source of GHG from agriculture (approximately 50% of total emissions). Unless improved management systems are adopted, a substantial increase in GHG emissions from the agri-food sector in Africa and South America will arise due to growing demand for food (Vergé et al. 2007).

Regardless of their GHG mitigation potential, maintaining the organic matter content of cropland soils is fundamental in supporting soil quality and sustainable production systems (Bradley et al. 2005), as recognised by the UK Soil Action Plan (Defra 2003) and the EU Thematic Soil Protection Policy (CEC 2004). There are also requirements for nations to maintain and regularly update their Greenhouse Gas Emissions Inventories (GHGEI), category on land use change (LUC) and forestry under the United Nations Framework Convention on Climate Change (UNFCCC) and the EU Monitoring Mechanism for greenhouse gas emissions (for EU members). Thus, by combining information on the impact of both land use change and/or climate change on climate and cropland GHG fluxes, we will be able to make more holistic estimates of the impact of different scenarios of land use in altering regional and global climate. In this way, the relative contribution of land use change and anthropogenic emissions to climate change can be assessed, and land management options may be planned to minimise impacts on the climate. Finally it is important to identify potential synergies between land-based adaptation and mitigation strategies, linking issues of carbon sequestration, emissions of greenhouse gases, land-use change and long-term sustainability of production systems within coherent climate policy frameworks (Smith et al. 2005; Rosenzweig and Tubiello 2007; Easterling et al. 2007). Further, progress in understanding how climate change might impact GHG mitigation potential in croplands, therefore, requires more global, and particularly regional studies taking a holistic systems approach, by including crops and cropland management in earth systems models.

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Chapter 6
Greenhouse Gases from Crop Fields

Zhengqin Xiong and M.A.K. Khalil

6.1 Introduction

A rapid increase in atmospheric concentrations of the three main anthropogenic greenhouse gases (GHGs), like carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), is evident from measurements taken over the past few decades as well as ice-core records spanning many thousands of years (IPCC 2007). The global increases in CO_2 concentration are due to fossil fuel and land-use change, while those of CH_4 and N_2O are primarily from agriculture (Cole et al. 1997; IPCC 2007). Despite large annual exchanges of CO_2 between the atmosphere and agricultural lands, the net flux is approximately balanced (IPCC 2007). Arable and permanent crops occupy 1,540 Mha in 2003 which is about 12% of the Earth's land surface (FAOSTAT 2006). In 2005, agriculture contributes about 47 and 58% of total anthropogenic emissions of CH_4 and N_2O, respectively, with a wide range of uncertainty in the estimates of both the agricultural contribution and the anthropogenic total.

If demands for food increase and diets shift as projected, then annual emissions of GHGs from agriculture may increase proportionately. But improved management practices and emerging technologies may permit a reduction in emissions per unit of food produced, and a reduction in emissions per capita food consumption. We will focus on the processes as sources of CH_4 and N_2O in crop fields, and on agricultural practices that can influence their emissions.

6.1.1 Methane (CH_4)

CH_4 is an important greenhouse gas with a global warming potential of about 20 over a 100 year period and it accounts for 20% of the total radiative forcing from all of the long-lived and globally mixed greenhouse gases over the last 100 years. The

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Table 6.1 Sources, sinks and atmospheric budgets of CH_4 (Tg CH_4 $year^{-1}$)

References	Indicative ^{13}C, permil[a]	Khalil and Shearer (1993)	Mikaloff Fletcher et al. (2004)[b]	Chen and Prinn (2006)[b]	IPCC (2007)
Natural sources			260	168	
Wetlands	−58	110	231	145	
Termites	−70	15–35	29	23	
Ocean	−60	4			
Anthropogenic sources			350	428	
Ruminants etc.	−60	55–90	91	189	
Rice paddy fields	−63	55–90	54	112	
Domestic sewage		27–80			
Landfills	−55	11–32	35		
Animal waste		20–30			
Coal Mining	−37	25–50	30	48	
Natural gas leakage	−44	30	52	36	
Biomass burning	−25	50	88	43	
Total sources		402–601	610	596	582
Total sinks			577		581
Tropospheric OH	5.4		507		511±103
Stratospheric loss	−12		40		40±8
Soils	22		30		30±15

[a] From Mikaloff Fletcher et al. (2004) and entries for sinks are the fractionation.

[b] Estimates from global inverse modeling (top-down method).

global atmospheric concentration of CH_4 has increased from a pre-industrial value of about 700 ppb to 1,745 ppb in 1998, and to 1,774 ppb in 2005 (IPCC 2007). Once emitted, CH_4 remains in the atmosphere for approximately 8 years before removal (Dentener et al. 2003). The major CH_4 sinks are oxidation by OH in the troposphere, biological CH_4 oxidation in dry soil, and loss to the stratosphere. Oxidation by chlorine (Cl) atoms in the marine atmospheric boundary layer is suggested as an additional sink for CH_4, possibly constituting an additional loss of about 19 Tg CH_4 yr^{-1} (Allan et al. 2005).

CH_4 sources can be divided into anthropogenic and natural sources. Anthropogenic emissions dominate present-day CH_4 budgets, accounting for more than 60% of the total global budget. The anthropogenic sources include rice agriculture, livestock, landfills and waste treatment, biomass burning, and fossil fuel combustion. Natural CH_4 is emitted from wetlands, oceans, forests, fire, termites and geological sources. CH_4 concentration is higher in the northern hemisphere as most sources are larger. The relative contributions from different source types are shown in Table 6.1.

Cultivated wetland rice soils emit significant quantities of CH_4 (Khalil et al. 1998c, Yan et al. 2003). Estimates of rice paddies, as a global source of CH_4, range from 20 to 100 Tg yr^{-1}, which is equivalent to 5–28% of the total CH_4 from all other anthropogenic sources. Uncertainty in this estimation is mostly due to the complexity of extrapolating measured fluxes on a global scale. Direct measurements result in a wide range of local CH_4 fluxes for different climatic conditions, soil types, and cultivation practices of the rice-growing regions of the world.

CH_4 from rice is emitted mostly from South and East Asia, where it is a dominant food source (82% of total emissions). In comparison to other CH_4 sources, rice paddies are unique because they provide the principal food source for over 50% of the world's population. Thus, methods to mitigate CH_4 emissions from paddies must also address the consequences of those methods on crop yield. The ultimate goal is to reduce CH_4 emissions while maintaining or increasing rice grain yield.

6.1.1.1 Processes Controlling Production, Oxidation, and Emissions

CH_4 is produced when organic materials decompose in oxygen-deprived conditions, notably from fermentative digestion by ruminant livestock, from stored manures, and from rice grown under flooded conditions (Conrad 1996; Mosier et al. 1998). Similar to the conditions of natural wetlands, CH_4 is produced by methanogenic archaea from acetate (i.e. the methyl group) and H_2/CO_2 as precursors in the anaerobic environments of flooded rice fields. The base material, from which CH_4 is produced, may be supplied by organic amendments as well as by the roots of the rice plants. Studies show that a large fraction is oxidized by the methanotrophic bacteria before it can reach the atmosphere. These bacteria live in the oxygenated root zone or perhaps the sub-surface region inside the roots. The vascular system of the rice plant facilitates the transport of the remaining CH_4 to the atmosphere through the aerenchyma.

The observed seasonally averaged CH_4 emissions vary between ~1 and $40\,mg\,m^{-2}\,hr^{-1}$. Several known factors determine the flux from any particular rice field or region. The most important factors that control emissions from rice fields over large spatial scales of the size of a country are water management and the supply of organic material. As expected, more organic amendments lead to more emission, and intermittent flooding leads to less emission as the anaerobic conditions are reduced. Chemical and microbiological soil properties control CH_4 production in the wetland ecosystems (Mitra et al. 2002).

The relative contributions of two methanogenic pathways to total CH_4 production can be quantified by the stable carbon isotopic signatures of CO_2 (Conrad 2005). Hydrogenotrophic methanogenesis leads to the production of more negative ^{12}C-enriched CH_4 than acetate-dependent methanogenesis (Krüger et al. 2002; Nakagawa et al. 2002). Stable and radiocarbon isotopes become important tools for studying CH_4 production and secondary isotope fractionation processes (CH_4 oxidation and transportation processes).

Plant growth controls net emissions by determining how much substrate will be available for either methanogenesis or methanotrophy (Matthews and Wassmann 2003). CH_4 emissions correlate strongly with plant growth in a Texas rice field (Sass et al. 1990). Any climate change scenario, that results in an increase in plant biomass in rice agriculture, may increase CH_4 emissions (Xu et al. 2004).

There is a considerable disparity in the published results on the rate of CH_4 that is oxidized before emitted. Studies, that determine the oxidation as the difference between measured flux and production rates, have shown high rates of oxidation between 60 and 90% (Schütz et al. 1989; Sass et al. 1990; Khalil et al. 1998a). But data from other direct methods in some cases show oxidation rates, as low as 7% in the study by Groot et al. (2003) and $< 30\%$ reported by van der Gon and Neue (1996). From the review by Groot et al. (2003), it is evident that oxidation rates varied widely both across methods and within methods. The in situ isotope ratio approach presented by Tyler et al. (1997) and Chanton et al. (1997) was to empirically determine the fraction of CH_4 oxidized during the growing season in rice fields by comparing $^{13}C/^{12}C$ in CH_4 dissolved in the below ground production zone that emitted through the plant. Enrichment with both ^{13}C and ^{12}C occurs during CH_4 oxidation by the methanotrophic bacteria. Using these methods, studies have determined in situ oxidation in the rice paddies which varied significantly (20–60%) and in general, it increases with the growing season (Tyler et al. 1997; Chanton et al. 1997; Bilek et al. 1999), while Krüger et al. (2002) reported that CH_4 oxidation was quantitatively important at the beginning of the season, but decreased later by combining isotope mass balances and in situ inhibition experiments with difluoromethane (CH_2F_2), as specific inhibitor of methanotrophic bacteria.

The most obvious limiting factor for methanotrophs is the availability of CH_4 and O_2. Another important factor for methanotrophs is the availability of N sources (Eller and Frenzel 2001). Recent findings of positive effects of NH_4^+ fertilization on methanotrophs in rice paddies (Bodelier and Laanbroek 2004) are in contrast to the results of other studies, in which $NH_4{}^+$ fertilization had an inhibitory effect on CH_4 oxidation in soils (Gulledge et al. 1997) and dryland rice fields (Dubey and

Singh 2000). The elevated CH_4 concentrations even in the rhizosphere in a flooded rice field lead to less competitive inhibition of the CH_4 monooxygenase by NH_4^+ (Cai and Mosier 2000). This can be further supported by the fast uptake of NH_4^+ and other ions by rice plants and no toxic compounds (e.g. NO_2^-) accumulated in the porewater.

The only known group of bacteria able to grow on CH_4 is the aerobic methylotrophic bacteria, which oxidize CH_4 with O_2 to CO_2. The methanotrophs are subdivided into two major groups, type I, which dominates at lower CH_4 concentrations and type II on higher concentration. A third group type X has characteristics of both of the other groups. The existing methanotrophs are unable to grow on atmospheric CH_4, an activity that is obviously accomplished by unknown and as yet un-isolated bacteria in soils (Conrad 1996). CH_4 oxidation is also important under anoxic conditions. The data usually show a coincidence of anaerobic CH_4 oxidation with sulfate reduction as well as ferric iron reduction in the sub-soil of the rice fields (Miura et al. 1992). Current study shows that an anaerobic oxidation of CH_4 can lead to denitrification with microbial mediation (Raghoebarsing et al. 2006; Xiong et al. 2007). However, microorganisms able to consume CH_4 in the absence of O_2 have not been isolated so far.

The area of rice grown globally is forecast to increase by 4.5% by 2030 (FAO 2003). However, CH_4 emissions from rice production are not expected to increase substantially. Instead, there may even be reductions, if more rice is grown under intermittent flooding, or if new rice cultivars that emit less CH_4 are developed and adopted (Wang et al. 1997). Emissions during the growing season can be reduced by various practices (Yagi et al. 1997; Khalil et al. 1998d; Adhya et al. 2000; Wassmann et al. 2000; Aulakh et al. 2001). For example, draining wetland rice once or several times during the growing season reduces CH_4 emissions (Li et al. 2002; Yan et al. 2003, 2005; Khalil and Shearer 2006). This benefit, however, may be partly offset by increased N_2O emissions (Akiyama et al. 2005), and this practice may even be constrained by water non-availability for flooding again. Rice cultivars with low exudation rates could offer an important CH_4 mitigation option (Aulakh et al. 2001). In the off-rice season, CH_4 emissions can be reduced by improved water management, especially by keeping the soil as dry as possible and avoiding water logging (Cai et al. 2000, 2003; Xu et al. 2003). CH_4 emissions can be reduced by adjusting the timing of organic residue additions (Xu et al. 2000; Yan et al. 2005), by composting the residues before incorporation, or by producing biogas for use as fuel for energy production (Wassmann et al. 2000).

6.1.1.2 Decreasing Emission Trends and Approaches to Reduce Uncertainty

The net rate of CH_4 emissions is generally estimated from three approaches: (1) the category based approach-extrapolation from direct flux measurements and observations, (2) process-based modelling (bottom-up approach) (Li et al. 1992a, b, 2004; Huang et al. 1998) and (3) inverse modelling that relies on spatially distributed, temporally continuous observations of concentration, and in some cases, isotopic composition in the atmosphere or aircraft and satellite observations (top-down approach)

(Frankenberg et al. 2005). The "categories" are chosen so that within each category, the emissions are expected to be about the same with some range of uncertainty obtained from direct measurements. In this approach, there are two sources of uncertainty: the reliability of assumption of constant emissions within the category and finding the spatial distribution of each category (Khalil 1992). The process models calculate the expected emissions based on a number of inputs that characterize the area under consideration. The actual field measurements are used indirectly to validate or define the values of the various parameters of the models. There are important overlaps and similarities between these two approaches. The process model requires much of the same environmental information as the category approach, such as organic inputs and water management. Obstacles to extensive application of the top-down approach include inadequate observations and insufficient capabilities of the models to account for error amplification in the inversion process and to simulate complex topography and meteorology (Dentener et al. 2003; Mikaloff Fletcher et al. 2004; Chen and Prinn 2006). Due to isotopic fractionation associated with CH_4 production and consumption processes, CH_4 emitted from each source exhibits a measurably different $\delta^{13}C$ value. Therefore, it is possible to constrain further the sources of atmospheric CH_4 using mixing models, but such data are even more limited (Lassey et al. 2000; Mikaloff Fletcher et al. 2004).

The global average CH_4 concentrations are reaching stable levels and the trends are approaching zero (Dlugokencky et al. 1998, 2003). This pattern of slowdown and a persistent fall in the trend over two decades is characteristic of a constant global source and not necessarily a decreasing one (Khalil et al. 2007). For a conversion factor of 2.78 Tg CH_4 per ppb and an atmospheric concentration of 1,774 ppb, the atmospheric burden of CH_4 in 2005 was 4,932 Tg, with an annual average increase (2000–2005) of about 0.6 Tg yr^{-1}. Total average annual emissions during the period considered here are approximately 582 Tg CH_4 yr^{-1}.

Total emissions are not increasing, but partitioning among the different sources may have changed. Bousquet et al. (2006) have argued that the present stable concentrations are merely a temporary condition brought about by a drought that has decreased wetland emissions offsetting increasing industrial emissions. These sources include pipeline losses in natural gas distribution systems, coal mining and drilling for oil and gas. Once the rainfall returns to normal, CH_4 concentrations will start increasing again. For rice fields, it seems that both the area of rice harvested and the average emissions from a hectare of rice grown have declined in various rice growing regions (Khalil and Shearer 2006).

It is possible that there are sources that are not yet identified at this time and may still affect the validity of our understanding of the CH_4 budget (Frankenberg et al. 2005). Recently Keppler et al. (2006) proposed that terrestrial plants are a major source of 60–240 Tg CH_4 yr^{-1} via an unidentified process. Some reports have shown that the global source from living plants, if it exists, is likely to be 20–50 Tg yr^{-1} and not as large as originally thought, but has a potential for contributing to future trends (Ferretti et al. 2006; Houweling et al. 2006; Kirschbaum et al. 2006; Parsons et al. 2006; Butenhoff and Khalil 2007). However, Dueck et al. (2007) have presented results that do not show any emissions of CH_4 from plants. It may be concluded that while plants may be a source of methane, the contribution to the global source is unknown at present.

6.1.2 *Nitrous Oxide (N_2O)*

6.1.2.1 Introduction

Despite its low concentration in the atmosphere, N_2O is the third largest greenhouse gas contributor to overall global warming, after CO_2 and CH_4. N_2O is also the major source of ozone-depleting nitric oxide (NO) and nitrogen dioxide (NO_2) in the stratosphere (Crutzen 1970). Control of N_2O is part of efforts to curb greenhouse gas emissions under the Kyoto Protocol. Once emitted, N_2O remains in the atmosphere for approximately 120 years before removal, mainly by destruction in the stratosphere (IPCC 2007).

The global atmospheric N_2O concentration has increased by 9.4% since pre-industrial times and continues to increase by 0.6 ppbv yr^{-1} (Khalil et al. 2002). From a pre-industrial value of about 270 ppb, its atmospheric level has reached to 319 ± 0.12 ppb in 2005 (IPCC 2007). Global emissions are about 17.3 (15.8–18.4) and 17.7 (8.5–27.7) Tg N yr^{-1} estimated by top-down and bottom-up methods, respectively. The growth rate of about 0.3% per year (WMO 2003) has been approximately constant for the past few decades. The annual source of N_2O from the Earth's surface has increased by about 40 to 50% over pre-industrial levels as a result of human activity (Khalil and Rasmussen 1992; Hirsch et al. 2006). More than a third of all N_2O emissions are anthropogenic and are primarily due to agriculture.

The individual N_2O source strengths are poorly understood at present and hence contribute to the largest uncertainties in N_2O's global budget. This is because the anthropogenic emissions are from many small sources, each one is very uncertain, as is the case with distributed small sources of any gas. Moreover, it is also possible that there are undiscovered sources and sinks, but the sink side seems to be better defined. The destruction of N_2O in the stratosphere causes enrichment of its heavier isotopomers and isotopologues, which helps to differentiate stratospheric and surface flux influences on tropospheric N_2O (Butenhoff and Khalil 2007; Morgan et al. 2004). A current estimates of N_2O emission from various sources have been in Table 6.2.

Agriculture remains the single biggest anthropogenic N_2O source (Bouwman et al. 2002a; Smith and Conen 2004; Del Grosso et al. 2005). Human activity has increased N supply to coastal and open oceans, resulting in N_2O emissions (Naqvi et al. 2000; Nevison et al. 2004; Kroeze et al. 2005). The results of various studies, that quantified the global N_2O emissions from coastal upwelling areas, continental shelves, estuaries and rivers, suggest that the coastal areas contribute 0.3–6.6 TgN yr^{-1} of N_2O or 7–61% of the total oceanic emissions (Nevison et al. 2004; Kroeze et al. 2005). Land use change continues to affect N_2O and NO emissions (Neill et al. 2005).

Table 6.2 Global sources of N_2O (Tg N yr^{-1})

Sources	Khalil and Rasmussen (1992)[a]	IPCC (2007)
Anthropogenic sources		
Fossil fuel combustion	0.5 (0.1–1.3)	0.2[b]
Industrial processes	0.45	0.5[b]
Agriculture	0.9 (0.5–2.5)	2.8 (1.7–4.8)[c]
Biomass and biofuel burning	1.0 (0.1–1.9)	0.7 (0.2–1.0)[c]
Human excreta	1.0 (0.2–1.9)	0.2 (0.1–0.3)[c]
Rivers, estuaries, coastal zones	0.5(0.1–1.4)	1.7 (0.5–2.9)[d]
Atmospheric deposition	–	0.6 (0.3–0.9)
Land use change	0.45	
Anthropogenic total	5.0	6.7
Natural sources		
Soils under natural vegetation	6.8 (6.6–7.0)	6.6(3.3–9.0)[c]
Oceans	3.6 (2.8–5.7)	3.8 (1.8–5.8)[e]
Atmospheric chemistry	0.4 (0.3–1.2)	0.6 (0.3–1.2)[f]
Natural total	10.8	11.0
Total sources	15.8	17.7 (8.5–27.7)
Stratospheric sink	12.3 (9.0–16.0)	12.5 (10.0–15.0)
Atmospheric increase	3.9 (3.1–4.7)	3.5

[a] Re-calculated in Tg N yr^{-1}, anthropogenic total including global warming source of 0.2 Tg N yr^{-1}. Natural sources from Bouwman et al. (1995); stratospheric sink and atmospheric increase from Houghton et al. (1995).

[b] Van Aardenne et al. (2001).

[c] Bouwman et al. (2002a) for the 1990s.

[d] Kroeze et al. (2005), Nevison et al. (2004); estimated uncertainty is ±70% from Nevison et al. (2004).

[e] Nevison et al. (2003, 2004), combining the uncertainties in ocean production and oceanic exchange.

[f] Mosier et al. (1998), Kroeze et al. (1999), Olivier et al. (1998): a single value indicates agreement between the sources and methodologies of the different studies.

Hirsch et al. (2006) inversion results further suggest that N_2O source estimates from agriculture and fertilizer may have increased markedly over the last three decades when compared with an earlier inverse model estimate (Prinn et al. 1990). Applications of N fertilizer will further increase in order to feed some 8 billion people over the next 30 years (Zhu et al. 2005). Therefore, agricultural N_2O emissions are projected to increase by 35–60% up to 2030 due to increased nitrogen fertilizer use and increased animal manure production (FAO 2003). Similarly, US-EPA (2006) estimates that N_2O emissions will increase by about 50% by 2020 relative to 1990.

6.1.2.2 Processes Controlling Production and Approaches to Reduce Uncertainty

Crop production, wherein anthropogenic reactive nitrogen is created and consumed, inevitably causes N_2O emissions directly and indirectly (Mosier et al. 1998). For example, up to 71% of the annual total N_2O released from anthropogenic reactive nitrogen is presently due to direct emission in Asia (Zhu et al. 2005). N_2O is often enhanced where available nitrogen (N) exceeds plant requirements, especially under wet conditions (Smith and Conen 2004; Oenema et al. 2005).

N_2O is produced during N transformation in soils, mainly by the microbial processes of nitrification and denitrification (Freney 1997) as well as chemo-denitrification (van Cleemput 1998) and fungal transformations. These microorganisms operate under various optimum conditions, and it is generally assumed that nitrification is an aerobic process, and denitrification is an anaerobic process (Granli and Bøckman 1994; Conrad 1996; Bouwman et al. 2002b). Techniques, such as selective nitrification inhibitor and ^{15}N labeling (Müller et al. 2004) have been used to identify the N_2O production processes. Another way to identify the processes producing N_2O is to monitor the changing isotopic composition of N_2O based on the findings that the stable isotope ratio of $^{15}N/^{14}N$ of denitrifier-derived N_2O can differ from that of nitrifier-derived N_2O (Yoshida 1988; Kim and Craig 1993; Webster and Hopkins 1996).

The microbial basis of N_2O production in soil is more complicated than that of CH_4. Production of N_2O has been demonstrated among bacteria that respire nitrate to nitrite and those that dissimilate nitrate to ammonium (DNRA) (Stevens et al. 1998; Xiong et al. 2007). Since denitrifiers, nitrate respirers, and DNRA cannot be differentiated in situ, and also autotrophic and heterotrophic nitrifiers are not differentiable, the exact microbial basis of N_2O production and consumption is still not clear (Conrad 1996). Moreover, these physiologically defined groups are obviously widespread among the various taxa, bacteria, and fungi.

Though N_2O is produced and consumed by defined reactions in individual microorganisms, much of the spatial and temporal variability of N_2O fluxes between the soil and the atmosphere can often be explained by only a limited number of factors (Conrad 1996; Bouwman et al. 2002b). Any factor that affects the equilibrium of N transformation in soil, could result in N_2O emissions. Nitrification is the predominant N_2O-producing process under moderately moist and warm conditions, while denitrification is the predominant process under wet (anaerobic) conditions when NH_4^+ and NO_3^- are available in soil. Both empirical models based on various physical and chemical parameters and mechanistic models that differentiate between nitrification and denitrification are successful without considering the structure of the microbial community (Li et al. 1992a, b, 2004; Bouwman et al. 2002a). General denitrification models have shown that N_2O production in soil is mainly controlled by the availability of nitrate, labile C compounds, and O_2 (Del Grosso et al. 2000). Soil nitrate content and soil water content are the key factors affecting denitrification and N_2O emissions from the agricultural soils (Vallejo et al. 2001; Dobbie and Smith 2003).

The potential source of N_2O from crop fields is the volatilization of ammonia, some of which is converted to N_2O in the atmosphere (Dentener and Crutzen 1994; Warneck 2000). The N_2O channel is 10–60% efficient according to a review by Atkinson et al. (2004). However, there is no laboratory or field observation of this mechanism till now.

Another new source of N_2O is emission from plants (Chen et al. 1997; Goshima et al. 1999; Zhang et al. 2000; Xu et al. 2001; Smart and Bloom 2001; Hakata et al. 2003; Zou et al. 2005). Many researches demonstrated the role of growing plants in N_2O production and emissions from agricultural systems (Chang et al. 1998; Müller 2003). Some studies showed a role of plant pathway in ecosystem N_2O emissions (Mosier et al. 1990; Yu et al. 1997; Chang et al. 1998; Yan et al. 2000). This source remains yet unquantified.

6.1.2.3 Characteristics of N_2O Emissions

Croplands are usually fertilized and therefore not likely to be sinks for N_2O, though some studies report N_2O uptake in fertilized fields (Xu et al. 1997; Kroeze et al. 2007). The crops usually don't efficiently utilize N fertilizers and other N sources (Zhu and Chen 2002; Galloway et al. 2004; Xiong et al. 2008). The surplus N is particularly susceptible to emission as N_2O (Velthof et al. 1996; McSwiney and Robertson 2005; Xiong et al. 2006a).

N_2O emission from croplands at site scales occurs essentially with great spatial and temporal variability (Veldkamp and Keller 1997; Dobbie and Smith 2003). Spatial and temporal variability is mainly caused by heterogeneity in soil properties and agricultural management (e.g. water, nutrient, crop, tillage, and residue management) (Veldkamp and Keller 1997; Brown et al. 2002; Dobbie and Smith 2003). Agronomic practices, such as tillage and fertilizer applications, can significantly affect the production and consumption of N_2O because of alteration in soil physical, chemical, and biochemical activities. Tillage could cause immediate changes in microbial community structure as reported by Jackson et al. (2003). They produce large N_2O emissions at the beginning of crop season (Xiong et al. 2006a). It's likely that some emission pulses would have been missed due to the insufficient sampling frequency since N_2O emissions mainly occur in pulses (Zheng et al. 2000; Xing et al. 2002; Zheng et al. 2004; Xiong et al. 2002a, b, 2006a).

Improving N use efficiency can reduce N_2O emissions (Cole et al. 1997; Paustian et al. 1998; Robertson et al. 2000; Dalal et al. 2003; Monteny et al. 2006). By reducing leaching and volatile losses, improved efficiency of N use can also reduce off-site N_2O emissions (i.e. indirect emissions) (Xiong et al. 2006b). Adopting reduced or no-till may affect N_2O emissions, but the net effects are inconsistent and not well-quantified globally (Smith and Conen 2004; Li et al. 2005). The effect of reduced tillage on N_2O emissions may depend on soil and climatic conditions (Helgason et al. 2005). In some areas, reduced tillage promotes N_2O emissions, while elsewhere, it may reduce emissions or have no measurable influence. Drainage of croplands in humid regions can suppress N_2O emissions by improving aeration (Monteny et al. 2006). Any nitrogen lost through drainage, however, may be susceptible to loss as N_2O (Reay et al. 2003). The use of rotations with legume crops is an important example to reduce reliance on external N inputs, although legume-derived N can also be a source of N_2O (Xiong et al. 2002a, b; Rochette and Janzen 2005).

6.1.2.4 CH_4 and N_2O Interactions in Rice Paddies

The fluxes of N_2O from paddy soil are small compared to aerated soils, because a larger percentage of the produced N_2O is further reduced to N_2 by denitrifiers. Submerged rice fields seem to act occasionally even as a sink for atmospheric N_2O (Minami and Fukushi 1984; Xu et al. 1997). The direct emissions of N_2O from rice agriculture are variable and robust global estimates are not yet possible. Since seasonal emission varied from ~ 1 to $150\,\mathrm{mg\,m^{-2}\,yr^{-1}}$, it accounts for a range of potential global emissions from 0.01 to $2\,\mathrm{Tg\,yr^{-1}}$ (Xu et al. 1997; Khalil et al. 1998b; Suratno et al. 1998; Abao et al. 2000).

Several agricultural practices and conditions, that favor reduced emissions of CH_4 from rice fields, tend to increase N_2O emissions (Lindau et al. 1990; Bronson et al. 1997b; Xu et al. 1997; Cai et al. 1997; Abao et al. 2000; Hou et al. 2000; Xiong et al. 2007). However, the conditions, that affect emissions of these two gases, vary widely. For example, alternate anaerobic and aerobic cycling increases N_2O emissions relative to constant aerobic or anaerobic conditions (Granli and Bøckman 1994; Chen et al. 1997; Tsuruta et al. 1997; Zheng et al. 1997; Xing 1998; Xing et al. 2002). In contrast, intermittent irrigation of rice paddies, which causes anaerobic and aerobic cycling, is considered to be one of the options for reducing CH_4 emissions (Yagi et al. 1996; Chen et al. 1997; Zheng et al. 1997; Cai et al. 2003; Huang et al. 2004). Antecedent water regime of a soil, independent of the current water status, also affects N_2O emissions (Xing et al. 2002) and CH_4 emissions (Xu et al. 2000, 2003). After the rice is harvested and the fields are no longer inundated, varying amounts of N_2O emissions are observed, but no CH_4 emissions occur (Bronson et al. 1997a, b; Xu et al. 1997; Khalil et al. 1998b). In China, 86% of the rice fields are under a rice-upland cropping system and mid-season drainage is widely adopted in rice cultivation.

Mitigation is one aspect of the relationship between CH_4 and N_2O emissions from rice fields that has been widely studied (Bronson et al. 1997a; Wassmann et al. 2000; Breiling et al. 2005). The question of whether the agricultural practice of intermittent flooding of rice fields causes a net reduction of non- CO_2 greenhouse gas emissions has been addressed, and it is argued that even though N_2O has a global warming potential some 10 times greater than CH_4, the radiative forcing of the additional N_2O is only partially off-set by the reduction of CH_4 emissions (Bronson et al. 1997a; Breiling et al. 2005). This finding is used to promote the practice of intermittent flooding as a means for mitigating greenhouse gas emissions from rice fields.

6.1.3 Carbon Dioxide (CO_2)

CO_2 is the most important anthropogenic greenhouse gas. The global atmospheric concentration of CO_2 has increased from a pre-industrial value of about 280 to 379 ppm in 2005 (IPCC 2007). CO_2 emissions from the agricultural soils are usually included in the land use, land use change and forestry sector. Land use change is the main mechanism by which agriculture can indirectly contribute to the increase of CO_2. So there are a few comparable estimates of emissions of this gas from agriculture. CO_2 is released largely from microbial decay or burning of plant litter and soil organic matter (Janzen 2004). Agricultural lands generate very large CO_2 fluxes both to and from the atmosphere, but the *net* flux is small or zero depending on the conditions (Paustian et al. 1998). Another source of CO_2, that can be attributed to the agriculture, is the energy used to grow and harvest the food produced, but there are no global estimates. This source, however, is included in the estimates of fossil fuel source of CO_2.

Soil carbon sequestration has strong synergies with sustainable agriculture and generally reduces vulnerability to climate change. Stored soil carbon may be vulnerable to loss through both land management change and climate change. The conversion of land from forested to agricultural land can have a wide range of effects as far as CO_2 emissions are concerned (Hymus and Valentini 2007). Soil disturbance and increased rates of decomposition in converted soils can both lead to emissions of CO_2 to the atmosphere, and increased soil erosion and leaching of soil nutrients further reduce the potential of the area to act as a sink for atmospheric carbon. Similarly, land reclamation and changes in land use management can affect an increase in terrestrial carbon uptake. Current estimates suggest that such land-use changes lead to the emission of $1.7\,\mathrm{Pg\,C\,yr^{-1}}$ in the tropics, mainly as a result of deforestation, and to a small amount, of uptake ($\sim 0.1\,\mathrm{Pg\,C}$) in temperate and boreal areas, thus producing a net source of $\sim 1.6\,\mathrm{Pg\,C\,yr^{-1}}$.

6.2 Prospectives

The global exchange rates of trace gases between the soils and the atmosphere are complicated, since the production and destruction processes involved are heterogenous. Even the same soils may be a source at one time and sink at another time of the year (Conrad 1996; Khalil and Rasmussen 2000).

Increased depositions of atmospheric NOx and NH_3 may influence the exchange of the three main greenhouse gases – CO_2, N_2O, CH_4–between biosphere and atmosphere. The net effect of anthropogenic nitrogen deposition on the net GHG budget is the resultant of complex interactions and ecosystem feedbacks, and is highly dependent on the local environmental conditions (van der Gon and Bleeker 2005; De Vries et al. 2007).

An additional mitigation of ~ 0.2 Pg C $\mathrm{yr^{-1}}$ could be achieved by 2030 by improved energy efficiency in agriculture through reduced fossil fuel use (Smith et al. 2007). There are 4.3 million harvester-threshers and 27.6 million tractors used for agricultural production in 2003 (FAOSTAT 2006). However, this is usually counted in the relevant user sector rather than in agriculture and so is not considered further here. Agricultural crops and residues are increasingly seen as sources of feedstock for energy industry to reduce fossil fuel, use (Paustian et al. 1998; Edmonds 2004; Cerri et al. 2004). These products can be burned directly, but can also be processed further to generate liquid fuels such as ethanol or diesel fuel. So, GHG emissions from agriculture could also be reduced in this way. The net benefit to atmospheric CO_2, however, depends on energy used in growing and processing the bio-energy feedstock (Spatari et al. 2005). The competition for other land uses and the environmental impacts need to be considered when planning is made to use energy crops.

6.3 Conclusions

Crop fields are a dynamic component of the global carbon and nitrogen cycle. Rice field plays an important role in both CH_4 and N_2O emissions. N_2O emissions will play an increasing role since the N fertilizer is the requisite for food production to meet the growing population. Agricultural fields show a significant potential for mitigating global greenhouse gas emissions while maintaining the crop yields.

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Chapter 7
Environmental Parameters Influencing the Methane Emissions in the Pantanal Floodplain, Brazil

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7.1 Introduction

Methane (CH_4) is a trace gas with an important role in both troposphere due to the reactions with the hydroxyl radical and the formation of organic radicals, and the stratosphere, because of its participation in the chlorine and water vapor chemistry. Methane is also an important greenhouse gas, with a relative contribution of about 20% for the global greenhouse effect (Wuebbles and Hayhoe 2002). Many works have evidenced that an increase of the methane concentration began around 1800 with the industrial era, and the consequent increase of human activities, including the utilization of fossil fuels, cattle raising and rice plantations (Blake and Rowland 1988; Stern and Kaufmann 1996; Dlugokencky et al. 1998; Etheridge et al. 1998). Presently, the CH_4 concentration is about 1,770 ppbv (part per billion by volume), but it presents a large inter-annual variation caused by eruptions, such as one of the Mt. Pinatubo, or by alterations of the wetlands emissions, mainly due to alterations in the precipitation and temperature regimes in the range between 30° and 90° N (Dlugokencky et al. 1996; Walter et al. 2001).

Global budget evaluations indicate that natural wetlands have a substantial contribution to tropospheric methane, estimated as $100\,\mathrm{Tg}\ CH_4\ y^{-1}$, or about 20% of the global emission (Wuebbles and Hayhoe 2002). Although most of the wetlands are located in the temperate region of the Northern Hemisphere (Lehner and Döll 2004), many experiments have pointed that the wetlands in the tropical region emit $66\,\mathrm{Tg}\ CH_4\ y^{-1}$, or 60% of the emissions from the natural wetlands (Bartlett and Harriss 1993). The emission of methane from the wetlands is influenced by many environmental factors, such as temperature, dissolved organic carbon (DOC), pH, oxyreduction potential (pO), water depth, and floating vegetation. Qualitative relations between the methane fluxes and the factors have been determined, but direct relations are more difficult to establish due to the complex mechanisms involved in

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the process, starting with the production in the sediment, which is followed by the transport though the water column and the subsequent liberation to the atmosphere. Many experiments have been performed in the tropical region, especially in the Amazonian wetland, showing that it is an important source of the global budget of atmospheric methane, with an emission range estimated from 1.73 to 21.0 Tg CH_4 y^{-1} for the whole Amazon basin (Devol et al. 1990; Bartlett et al. 1990; Melack et al. 2004).

Previous work in wetlands indicated that the transport of methane from the sediments to the water surface might occur by three main pathways, which are dependent on each ecosystem: (i) diffusion through the water column, (ii) ebullition from the sediments and (iii) transport through the plant stems (Bartlett et al. 1990; Devol et al. 1990; Keller and Stallard 1994; Bastviken et al. 2004). In Amazonia, the ebullition transport is very important, sometimes accounting for more than 60–70% of the total emission to the atmosphere, while the transport by plants is dependent on the plant type and how it covers the surface.

Present estimates of methane emission from tropical wetlands have an excessive dependence on data from Amazonia, with insufficient information of seasonal and spatial variation in the emission from other tropical wetlands, as evidenced by Smith et al. (2002). There are a few studies covering some important tropical and subtropical wetlands, which are not yet characterized.

One of the most important wetlands in Brazil is the Pantanal, which is covered by savanna-like vegetation and is flooded seasonally. Its total area is estimated about 138,183 km^2, with most of this area within Brazil, in the upper Paraguay River basin. Flooding in the region is clearly seasonal, with its maximum flooded area occurring at the end of March. Marani and Alvalá (2007) showed flux measurements for one year, and they identified the importance of the Pantanal floodplains as a source of methane to the atmosphere, with bubble fluxes constituting the major contribution to the general flux. This work presents a study of flux data obtained in five Pantanal sites during 2004 and 2005, and evaluates emissions from floodplains and lakes and their possible relationship with some environmental variables.

7.2 Methodology

7.2.1 Methane Flux

Methane fluxes were determined using the static chamber technique, described by Devol et al. (1988, 1990) and Bartlett et al. (1988). The chambers were covered with a thermal and reflective sheet to avoid temperature variations, had an area of 0.066 m^2 and a volume of 26 l. Inside the chamber, a small fan was installed to avoid any air stratification and was turned on at least 30 s before the sampling. The chambers were placed in the sampling site using a boat, with care to avoid perturbations in the water surface and surrounding vegetation. All samples were taken between 11:00 and 16:00 LT (local time), and they were done under conditions of almost no wind. Every 6 min, during 18 min, gas samples were removed through a septum with a 60 ml polyethylene syringe equipped with a 3-way polypropylene stopcock. To verify the linearity and possible perturbation in the chamber deployment, a sample was taken in the first minute after the chamber was placed. Six to eight locations were sampled for each site. Environmental variables that may affect the methane emissions were also measured: water depth, water and air temperatures, and pH.

The methane concentration of all samples was determined with a commercial gas chromatograph (Shimadzu, GC-14A), equipped with a flame ionization detector (FID), a 2.2 ml sample loop and two stainless steel columns that were optimized to perform methane analysis in the Ozone Laboratory at INPE, São José dos Campos, Brazil. The first column was packed with silica gel (2.5-m long and 1/8″ diameter) and it was used to remove the water vapor, CO_2 and others heavy organic compounds from the samples, in order to reduce the total retention time. The analysis column (3.0-m long and 1/8″ diameter) was packed with a zeolite 5 Å molecular sieve. The methane standard (1,749.4 ± 4.5 ppbv) used for calibration was acquired from the Climate Monitoring and Diagnostic Laboratory of the National Oceanic and Atmospheric Administration (CMDL/NOAA).

For each syringe, three aliquots were analyzed with a relative precision of 0.7% or better. The minimum detectable methane flux was about 1 mg CH_4 m^{-2} d^{-1}. The methane flux was determined from the temporal variation of its mixing ratio inside the chamber during the sampling time, following the method of Schiller and Hastie (1994). The methane fluxes were considered diffusive, if the linear correlation between the mixing ratio change and the elapsed time showed a correlation coefficient (r^2) greater than 0.90 (Sass et al. 1992). A second criterion was that the initial concentration obtained by the linear regression (at time $t=0$) must be close to the measured environmental air concentration. If the flux did not follow the first criterion, and if an abrupt increase of the methane concentration did occur after the first sampling, the flux variation was interpreted as an ebullition. The rate of change of the methane mixing ratio was then determined by subtracting the environmental methane mixing ratio from the mixing ratios obtained at the end of the sampling time, divided by the enclosure time, as proposed by Cicerone et al. (1992) and Keller and Stallard (1994).

7.2.2 Site Location

The Pantanal is a hydrological complex plain constituting a large sedimentary basin, which is periodically flooded by the Paraguay River and its tributaries. Its altitude varies between 80 and 120 m, with a total area estimated 138,183 km^2, located mostly within Brazil, but with small areas in Paraguay and Bolivia. The Paraguay River and its tributaries carry continuously organic material and during the flood period, the water overflow spreads the sediments over the entire region. This organic material that is deposited in the lakes and in the floodplain is the most important source of nutrients for the methanogenic bacteria. The total area under flood

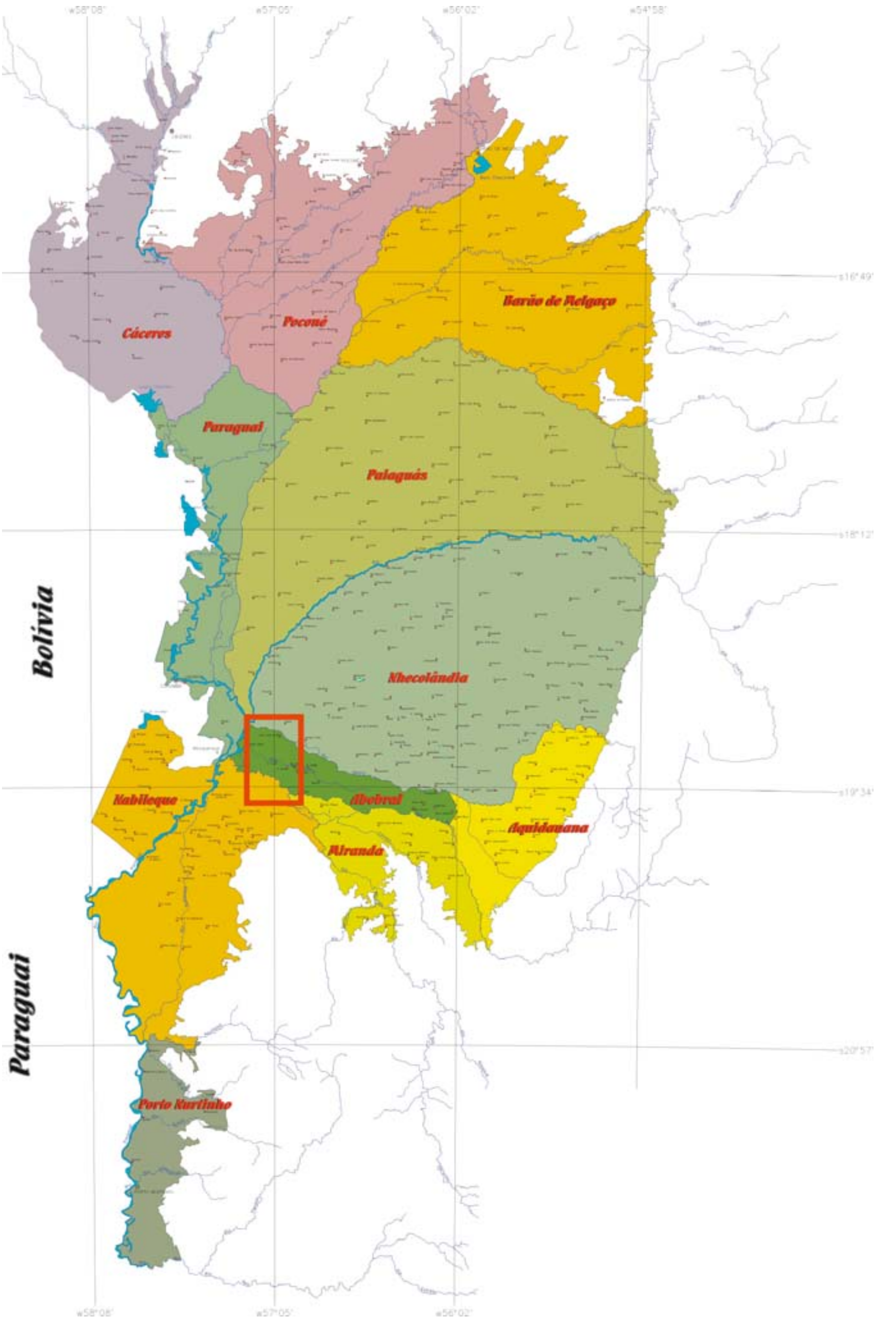

Fig. 7.1 The Brazilian Pantanal basin and its sub-division. The measurement was performed inside the rectangle in the Abobral region

conditions, including the floodplain, lakes, rivers, and channels that link lakes and rivers, was estimated by Hamilton et al. (2002) as 130,920 km^2, thus constituting the largest floodplain in South America. Using a 100-year data set of the Paraguay River level (from 1900 to 1999), Hamilton et al. (2002) also estimated an average flooded area of 34,880 km^2. The flooding of the region occurs after December, during the Southern Hemisphere summer, when the rain water that is falling on the riverhead of the basin since October reaches the region. The maximum flooded area occurs at the end of March. After April, up to the end of September, there are only a few rain episodes and the river level thus decreases to its lowest depth (Alvalá and Kirchhoff 2000). Normally, the period in which half of the maximum-flooded area stays flooded is of 172 days per year, and there is a delay of months between the summer rains and the flooding period, due to the slow passage of flood waters through the Pantanal plain.

The flux measurements were performed over water environments, consisting of open water and emergent macrophytes, at five sites near the Miranda River as shown in Fig. 7.1. Among these sites, three are considered floodplains: São João (19°24′S, 57°03′W), Baú (19°19′S, 57°03′W) and Arara Azul (19°19′S, 57°03′W); and two lakes: Mirante (19°23′S, 57°03′W) and Medalha (19°34′S, 57°00′W) where the water body is permanent, with less influence of the seasonal flooding. During the experiments, the Base de Estudos do Pantanal – BEP (19°34′S, 57°01′W), which belongs to the Universidade Federal do Mato Grosso do Sul, was used as the local logistic support. Hamilton et al. (2002) evaluated that permanent lakes and rivers in the Pantanal cover about 3,120 km^2, with small variation during the year. The remaining flooding areas of the Pantanal are characterized by vast plains subjected to seasonal flooding, mainly due to water overflow from the rivers, but also caused by local rain (Marani and Alvalá 2007). In general, these plains are shallow in comparison with the lakes and many of them dry completely during the dry season.

7.3 Results and Discussions

7.3.1 Overall Flux

Eight methane campaigns were performed during the years 2004 and 2005, thus resulting in the determination of 560 methane fluxes in two lakes and three floodplains of the Southern Pantanal Region. The overall average methane flux was 116.8 ± 276.0 mg CH_4 m^{-2} d^{-1}, with the individual measurements varying from 1 to 2,187.0 mg CH_4 m^{-2} d^{-1}. Figure 7.2 presents the frequency distribution of all fluxes on a logarithmic scale. Although the fluxes to the atmosphere show skewness toward the smaller fluxes in the logarithmic scale, a Gaussian distribution follows at the 1% level. This skewness resulted from the relative higher number of diffusive fluxes (n=342, but with lower flux values), in contrast with the ebullitive fluxes (n=218, with higher flux values).

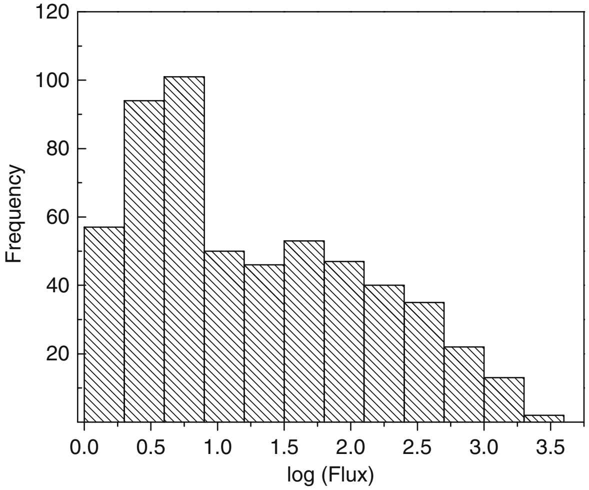

Fig. 7.2 Frequency distribution of all fluxes on a logarithmic scale

The Pantanal region was characterized by a drought during the years of the campaings, mainly in 2005, when the water maximum peak level in the Paraguai River was 3.29 m, while for the normal flooding condition, it ranges from 5.0 to 6.0 m (Embrapa 2006). In 2004, the maximum water level was 4.26 m, what characterizes a small flooding condition. This strong drought also contributed for lowering the depth of the lakes to 1.4 m in 2004, and to 0.9 m in 2005.These drought levels possibly influenced the organic matter distribution in the water.

The pH measured at 10 cm below the surface ranged from 6.4 to 9.8, and 90% of them fell in the range 6.5–7.7, which is the optimum range for the methane production by the methanogenic bacteria (Yang and Chang 1998). However, for this short range, relationships between fluxes and the pH are unlikely.

Marani and Alvalá (2007) observed differences of occurrence in the diffusive and in the ebullitive (bubbles) fluxes while considering lakes or floodplains, with a tendency of the higher occurrence of bubbles in the floodplains. While the diffusive fluxes were concentrated in the lower portion of the emission range (average: 13.1±20.7 mg CH_4 m^{-2} d^{-1}, median: 5.0 mg CH_4 m^{-2} d^{-1}), the ebullitive fluxes had higher values, with a greater dispersion (280.7±390.0 mg CH_4 m^{-2} d^{-1}, median: 128.5 mg CH_4 m^{-2} d^{-1}). This later result reflects the more intense nature of the emission, which occurred only in 39% of the measured fluxes. These higher bubble fluxes observed in the Pantanal floodplains reveal the importance of ebullitive fluxes in the emission of methane in the region when considering regional emission. Bubbling events are episodic, but when they occur, they dominate the methane release, thus resulting high average fluxes (Devol et al. 1990).

The fraction of occurrence of bubble events at different depth ranges for floodplains and for lakes is presented in Fig. 7.3. In the first ones, the bubble fluxes represented 51% of the occurrences, with a depth range from 0.1 to 1.4 m, while in the lakes they represented 27%, with a depth range from 0.1 to 3.5 m; also, in

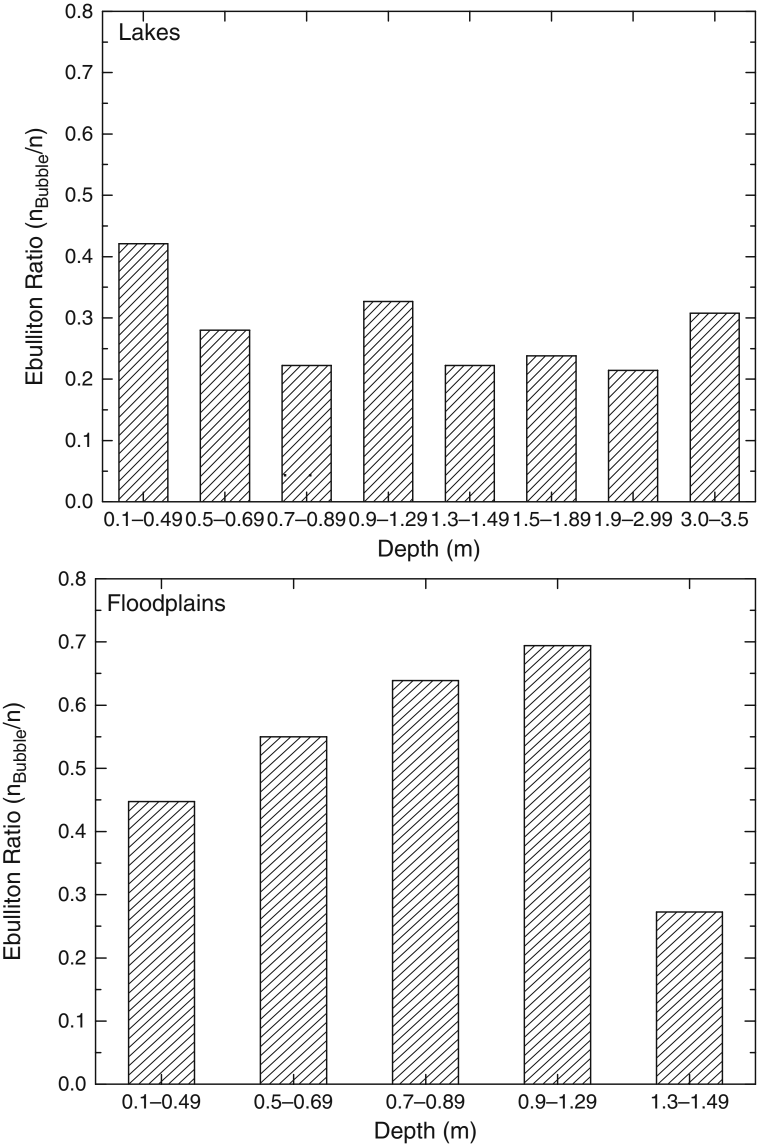

Fig. 7.3 Ratio of ebullition (i.e., fraction of fluxes measurements with detectable bubbles) at different depths to lakes and floodplains

the same range of depth (0.1–1.4 m), the relative number of ebullitive fluxes in the floodplains was higher than the one observed in the lakes.

In contrast, in Amazonia, the bubbling emission corresponded to up to 73% of the overall fluxes, with each one showing different contribution (Bartlett et al. 1990; Devol et al. 1990). Bastviken et al. (2004) observed that, in temperate lakes, there is no linear correlation between the frequency of occurrence of ebullitive fluxes and the depth; their frequency ranged from 25 to 80% of the measurements, depending on the water depth, with the higher values occurring in deep waters. Smith et al. (2002) verified that although the correlations of methane fluxes with a number of environmental variables are statistically significant, they are too weak to serve as a basis for either the prediction, or the analysis of emission mechanisms.

7.3.2 Seasonality

The seasonality of the emission rates during the wet and dry periods may be observed in Fig. 7.4, which presents the box plot diagrams for both the periods. For the average diffusive fluxes, a statistically difference ($\alpha=0.05$) was observed between the wet (18.5±26.1 mg CH_4 m^{-2} d^{-1}) and the dry (9.5±15.1 mg CH_4 m^{-2} d^{-1}) periods, with greater fluxes during the wet periods, following increases in the water temperature and depth.

Although the correlation between the fluxes and the environmental parameters was low, the statistical difference observed between both periods shows some influence on the fluxes measured. The separation of the sites between floodplains and lakes showed that the variability with the depths was experienced mainly in the lakes. In general, the floodplains present a small slope, so that in the wet period, the depth increased about 0.5 m, while in the lakes, the depths increased more than one meter. In all floodplains, the influence of seasonality was observed mainly in their area, with a large decrease during the dry period, such that the São João site was completely dry in 2005.

Depth and temperature were altered from the wet to dry season in the lakes, but their flooded area had a small increase in the wet season. The results of this work, in some aspects, are different from those obtained in Amazonia by Bartlett et al. (1990) and Devol et al. (1990), since in the Pantanal, the diffusive fluxes, although lower in both periods, showed statistic differences between the wet and the dry periods, mostly in the floodplains. Following Keller and Stallard (1994), higher temperatures and input of substrates due to flooding in the wet period, could have influenced positively the diffusive fluxes, including some effects of depth.

The ebullitive fluxes did not show a statistically significant variation from wet (average: 281.6 ± 376.3 mg CH_4 m^{-2} d^{-1}) to dry (average: 277.2 ± 405.3 mg CH_4 m^{-2} d^{-1}) season; there was only an increase in the data dispersion in the dry season. During the dry period, were observed some higher fluxes, including the two highest ones. The separation of the ebullitive fluxes in the lakes and the floodplains components shows that the average fluxes were higher in the floodplains than in the lakes, with similar values in both periods, although the observed temperature and depth were statistically different for the wet and the dry seasons. This result may be associated with the buoyancy of the bubbles in the sediments, which is related to the depth, even if a correlation was not detected statistically. One indication of this is

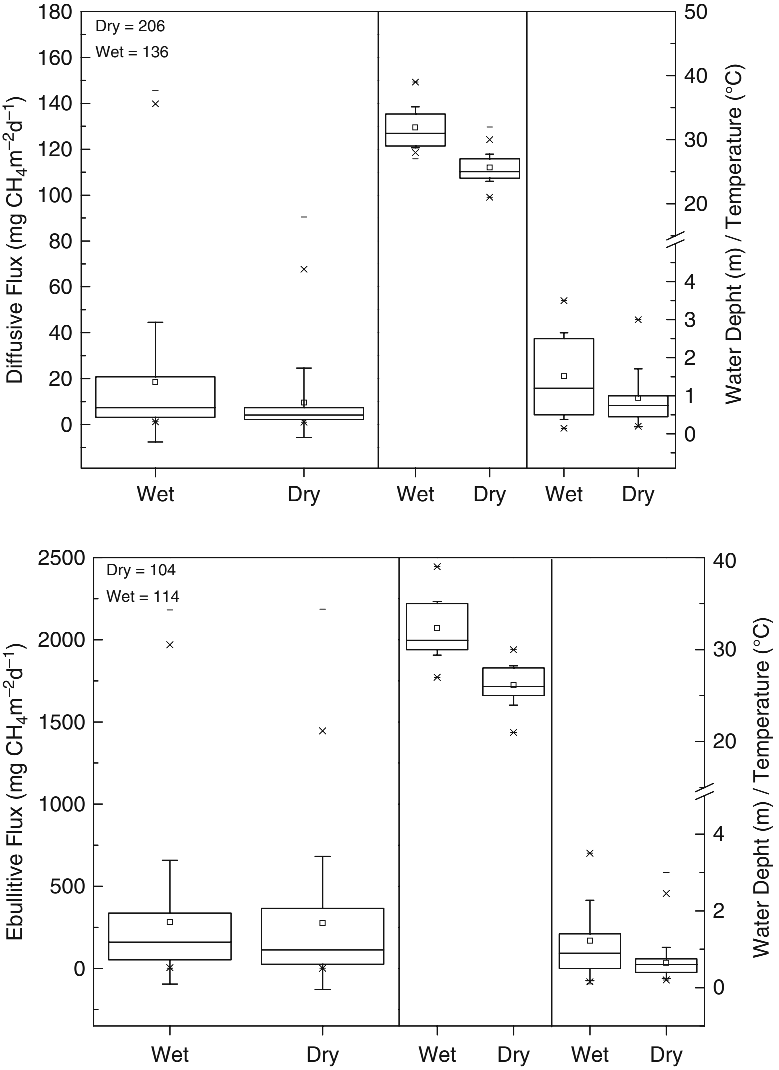

Fig. 7.4 Diffusive and ebullitive fluxes, water depth and water temperature in wet and dry seasons

that the average ebullitive flux for floodplains was 354.1 mg CH_4 m^{-2} d^{-1} (wet and dry seasons), while for lakes it was 134.3 mg CH_4 m^{-2} d^{-1}.

7.3.3 Vegetation Influence

To investigate the influence of vegetation on the methane fluxes, the diffusive and ebullitive fluxes collected over aquatic vegetation were considered separately.

Table 7.1 Diffusive and ebullitive fluxes in lakes and floodplains, separated by the presence of vegetation. Measured environmental variables are also presented

		Lakes		*Floodplains*	
		Vegetated (n=26)	*Open Water* (n=175)	*Vegetated* (n=55)	*Open Water* (n=86)
DIFFUSIVE	Flux, $mgCH_4m^{-2}d^{-1}$	12.3 ± 20.3	12.3 ± 16.0	22.6 ± 35.3	8.7 ± 13.9
	Depth, m	0,7 ± 0,3	1.6 ± 1.0	0,5 ± 0,3	0,5 ± 0,3
	Water Temperature, °C	26,8 ± 3,4	27.8 ± 3.8	29,3 ± 4,1	28,5 ± 4,4
	pH (*range*)	6.7–9.2	6.5–7.9	6.6–9.5	6.4–9.8
		Lakes		*Floodplains*	
		Vegetated (n=8)	*Open Water* (n=67)	*Vegetated* (n=60)	*Open Water* (n=83)
EBULLITIVE	Flux, $mgCH_4m^{-2}d^{-1}$	94.4 ± 94.2	139.1 ± 226.3	367.3 ± 457.5	348.7 ± 422.9
	Depth, m	0.7 ± 0.4	1.7 ± 1.1	0.6 ± 0.3	0.6 ± 0.3
	Water Temperature, °C	25.6 ± 4.6	26.6 ± 3.6	29.9 ± 4.5	29.1 ± 3.8
	pH (*range*)	6.9–7.8	6.6–7.8	6.5–9.5	6.4–9.6

A statistically significant difference (α=0.05) was observed only for the diffusive fluxes, with an average of 19.3 ± 31.5 mg CH_4 m^{-2} d^{-1} in vegetated waters (n=261) and 11.1 ± 15.4 mg CH_4 m^{-2} d^{-1} in open waters (n=81). The correlation among the fluxes and the measured variables (depth, water and air temperature), although positive, but were weak, ($r<0.4$ for all cases). Table 7.1 presents the fluxes divided in diffusive and ebullitive, for lakes and floodplains, with vegetated or open water. It shows that the vegetation influence was higher in the floodplains than in the lakes. In the floodplains, the methane diffusive flux reached an average of 22.6 ± 35.3 mg CH_4 m^{-2} d^{-1} in vegetated waters (n=55), while the average in open waters (n=86) was 8.7±13.9 mg CH_4 m^{-2} d^{-1}, thus revealing the importance of the vegetation in the methane transport in this environment.

As reported by Devol et al. (1988) and Christensen et al. (2003), the transport by the vegetation is effective when the roots are linked to the sediments, and it is also dependent on the density of the plants. In the Pantanal, the vegetation covers large extensions of the flooded areas, but in general, the vegetation is linked to the sediments mainly in the margins, where the depth is lower. The majority of the measurements in the lakes were performed in the central part, where the depth is sufficient to avoid the fixation of the roots to the sediments; so, the methane transport by vegetation was minimized. As the floodplains are shallower, the linking of the roots to the sediments is increased which facilitates the transport of methane by the stems. The comparison of the ebullitive fluxes of the open and vegetated areas did not show statistical significant differences, mainly due to the large variability observed in these fluxes.

The correlation analysis among the fluxes and the ambient variables (depth, water temperature and pH) showed that the correlations were weak ($r<0.4$) in all cases, although some of then had significant differences as seen above. The same correlations resulted from a Principal Component Analyses (PCA) of the data.

7.3.4 Regional Emission Estimates

The methane emission estimates are derived from the average fluxes and information about the flooded area. In spite of many questions about the relative contribution of the different transport mechanisms and the influence of the environmental variables about the Pantanal methane emission, this estimate may be a good indicator of the importance of the region to the global methane budget.

Hamilton et al. (2002) made estimates of flooding in the Pantanal region considering different flooding patterns. They estimated that the maximum flooding area is about 130,920 km^2, so that the Pantanal may represent the biggest flooding area in South America. From this area, about 3,120 km^2 represent open waters, like rivers and lakes, which have a small variation in their area during the year. They also estimated 172 days as the floodplain hydroperiod, which is defined as the time with the floodplain inundated above 50% of its maximum inundation area. Thus, the Pantanal has also the longest flooding period in South America.

One first crude regional estimate was determined from the average of all fluxes (116.8 mg CH_4 m^{-2} d^{-1}) and the average flooded area computed for both years (2004 and 2005), using the relation presented by Hamilton et al. (2002). This was worked out to be an emission of 1.37 Tg CH_4 y^{-1} for the whole Pantanal. Considering separately the diffusive and the ebullitive fluxes, the contributions are respectively 0.15 Tg CH_4 y^{-1} and 1.26 Tg CH_4 y^{-1}, with an annual emission of 1.41 Tg CH_4 y^{-1}. As the fluxes showed differences when they were stratified between the lakes and the floodplains, and as the vegetation increases the diffusive fluxes in the floodplains, a new evaluation may be obtained by considering the areas for the two habitats separately. Using the Hamilton et al. (2002) formulation, the floodplains area was estimated as 28,441 km^2, which leads to an emission of 1.95 Tg CH_4 y^{-1} from this environment. The lakes, with an annual flooded area of 3,120 km^2, caused an estimated methane emission of 0.25 Tg CH_4 y^{-1}. The total annual emission resulting from the sum of each component computed separately is 2.20 Tg CH_4 y^{-1}. A summary of these results is presented in Table 7.2.

For Amazonia, with measurements performed in different areas and flooding conditions, the estimates of the annual emission present a range from 2 to 21 TgCH_4 y^{-1}, which is a large range mainly due to the uncertain flooded area estimates (Bartlett et al. 1990). Based on many measurements performed in Amazonia and using remote sensing techniques, Melack et al. (2004) estimated the flooding area as 42,700 km^2, so that the revised annual methane emission from the Amazon basin was 1.73 Tg CH_4 y^{-1}. These authors also made an estimate for the Pantanal using measurements performed in Amazonian environments with savanna-like vegetation, which resulted in a mean methane flux of 95.2 mg CH_4 m^2 d^{-1}. This result is near the value obtained in this work (116.8 mg CH_4 m^2 d^{-1}) and higher than that one obtained by Smith et al. (2002) for the floodplains of the Orinoco River region (41.6 mg CH_4 m^2 d^{-1}). Melack et al. (2004) computed the flooded area using the long-term mean flooded area (34,800 km^2) estimated from Paraguay River stage records measured at Porto Ladário (from 1900 to 1999) by Hamilton et al. (2002); so, the annual emission was estimated as 3.32 Tg CH_4 y^{-1}. The results obtained above show that although there are several questions to clarify about the methane emissions from the Pantanal, this region is as important as Amazônia for the atmospheric methane balance.

Table 7.2 Annual emission of methane for each Pantanal habitat, with diffusive and ebullitive components and total flux (Tg CH_4 y^{-1})

	Diffusive	Ebullitive	Total
Pantanal (overall average area: 31,561 km^2)	–	–	1.37
Pantanal (overall average area: 31,561 km^2)	0.15	1.26	1.41
Lakes (average area: 3,120 km^2)	0.02	0.23	0.25
Floodplains (average area: 28,441 km^2)	0.04	1.91	1.95
Pantanal (lakes+floodplains)	0.06	2.14	2.20

7.4 Conclusions

Methane emission by ebullitive transport was detected only in about 40% of the measurements, but it contributed 90% to total methane release to the atmosphere in the southern Pantanal region, so confirming the early results presented by Marani and Alvalá (2007). Although a correlation between the fluxes and the environmental parameters was very weak, the diffusive transport events presented a statistically significant difference between the wet and dry seasons, however, the ebullitive fluxes did not present statistically significant differences between the wet and dry seasons, while the environmental parameters showed statistically significant differences between the two seasons. The severe drought that the region was subjected to during two years of the study may have influenced the flooding and the emission patterns. The methane emission estimates using different approximations, since a simplified general average flux, up to considering the diffusive, ebullitive and vegetation influences, resulted in methane release to atmosphere in the range of 1.4–2.2 Tg CH_4 y^{-1}, corresponding to about 2.2% of the global emission from natural wetlands. Thus, Pantanal may have the same potential of emission as Amazônia. Finally, additional areas of the Pantanal should be investigated to increase the confidence in the emission pattern obtained in this study.

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Chapter 8
Nitrous Oxide Emission from Crop Fields and Its Role in Atmospheric Radiative Forcing

Deepanjan Majumdar

8.1 Introduction

Nitrous oxide, popularly known as a laughing gas has emerged as an important gas for environmental sustainability. In the troposphere, N_2O is a chemically inert gas, but acts as a potential greenhouse gas. The greenhouse effect of N_2O was first reported by Yung et al. (1976). Global warming potential of N_2O, over a time horizon of 100 years, is measured 296 times that of CO_2 (Ramaswamy et al. 2001). Therefore, an increasing trend of atmospheric N_2O is a serious concern, although N_2O emission is very low as compared to CO_2, the most abundant greenhouse gas in the atmosphere (IPCC 1996a). Nitrous oxide emission from 1750 to 2000 has caused an atmospheric radiative forcing of $0.15\,\mathrm{Wm^{-2}}$ or 6% of the enhanced radiative forcing by well-mixed greenhouse gases during this time, equivalent to $2.43\,\mathrm{W\,m^{-2}}$. While other greenhouse gases, like CO_2, CH_4 and halocarbons, have contributed 1.46, 0.48 and $0.34\,\mathrm{Wm^{-2}}$, respectively. Thus, global average surface temperature (the average of near surface air temperature over land and sea surface temperature) has increased by $0.6 \pm 0.2°\mathrm{C}$ over 20th century (IPCC 2001).

Before the effect of N_2O on radiative balance of atmosphere could be recognized, Crutzen (1970) had pointed out its vital role in the destruction of stratospheric ozone (O_3), which protects the biosphere from harmful UV-B radiation ($280\,\mathrm{nm} < \lambda < 320\,\mathrm{nm}$) (Baird 1995). Photooxidation of N_2O is the major source of stratospheric NO_x ("active nitrogen"), which is the main catalyst of gas-phase O_3 destruction at altitudes between 25 and 40 km above earth's surface (Brasseur et al. 1999). A doubling of the atmospheric N_2O might increase the UV-B entry to atmosphere by about 20% causing a 10% decrease in the O_3 layer (Crutzen and Ehhalt 1977). Shea (1988) estimated that there would be 4–6% increase in skin cancer with each 1% drop in O_3 layer. Because N_2O is photolysed more rapidly in the stratosphere by the enhanced UV flux, the increase in N_2O mixing ratio is diminished (Prather 1998).

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Measurements on entrapped N_2O in polar ice indicate that global concentration of N_2O is highest in recent times during the past 45,000 years (Leuenberger and Siegenthaler 1992). After the last ice age, N_2O concentration increased and then remained constant at $\sim$275 ppbv for about 10,000 years until the 19th century. Then after, globally averaged surface-mixing ratio increased from about 270 nmol mol^{-1} (ppbV) in pre-industrial times (1750) to about 314 nmol mol^{-1} in 1998 and continued to grow at a current rate of 0.8 ± 0.2 nmol mol^{-1} yr^{-1} or $+0.25 \pm 0.05\%$ yr^{-1} (Prather et al. 2001).

Emission of N_2O from soil is termed as a loss of added and residual soil N, which is indeed an economic loss. However, this loss is marginal in terms of nitrogen. The deleterious effects of N_2O in atmosphere assume a greater significance due to its high atmospheric lifetime which is estimated between 110 and 168 years (Cicerone 1989; Prinn et al. 1990). Due to its high residence time in the atmosphere, atmospheric burden of N_2O will persist for many years, even if its emission is completely stopped now. Even a low rate of N_2O increase results in its atmospheric accumulation and any concentration perturbation will last for centuries. So, role of N_2O, both as a greenhouse and O_3 depleting gas combined with its long atmospheric lifetime, calls for its mitigation. It is one of the targetted greenhouse gases under 1997 Kyoto Protocol, which warrants emission limitations and reduction commitments for greenhouse gases, not controlled by the Montreal Protocol.

8.2 Global N_2O Budget

Among the atmospheric nitrogen species, N_2O abundance comes second to molecular nitrogen (N_2). Nitrous oxide budget is still not well quantified today due to uncertainties in the estimation of individual sources. Several researchers have come out with budgetary estimates at different times using available literature and data (McElroy and Woofsy 1986; Davidson 1991). Frequent updating of estimates is necessary since individual source strengths vary widely with time due to wide heterogeneity of the systems emitting N_2O and the complex interactions of physical, chemical and biological variables. Table 8.1 shows a compilation of recent estimates on global N_2O emissions from different sources (IPCC 1996b).

8.3 Guidelines for Greenhouse Gas Inventorization from Crop Fields *vis-a-vis* Agriculture

IPCC has provided a detailed methodology to estimate national greenhouse gas inventories from agriculture, which has been published in documents, like "Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories" (http://www.ipcc-nggip.iges.or.jp/public/gl/invs1.htm) and "Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories" (http://www.ipcc-nggip.iges.or.jp/public/gp/english/). The methodology attempts to relate N_2O

Table 8.1 Estimates of global N_2O budget (Tg N year^{-1}) (IPCC 1996b)

References	Mosier et al. (1998a) Kroeze et al. (1999)		Olivier et al. (1998)		*SAR		#TAR
Base year	1994	Range	1990	Range	1980s	Range	1990s
Sources							
Ocean	3.0	1–5	3.6	2.8–5.7	3	1–5	–
Atmosphere (NH_3 oxidation)	0.6	0.3–1.2	0.6	0.3–1.2			
Tropical soils	–	–	–	–	–	–	–
Wet forest	3.0	2.2–3.7	–	–	–	3	2.2–3.7
Dry savannahs	1.0	0.5–2.2	–	–	–	1	0.5–2.0
Temperate soils	–	–	–	–	–	–	
Forests	1.0	0.1–2.0	–	–	–	1 0	0.5–2.3
Grasslands	1.0	0.5-2.0	–	–	–	1	–
All soils	–	–	6.6	3.3-9.9	–	–	–
Natural sub-total	9.6	4.6-15.9	10.8	6.4–16.8	9	–	–
Agricultural soils	4.2	0.6–14.8	1.9	0.7–4.3	3.5	1.8–5.3	–
Biomass burning	0.5	0.2–1.0	0.5	0.2–0.8	0.5	0.2–1.0	–
Industrial sources	1.3	0.7–1.8	0.7	0.2–1.1	1.3	0.7–1.8	–
Cattle and feedlots	2.1	0.6–3.1	1.0	0.2–2.0	0.4	0.2–0.5	–
Anthropogenic sub-total	8.1	2.1–20.7	4.1	1.3–7.7	5.7	–	+6.7
Total sources	17.7	6.7–36.6	14.9	7.7–24.5	14.7	10–17	–
Imbalance (trend)	3.9	3.1–4.7	–	–	3.9	3.1–4.7	3.8
Total sinks (stratospheric)	12.3	9–16			12.3	9–16	12.6
Implied total source	16.2	–	–	–	16.2	–	16.4

*Second IPCC Assessment Report (Houghton et al. 1994).

#Third IPCC Assessment Report (Prather et al. 2001).

+corrected according to IPCC Special Report on Emission Scenarios (Nakicenovic et al. 2000).

emissions to agricultural N cycle (direct N_2O emissions) and systems to which N is transported once it leaves agricultural systems (indirect N_2O emissions).

Through a Tier 1a equation, direct N_2O-N emission from agricultural soils could be calculated by entering values of inputs, like amounts of synthetic fertilizer-N, animal manure-N, biologically fixed-N, crop residue-N and area of cultivated organic soils, all on annual basis. Each input has to be calculated exclusively via separate equations to reach a reliable input value, since no direct estimate of these inputs are available. A Tier 1 b equation, which is available for direct N_2O emission calculation as well as some input calculations, is an expanded form of Tier 1a, with more detailed options on input parameters and could be used, if the concerned country had these explicit data. These country-wise inputs are multiplied with respective country-wise emissions factors (kg N_2O-N/kg N input for first four and kg N_2O-N/ha-yr for the last input mentioned earlier in this section) for calculating national N_2O inventory. The method requires reliable estimates of all the inputs from frequent experimentation carried out for sufficiently long time to incorporate the variability of the systems. In case of unavailability of country-specific emission factor, the value could be taken from any other country's with comparable management practices and climate. Otherwise, IPCC default emissions factors could be used which are updated on the basis of reliable new evidence or data. A mix of country specific and default emission factors could be suitably used for finalizing national N_2O inventory calculation. For these estimations, IPCC has also streamlined "good practices" in derivation of country specific emission factors, which say that N_2O emission measurements should be carried out over wide ranging crop growing regions, seasons, soils and management practices. Further, measurements should be carried out over an entire year, preferably over series of years and once per day after major disturbances, which could increase N_2O emissions beyond background levels (e.g. rainfall, fertilizer N application, tillage etc.). Also provided is a decision tree for the choice of IPCC methodology for national N_2O inventory calculation.

For calculating indirect N_2O emissions, five separate pathways have been identified whereby agricultural N input could be available for N_2O emissions elsewhere. These are depositions of NO_x and NH_4^+ from atmosphere originated from volatilization of applied N to soils, leaching and run-off-N deposited on soil elsewhere, disposal of sewage-N, N_2O formation from NH_3 in atmosphere and disposal of food processing and miscellaneous effluents. The principal equation for estimating indirect N_2O emissions from nitrogen used in the agriculture incorporates first three inputs only, as the last two are still not understood or quantified with certainty. Although the principal equation does not have an expanded form, each of the first three inputs has a truncated and an expanded form of equation, requiring very explicit inputs. Default emission factors for the first three inputs, good practices and decision tree for methodology selection have been provided. To ensure correctness of inventories, IPCC suggests review of emission factors, checking of activity data (i.e. fertilizer consumption, livestock N excretion etc. used

Table 8.2 N_2O-N emissions ($Tg\ y^{-1}$) from Indian agricultural soils (1961–2000)

N_2O emissions	1961 ($Tg\ yr^{-1}$)	2000 ($Tg\ yr^{-1}$)
A. Direct soil emissions		
Synthetic fertilizer use	0.00281	0.122
Animal waste	0.000143	0.000232
Biological N_2 fixation	0.008201	0.011795
Crop residue	0.03509	0.0947
Cultivated histosols	0.000259	0.000259
Sub-Total	0.031	0.183
B. Animal production		
Animal waste management systems	0.0000027	0.0000046
C. Indirect emissions		
Atmospheric deposition	0.00028	0.011
Nitrogen leaching and run off	0.00133	0.054
Human sewage	–	–
Sub-Total	0.0016	0.065
Total emissions	0.048	0.294

for indirect emissions) and external review of all parameters, emission factors and methodologies by specialists.

Indeed, IPCC method has been used to prepare regional estimates on N_2O emission. Yang et al. (2003) have estimated N_2O emissions from rice fields and uplands in Taiwan during 1990–2000. Few estimates have been done for Indian agriculture as well. Using FAO database and IPCC 1996 guidelines, Krishna Prasad et al. (2003) have estimated N_2O emissions from Indian agriculture during 1961–2000 (Table 8.2). According to another estimate, N_2O emission from Indian agriculture, for the base year 1994–1995 was calculated to be 0.08 Tg (39 Tg CO_2 equivalent) (Bhatia et al. 2004). Several other regional estimates for N_2O emissions from agriculture or particular crops calculated by IPCC methodology are now available (Sokona 1995; Van Moortel et al. 2000; Boeckx et al. 2001; Li et al. 2001). Comparison of IPCC methodology derived N_2O emissions with other UK-derived inventories suggested that the IPCC methodology had overestimated N_2O emissions (Brown et al. 2001).

8.4 Contribution of Agriculture to Atmospheric N_2O Loading

Nitrous oxide budget is generally estimated by using available data on N inputs, area under cultivation and N_2O emission factors for various agricultural activities in varied cropping systems under different soil and climate. Initially, several researchers have estimated fertilizer derived annual N_2O emissions by using annual fertilizer consumption viz. 6–20 Tg N_2O-N yr^{-1} (Hahn and Junge 1977); 0.5 (0.2–2.1) Tg yr^{-1} (Eichner 1990); 0.6–2.3 Tg yr^{-1} (Bolle et al. 1986); 0.2–0.6 Tg yr^{-1} (McElroy and Woofsy 1985) etc. A continent-wise break up of direct and indirect emissions of N_2O from agricultural fields is also available (IPCC 1996b) (Table 8.3). Various other estimates differ widely on the percentage contribution of agriculture in anthropogenic N_2O emissions viz. 96% (Duxbury et al. 1993); 81% (Iserman 1994); 76% (Cole et al. 1996) and 65% (Mosier et al. 1998a). It was once suggested that

Table 8.3 Estimates of direct and indirect emissions of N_2O from applications of fertilizer N (synthetic or animal waste) to agricultural soils and from soils growing N-fixing crops (Mt N_2O-N yr^{-1}) (IPCC 1996b)

	Estimated N_2O from				
Region	Mineral N	Animal waste	N-fixation	Total	Range
Africa	0.04	0.21	0.05	0.3	0.15–0.45
North and Central America	0.26	0.11	0.11	0.48	0.24–0.72
South America	0.03	0.22	0.09	0.34	0.17–0.51
Asia	0.75	0.52	0.19	1.46	0.73–2.19
Europe	0.27	0.22	0.02	0.51	0.26–0.77
Oceania	0.01	0.03	0.01	0.05	0.03–0.08
FSU	0.17	0.18	0.03	0.30	0.19–0.57
Total	1.53	1.49	0.50	3.50	1.8–5.3

agriculture's contribution to global N_2O loading in coming two decades (1986–2006) would increase by 90% mainly due to N fertilizer application (Iserman 1994).

8.5 Factors Affecting N_2O Emission from Crop Fields

A number of factors that affect N_2O emission from soils are inherent soil factors, like N content, organic matter, moisture, pH, aeration, temperature (Freney et al. 1979; Sahrawat and Keeney 1986; Pathak 1999), soil compaction (Ball et al. 1999), nitrifying or denitrifying bacteria (Webster and Hopkins 1996; Yue et al. 2005), earthworm activity (Bertora et al. 2007), texture (Weitz et al. 2001), metals (Holtan-Hartwig et al. 2002) and crop management factors like presence of floodwater (Majumdar et al. 2001; DeDatta 1995), application of N and N-inhibitors (Majumdar et al. 2000, 2002), presence of crops (Xu et al. 2002), tillage operations (Liu et al. 2007), soil fumigation (Spokas and Wang 2003), pesticide application (Singh et al. 1999), mulching (Flessa et al. 2002), liming (Mosier et al. 1998b) and natural forcing like freezing and thawing (Morkved et al. 2006). Bouwman (1996) had observed that it was impossible to determine precisely the relative contribution of crops, amount and type of N fertilizer, management practices and weather conditions to N_2O emissions. Eichner (1990) suggested that N_2O emission is primarily a function of N source, while Mosier et al. (1996) emphasized that cropping systems and soil management practices had a greater impact than N source. It is understood that a complex interplay of all these driving factors will actually determine N_2O emissions from crop fields. While factors, like soil moisture, aeration, N content, application of fertilizer and nitrification inhibitors, organic matter content, pH, temperature and microbes, influence the production of N_2O gas, the emission is

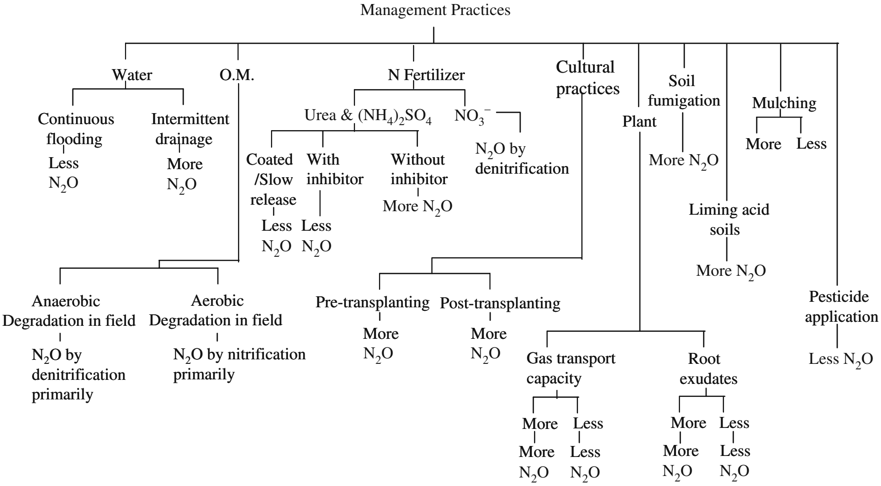

Fig. 8.1 Impact of management practices on N_2O emissions from crop fields

primarily controlled by water regime, plant population, soil texture and interculture operations. Figure 8.1 summarizes the influence of various crop management practices on N_2O emission.

8.6 Special Conditions in Rice Field

Although all the factors mentioned earlier influence N_2O emission from rice fields also, but exclusive conditions in a rice field need a special mention. Rice is cultivated under a wide variety of climate, soil and water regime. A majority of rice cultivation practices require standing water over the field for extended periods during their life cycle, which makes the saturated plow layer develop a reducing layer below an oxidizing layer at the soil surface. Below the water-saturated plow layer, there is an underground saturated soil layer in rice-growing season. In the plains, the underground saturated layer is usually located at 0.7–1.5 m below the surface layer and the its depth varies with a number of factors, such as topography, season, precipitation and irrigation. Flooding of rice fields during land preparation (as in case of wetland rice) or later (as done in many regions e.g. US, Australia, parts of Europe and in some Asian countries) initiates several physico-chemical and microbiological processes in soil, which are starkly different from other cultivated soils (DeDatta 1995). Flooded rice soils are characterized by limitation of O_2, since the diffusivity of O_2 through water is 10,000 times smaller than that in gas and thus anoxic conditions are easily created. Only in the first few millimeters of surface soil, partially aerobic conditions are found which is primarily due to the dissolved oxygen present in water (Ponnamperuma 1972).

N need for rice crop is primarily met through chemical fertilizers and urea has approximately 90% share in it (Vlek and Byrnes 1986). While ammonium sulphate is also used in many regions, but due to low use efficiency, NO_3^- fertilizers are seldom used. Thus, only 40% of added N is used by the rice crop (DeDatta et al. 1968). After application, NH_4^+-N in fertilizers gets accumulated by fixation in soil and is lost as NH_3 volatilization, leaching, run-off and nitrified in the oxidizing layer, at the water-and-soil interface, forming NO_3^- which moves downwards to the reducing layer and there it becomes denitrified subsequently (DeDatta 1995). Most research on denitrification in paddy fields focused on the submerged upper plow layer and little has been reported on denitrification in the underground saturated soil layer, but it has been reported that in regions with high N input in rice, denitrification in the underground saturated soil mitigates NO_3^- pollution in groundwater, but contributes N_2O to atmosphere (Xing et al. 2002a). In the thin oxidized zone present at surface of soil, nitrifiers can transform NH_4^+ to NO_3^- via NO_2^-, which gets transported to lower anaerobic layers where it is transformed to N_2O and N_2 by denitrifiers (Zhu et al. 2003). Further down, where rice roots occupy much of the soil volume, a significant amount of O_2 might be present through transportation by aerenchyma (Savant and DeDatta 1982). So, in the predominantly oxic zone, nitrification can also take place to produce N_2O directly or to produce NO_3^-, which gets finally denitrified to N_2O. Thus a coupling of nitrification and denitrification reactions in rice is evidenced indirectly by balancing N supply with N recovery from plant and soil (Fillery et al. 1984, 1986; Reddy and Patrick 1986). Rice plants will also affect nitrification and denitrification indirectly by immobilizing NH_4^+ and NO_3^- from the rhizosphere and also by supplying root exudates and dead root debris, which act as substrate for microorganisms. Savant and DeDatta (1982) have suggested mechanisms by which nitrification and denitrification occur near rice root. In upland rice cultivation, rice fields do not remain permanently flooded (except where permanent flooding is deliberately done) as the applied water drains from soil depending on soil type and extent of puddling done before transplanting. During drying cycles, N_2O emission increases (Chen et al. 1997; Majumdar et al. 2000) as nitrification gets a boost due to diffusion of O_2 into soil in absence of standing water and also because N_2 formation gets reduced under less intense anaerobic condition (Granli and Bockman 1994). During drying cycles, a significant amount of NO_3^- gets accumulated in soil due to nitrification of previously accumulated NH_4^+-N. This NO_3^- is lost by denitrification once the soil is flooded again and at the end of rice season, only limited amount of mineral N (NH_4^+ and NO_3^-) will remain due to different losses and plant uptake (DeDatta 1995). Permanently flooded rice fields are not considered a potent source of atmospheric N_2O, because N_2O is further reduced to N_2 under the strict anaerobic conditions (Granli and Bockman 1994).

Topography of a rice field can influence soil water, thermal regime and soil organic matter content and hence might control N_2O emission. Xu et al. (1997) have reported that the soil of low topography at Yingtan in China had higher total N and organic matter contents than those of higher topography and rice paddy field of low topography emitted more N_2O than those of high topography, with the exception of N_2O flux from middle plot of early rice. The exception may be due to the difference of water management between middle plot and the other two plots of early rice.

8.7 Production Versus Emissions of N_2O from Crop Fields

Nitrous oxide emission does not match with the real time N_2O production in soil as the gas is attenuated by further denitrification to N_2 (Granli and Bockman 1994), entrapment in soil pores, dissolution in soil water and floodwater (Linke 1965), and leaching in case of rice fields (Nevison 2000). Since our immediate concern is N_2O loading in the atmosphere, we limit N_2O monitoring to soil surface emissions only, rather than soil profile measurements. Real time N_2O production in soil is difficult to be estimated with precision, since soil is not a closed system per se and N_2O gets diffused in many directions quickly, depending on soil porosity, compaction, water content etc. Depending on several simultaneous spatial measurements at a particular depth, N_2O diffusion coefficient in soil could be determined (Li and Kelliher 2005). Soil profile N_2O concentrations have been estimated by various researchers, which possibly gave us an idea only on the relative preponderance of gas production zones at the sampling point only and temporal and spatial trends of production, but not the exact amount of real time production.

N_2O production potential of soils could be more realistically determined in undisturbed soil cores under controlled conditions in laboratory where the entire N_2O emitted could be sampled from a definite quantity of soil. Though N_2O diffusion and leaching could be minimized in a small incubation chamber, real time entrapped or dissolved N_2O cannot be sampled without disturbing the soil core. Entrapped N_2O could be sampled by soil shaking and sampling incubation chamber headspace, but only with loose soil samples, which are not ideal representative of the field condition.

Leaching of N_2O in groundwater could also pose a problem in the quantification of real time production in soil. The importance of N_2O leaching in crop fields has been identified (IPCC 2001), but a few field experiments have been done to quantify this loss. A very recent study on dissolved N_2O in paddy agro-ecosystems in China has indicated that the dissolved N_2O/NO_3^- ratios in leachate and groundwater in paddy ecosystem could be significant, but is far lower than the current IPCC default value (Xiong et al. 2006).

Appreciable amount of N_2O may remain dissolved in floodwater in rice fields after emission from soil and thus escapes sampling for surface emissions. Dissolved N_2O needs to be estimated for a better quantification of N_2O emissions from soil. An equation had been proposed by Moraghan and Buresh (1977) to calculate dissolved N_2O for laboratory incubation systems only, which is based on Henry's Law:

$Y = \alpha.\ x.$ (solution volume/atmosphere volume)
Where Y = amount of dissolved N_2O (mg) in a closed system
α = solubility of N_2O (cm^3 N_2O dissolved per cm^3 of water)
x = amount of N_2O (mg) in the flask atmosphere (i.e. flask headspace).

Solubility of N_2O in water at different temperatures is known (Linke 1965) and can be used in this formula. The solubility of $0.7\,g\,N_2O\text{-}N\,L^{-1}$ at 25°C (Wilhelm et al. 1977) can be used for estimating N_2O dissolution at NTP. It might be interesting to study the applicability of this equation for estimating floodwater N_2O in rice by tallying measured and calculated dissolved N_2O in a flooded rice by static chamber method. Minami and Fukushi (1984) have described a method to effectively measure N_2O dissolved in floodwater and released from floodwater surface. They have found dissolved N_2O concentration in the range of $0\text{–}0.38\,\mu g\,L^{-1}$ in the floodwater. But, floodwater N_2O has rarely been estimated in N_2O monitoring activities in spite of availability of this technique.

Appreciable amount of N_2O might remain entrapped in the soil pore spaces (Lindau and DeLaune 1991), but its extent will be governed by soil texture, pore geometry, soil water content and pressure of standing water in case of rice. During field drying in a rice field, N_2O flux might increase due to withdrawn water pressure, which, otherwise, would push back some N_2O inside soil. More the time of N_2O entrapment, more is the chance for its conversion to N_2 by denitrification. One would desire this to happen, as this would reduce atmospheric load of N_2O from the rice fields.

8.8 N_2O Consumption by Crop Fields

Although much research has been done on the source strength of crop fields with regard to N_2O, a little focus has been given on its possible role as a sink of atmospheric N_2O. The N_2O uptake capacity of flooded rice soils comes to light, when published results on N_2O emission from rice fields are carefully examined (Majumdar 2005). Available literature on N_2O emissions from other agricultural crops indicates that negative fluxes from aerobic soils are not very uncommon and an excellent review on this subject is available (Chapuis-Lardy et al. 2007).

8.9 Simulating N_2O Emissions from Crop Fields

Simulation of greenhouse gas emissions from agro-ecosystems was taken up after a few decades of understanding of the subject. It has assumed a great significance since emissions from agro-ecosystems are highly variable, both temporally and spatially and it is tedious to monitor all types of agro-ecosystems to come to a finer adjustment of regional and global estimates of greenhouse gas emissions. Among few models available for simulating N_2O emissions, two important models are discussed briefly here. The DAYCENT ecosystem model (a daily version of Century model) has been applied to simulate soil organic carbon levels, crop yields, and annual trace gas fluxes including N_2O for various soils (Del Grosso et al. 2001). This model utilizes a N gas sub-model to simulate N_2O and NO_x emissions from nitrification and denitrification as well as N_2 emissions from denitrification. Significantly, results from DAYCENT simulation indicated that conversion to no tillage at the national scale could mitigate $\sim$20% of US agricultural emissions or $\sim$1.5% of total US emissions of greenhouse gases (Del Grosso et al. 2005).

The DeNitrification-DeComposition (DNDC) model is a process-based model that focuses on N_2O, CO_2, NO, CH_4 and NH_3 emissions and has been applied to estimate N_2O emissions from agricultural fields (Li et al. 1992; 1994). Comparison of DNDC with the IPCC methodology for developing a national inventory of N_2O emissions has been done for arable lands in China and it was found that although total N_2O emissions were similar in both estimates, but geographical patterns of emissions were quite different (Li et al. 2001). Only recently, DNDC model has been modified to enhance its capacity of predicting greenhouse gas (GHG) emissions from rice ecosystems (Li et al. 2004). According to the simulation fir rice, shifting water management from continuous flooding to mid-season drainage increased N_2O fluxes by 0.13–0.20 Tg N_2O-N yr^{-1}.

8.10 A Summary of N_2O Emissions from Crop Fields

Although the role of soils in N_2O emissions is long known (Arnold 1954), monitoring of N_2O from crop fields has been started effectively from late 1970's to early 1980's (Delariche et al. 1978; Denmead et al. 1979; Freney et al. 1981; Smith

Table 8.4 A glance at N_2O emissions from rice fields worldwide under different management conditions

Site (Country)	Practice	Time span	Soil type	Fertilizer-N* (kg N ha^{-1})	N_2O Flux	Total N_2O emission	N lost (%)	Reference
Denilinquin (Australia)	Irr., CF, DS	18 d	Red clay	No N	* ~ 0–370 ng N m^{-2} s^{-1}	0.038 g N m^{-2}	–	Denmead et al. (1979)
Crowley (USA)	Irr., CF, DS	105 d	Typic Albaqualf, Silt loam	No N	<0.1–2.5 g N ha^{-1} d^{-1}	74 g N ha^{-1}	–	Smith et al. (1982)
				Drilled urea (90)	* ~0.2–3.5 g N ha^{-1} d^{-1}	108 g N ha^{-1}	0.04	
				Drilled urea (180)	5.4 g N ha^{-1} d^{-1} (Max)	171 g N ha^{-1}	0.05	
				Broadcasted urea (90)	* ~0–3.3 g N ha^{-1} d^{-1} (Max)	109 g N ha^{-1}	0.04	
				Broadcasted urea (180)	* ~0–1.7 g N ha^{-1} d^{-1} (Max)	90 g N ha^{-1}	0.01	
Ibaraki (Japan)	Irr., CF, Tr.	82 d	Volcanic ash soil	Urea (12.5)	−0.22–1.48 g N 10 a^{-1} d^{-1}	NF	NF	Minami and Fukushi (1984)
Crowley (USA)	Irr., CF, DS	21 d	Typic Albaqualf, Silt loam	No N	0.4 g ha^{-1} d^{-1} (M)	NF	–	Lindau et al. (1990a)
				Urea (120)	<0.3 g ha^{-1} d^{-1} (M)	NF	–	
				Am. sul. (120)	<3 g ha^{-1} d^{-1} (Max)	NF	–	
				Pot. Nitrate (120)	<1–122 g ha^{-1} d^{-1}	NF	–	
Crowley (USA)	Irr., CF, DS	21 d	Typic Albaqualf, Silt loam	Urea (120)	<3 g N ha^{-1} d^{-1} (Max)	NF	–	Lindau et al. (1990b)
				Pot. Nitrate (120)	<3 g N ha^{-1} d^{-1} (Max)	NF	–	
Griffith (Australia)	Irr., CF, DS	40 d	Grey clay	No N	38 g N ha^{-1} d^{-1} (M)	NF	–	Mosier et al. (1994)
				No N+CCC (40 kg CaC_2 ha^{-1})	14 g N ha^{-1} d^{-1} (M)	NF	–	
				Urea (80)	73 g N ha^{-1} d^{-1} (M)	NF	–	
				Urea(80)+NP (2.4 L ai ha^{-1})	99 g N ha^{-1} d^{-1} (M)	NF	–	
				Urea(80)+CCC (40 kg CaC_2 ha^{-1})	16 g N ha^{-1} d^{-1} (M)	NF	–	
Shenyang (China)	Irr., AFD, Tr.	240 d	NF	Urea (350)	NF	1.63 kg N ha^{-1}	0. – 46	Chen et al. (1995)

Table 8.4 (continued)

Site (Country)	Practice	Time span	Soil type	Fertilizer-N* (kg N ha^{-1})	N$_2$O Flux	Total N$_2$O emission	N lost (%)	Reference
Suzhou (China)	Irr., AFD, Tr.	117 d	NF	Am. sulphate (220)	NF	3.27 kg N ha^{-1}	0.48	Xu et al. (1995)
Yingtan (China)	Irr., AFD, Tr.,	59 d	NF	Urea (266)	NF	1.14 kg N ha^{-1}	0.42	
Yingtan (China)	Early rice Irr., AFD, Tr., Late rice	98 d	NF	Urea (266)	NF	0.025 kg N ha^{-1}	0.009	
Fengqiu (China)	Irr., AFD, Tr.	109 d	Sandy	Urea+Am.sul+ Manure(409)	182.2 μg m^{-2} h^{-1} (M)	472.26 mg m^{-2}	1.15	Xu et al. (1997)
			Loamy	Urea+Am.sul+ Manure(409)	82.85 μg m^{-2} h^{-1} (M)	214.75 mg m^{-2}	0.52	
			Clayey	Urea+Am.sul+ Manure(409)	68.69 μg m^{-2} h^{-1} (M)	178.04 mg m^{-2}	0.43	
Nanjing (China)	Irr., AFD, Tr.	112 d	Loam clay	Urea (100)	6.33 μg m^{-2} h^{-1} (M)	17.02 mg m^{-2}	0.031	
				Urea (300)	23 μg m^{-2} h^{-1} (M)	61.82 mg m^{-2}	0.16	
				Am. sulphate (100)	6.49 μg m^{-2} h^{-1} (M)	17.45 mg m^{-2}	0.28	
				Am. sulphate (300)	36.5 μg m^{-2} h^{-1} (M)	98.11 mg m^{-2}	0.38	
Yingtan (China)	Irr., AFD, Tr., Early rice	36 d	Loam clay	Urea (122)+Top slope	-0.63 μg m^{-2} h^{-1} (M)	-0.54 mg m^{-2}	–	
			Loam clay	Urea (122)+Middle slope	129.8 μg m^{-2} h^{-1} (M)	112.15 mg m^{-2}	0.91	
			Clay loam	Urea (122)+Bottom slope	1.15 μg m^{-2} h^{-1} (M)	0.99 mg m^{-2}	0.008	
Yingtan (China)	Irr., AFD, Tr., Late rice	36 d	Loam clay	Urea (122)+Top slope	0.8 μg m^{-2} h^{-1} (M)	1.73 mg m^{-2}	0.014	
			Loam clay	Urea (122)+Middle slope	1.04 μg m^{-2} h^{-1} (M)	2.25 mg m^{-2}	0.018	
			Clay loam	Urea (122)+Middle slope	1.69 μg m^{-2} h^{-1} (M)	3.65 mg m^{-2}	0.029	
Suzhou (China)	NF	117 d	NF	Urea (210)	NF	2.57 mg m^{-2}	0.22	Xing and Zhu (1997)
	NF		NF	Urea (310)	NF	2.82 mg m^{-2}	0.19	
Guangzhou (China)	NF	100 d	NF	Urea (141)	NF	0.27 mg m^{-2}	0.001	
Jiangsu (China)	Irr., CF, Tr.,	112 d	Stagnic luvisol, silty clay	No N	5.1 μg m^{-2} h^{-1} (M)	NF # 33 g ha^{-1}	–	Cai et al. (1997)
				Urea (100)	6.3 μg m^{-2} h^{-1} (M)	# 480 g ha^{-1}	0.033	
				Urea (300)	23.0 μg m^{-2} h^{-1} (M)	# 38 g ha^{-1}	0.16	
				Am. sul. (100)	6.5 μg m^{-2} h^{-1} (M)	# 840 g ha^{-1}	0.038	
				Am. sul. (300)	36.5 μg m^{-2} h^{-1} (M)		0.28	

Table 8.4 (continued)

Site (Country)	Practice	Time span	Soil type	Fertilizer-N* (kg N ha^{-1})	N$_2$O Flux	Total N$_2$O emission	N lost (%)	Reference
Los Banos (Philippines)	Irr., PI drainage	111 d	Aquandic Epiaqualf, Clay	Urea (200)	* ~0–0.5 mg m^{-2} d^{-1}	8.9 mg N m^{-2}	0.04	Bronson et al. (1997a)
				Am. Sul. (200)	* ~0–3.7 mg m^{-2} d^{-1}	22.9 mg N m^{-2}	0.11	
				Urea (140)+GM (60)	* ~0–1.5 mg m^{-2} d^{-1}	12.1 mg N m^{-2}	0.06	
	Irr., flooded		Aquandic	Urea (160)+ Straw (32)	* ~0–0.7 mg m^{-2} d^{-1}	5.7 mg N m^{-2}	0.03	
	Irr., MT drainage	97 d	Epiaqualf, Clay	Urea (200)	* ~0–2.5 mg m^{-2} d^{-1}	25.9 mg N m^{-2}	0.13	
				Urea (160)+ Straw (32)	* ~0–0.4 mg m^{-2} d^{-1}	2.8 mg N m^{-2}	0.01	
		97 d	Aquandic Epiaqualf, Clay	Urea (200)	* ~0–6 mg m^{-2} d^{-1}	63.8 mg N m^{-2}	0.53	
				Urea (160)+ Straw (32)	* ~0–0.9 mg m^{-2} d^{-1}	16.5 mg N m^{-2}	0.15	
Los Banos (Philippines)	Irr., CF, Tr., Wet season	98 d	Aquandic Epiaqualf, Guadaulpe clay	Urea (120)	* ~0–1.7 mg N m^{-2} d^{-1}	13.1 mg N m^{-2}	0.11	
				Am sul (120)	* ~0–0.9 mg N m^{-2} d^{-1}	9.7 mg N m^{-2}	0.08	
				Urea (60)+GM (60)	* ~0–0.4 mg N m^{-2} d^{-1}	5.5 mg N m^{-2}	0.05	
				Urea (80)+ Straw (32)	* ~0–0.7 mg N m^{-2} d^{-1}	5.9 mg N m^{-2}	0.05	
Los Banos (Philippines)	Fallow (wet-dry season)	46 d	Aquandic Epiaqualf, Guadaulpe clay	Urea residual	* ~0–7.5 mg N m^{-2} d^{-1}	61.0 mg N m^{-2}	–	Bronson et al. (1997b)
				Am. sul. residual	* ~0–17 mg N m^{-2} d^{-1}	94.3 mg N m^{-2}	–	
				urea+ GM residual	* ~0–22 mg N m^{-2} d^{-1}	78.0 mg N m^{-2}	–	
	Fallow (dry-wet season)	36 d		urea+ straw residual	* ~0–22 mg N m^{-2} d^{-1}	103.8 mg N m^{-2}	–	
				Urea residual	* ~0–70 mg N m^{-2} d^{-1}	171.3 mg N m^{-2}	–	
				Am. sul. residual	* ~0–50 mg N m^{-2} d^{-1}	119.8 mg N m^{-2}	–	
	Fallow (wet-dry season)	89 d		Urea+ GM residual	* ~0–80 mg N m^{-2} d^{-1}	182.7 mg N m^{-2}	–	
				Urea+ straw residual	* ~0–65 mg N m^{-2} d^{-1}	171.6 mg N m^{-2}	–	
				Urea residual	* ~0–1.4 mg N m^{-2} d^{-1}	19.3 mg N m^{-2}	–	
				Am. sul. residual	* ~0–1.0 mg N m^{-2} d^{-1}	16.0 mg N m^{-2}	–	
				Urea+ GM residual	* ~0–1.1 mg N m^{-2} d^{-1}	18.1 mg N m^{-2}	–	
				Urea+ straw residual	* ~0–3.3 mg N m^{-2} d^{-1}	36.1 mg N m^{-2}	–	
Wuxian (China)	Irr., IFD, Tr.	240 d	NF	Urea (160)	–11.2–751.1 μg N m^{-2} ha^{-1}	2.99 kg ha^{-1}	1.87	Zheng et al. (1997)
Shenyang (China)	Irr., CF, Tr.	300 d	Meadow brown	No N	* ~–100–125 μg m^{-2} ha^{-1}	NF	–	Chen et al. (1997)
				Urea (374)+manure (37.5 tha^{-1})	−39 − 164 μg m^{-2} ha^{-1}	NF	–	
				Urea (374)+manure (37.5 tha^{-1})+ Azolla(156 g m^{-2})	* ~–50–440 μg m^{-2} ha^{-1}	NF	–	

Table 8.4 (continued)

Site (Country)	Practice	Time span	Soil type	Fertilizer-N* (kg N ha^{-1})	N_2O Flux	Total N_2O emission	N lost (%)	Reference
Guangzhou (China)	Irr., CF, Tr.	97 d	NF	Urea (150)	−22–53.8 μg N m^{-2} ha^{-1}	0.18 kg ha^{-1}	0.12	Lu et al. (1997)
Guangzhou (China)	Irr., AFD, Tr.	97 d	NF	Urea (150)	0–74 μg N m^{-2} ha^{-1}	0.31 kg ha^{-1}	0.21	
NCHU Exp. Station (Taiwan)	Irr., AFD, Tr.	NF	NF	NF	NF	−0.11 − 0.6 kg ha^{-1}	–	Chao (1997)
				NF	NF	−0.11 − 0.5 kg ha^{-1}	–	
Louisiana (USA)	Irr., CF, Tr.	10 d	Typic albaqualf, silt loam	Urea (150)	10–29 g ha^{-1} d^{-1} above background level	NF	–	Lindau et al. (1998)
Bogor (Indonesia)	Irr., CF, Tr.	126 d	Latosol, Clay	No N	12.76 μg N m^{-2} h^{-1} (M)	352 g ha^{-1}	–	Suratno et al. (1998)
				No N	17.67 μg N m^{-2} h^{-1} (M)	594 g ha^{-1}	–	
				Granular urea (86)	22.99 μg N m^{-2} h^{-1} (M)	632 g ha^{-1}	0.73	
				Granular urea (86)	24.19 μg N m^{-2} h^{-1} (M)	813 g ha^{-1}	0.95	
				Tablet urea (86)	24.58 μg N m^{-2} h^{-1} (M)	678 g ha^{-1}	0.79	
				Tablet urea (86)	31.14 μg N m^{-2} h^{-1} (M)	1046 g ha^{-1}	1.22	
Bogor (Indonesia)	Irr., AFD, Tr.	126 d	Latosol, Clay	No N	16.72 μg N m^{-2} h^{-1} (M)	461 g ha^{-1}	–	Suratno et al. (1998)
				No N	18.70 μg N m^{-2} h^{-1} (M)	628 g ha^{-1} 740 g ha^{-1}	–	
				Granular urea (86)	26.84 μg N m^{-2} h^{-1} (M)	1096 g ha^{-1}	0.86	
				Granular urea (86)	32.63 μg N m^{-2} h^{-1} (M)	903 g ha^{-1}	1.27	
				Tablet urea (86)	32.75 μg N m^{-2} h^{-1} (M)	1142 g ha^{-1}	1.05	
				Tablet urea (86)	33.98 μg N m^{-2} h^{-1} (M)		1.33	
NTU exp. Station (Taiwan)	Irr., AFD, Tr.	NF	NF	NF	NF	0.30–0.39 g m^{-2}	–	Lai (1998, 2000)
	Irr., AFD, Tr.			NF	NF	0.07–0.75 g m^{-2}	–	
	Irr., AFD, Tr.	NF	NF	NF	NF	0.44–0.54 g m^{-2}	–	
Taoyuan (Taiwan)				NF	NF	0.06–0.40 g m^{-2}	–	
		NF	NF	NF	NF	0.35–0.78 g m^{-2}	–	
Hwalien-Lotung (Taiwan)				NF	NF	0.0–0.06 g m^{-2}	–	
Fengqiu (China)	Irr., CF, Tr.	108 d	Sandy Clayey Loamy	Urea+Am. bicarbonate (292)	168.8 μg N m^{-2} h^{-1} (M)	NF	–	Cai et al. (1999)
				Urea+Am. bicarbonate (292)	65.2 μg N m^{-2} h^{-1} (M)	NF	–	
				Urea+Am. Bicarbonate (292)	76.9 μg N m^{-2} h^{-1} (M)	NF	–	

Table 8.4 (continued)

Site (Country)	Practice	Time span	Soil type	Fertilizer-N* (kg N ha^{-1})	N_2O Flux	Total N_2O emission	N lost (%)	Reference
Los Banos (Philippines)	Irr., CF , Tr., Wet rice (1994)	120 d	Maahas Clay	Urea (90)	1.7 mg N m^{-2} d^{-1} (Max.)	0.097 kg ha^{-1}	0.11	Abao et al. (2000)
				Polyon-slow release N (90)		0.3 kg ha^{-1}	0.003	
	Irr., CF, Tr., Wet rice (1995)	120 d	Maahas Clay	Urea (90) in fallow plot	2.5 mg N m^{-2} d^{-1} (Max.)	24.9 kg ha^{-1}	0.27	
				Urea (90) +No Cow Pea residue		24.5 kg ha^{-1}	0.27	
				Urea (30)+Dry Cow Pea (3 t ha^{-1})		23.2 kg ha^{-1}	NF	
				Urea (90)+Wheat straw (3 t ha^{-1})		11.5 kg ha^{-1}	NF	
New Delhi (India)	Irr.,AFD, Tr.	70 d	Typic Ustochrept, Sandy loam	No N	61 μg N m^{-2} d^{-1} (M)	0.343 kg ha^{-1}	–	Majumdar et al. (2000)
				Urea (140)	110 μg N m^{-2} d^{-1} (M)	0.59.9 kg ha^{-1}	0.04	
				Urea+DCD (10% of N)	90 μg N m^{-2} d^{-1} (M)	0.49.0 kg ha^{-1}	0.03	
				Nimin coated urea (140)	129 μg N m^{-2} d^{-1} (M)	0.57.0 kg ha^{-1}	0.04	
				Neem coated urea (140)	114 μg N m^{-2} d^{-1} (M)	0.53.2 kg ha^{-1}	0.03	
New Delhi (India)	Irr., AFD, Tr.	98 d	Typic Ustochrept, Sandy loam	No N	664 mg N ha^{-1} d^{-1} (M)	0.46.0 N ha^{-1}	–	Kumar et al. (2000)
				Urea (140)	1875 mg N ha^{-1} d^{-1} (M)	0.160 N ha^{-1}	0.11	
				Urea+DCD (10% of N)	1407 mg N ha^{-1} d^{-1} (M)	0.142 N ha^{-1}	0.10	
				Urea+ thiosulphate (140)	1680 mg N ha^{-1} d^{-1} (M)	0147 N ha^{-1}	0.10	
				Am. Sulphate (140)	2850 mg N ha^{-1} d^{-1} (M)	0.235 N ha^{-1}	0.16	
				Am. sul. +DCD (140)	2029 mg N ha^{-1} d^{-1} (M)	0.147 N ha^{-1}	0.10	
Shenyang (China)	Irr., CF, Tr.	270 d	Meadow brown	Urea (172)	$-39.3 - 164.2$ μg m^{-2} ha^{-1}	1.08 kg ha^{-1}	0.63	Chen et al. (2000)
Shenyang (China)	Irr., CF, Tr.	250 d	Meadow brown	Urea (375)+ manure	* ∼–19–70 μg m^{-2} h^{-1}	NF	NF	Hou et al. (2000)
					* ∼–25–40 μg m^{-2} h^{-1}	NF	NF	
New Delhi (India)	Irr., AFD and saturated (s), Tr.	90 d	Typic Ustochrept, Sandy loam	No N (s)	* ∼1–12 g N ha^{-1} d^{-1}	0.323 kg ha^{-1}	–	Pathak et al. (2002)
				Urea (120) (s)	* ∼4–18 g N ha^{-1} d^{-1}	0.735 kg ha^{-1}	0.34	
				Urea+DCD (10%) (s)	* ∼1.5–14 g N ha^{-1} d^{-1}	0.483 kg ha^{-1}	0.13	
				Urea (60)+FYM (60) (s)	* ∼1.5–16 g N ha^{-1} d^{-1}	0.593 kg ha^{-1}	0.23	
				No N (AFD)	* ∼1–17 g N ha^{-1} d^{-1}	0.307 kg ha^{-1}	–	
				Urea (120) (AFD)	* ∼4.5–21 g N ha^{-1} d^{-1}	0.929 kg ha^{-1}	0.51	
				Urea+DCD (10%) (AFD)	* ∼3–17 g N ha^{-1} d^{-1}	0.540 kg ha^{-1}	0.19	
				Urea(60)+FYM (60) (AFD)	* ∼3–21 g N ha^{-1} d^{-1}	0.714 kg ha^{-1}	0.33	

Table 8.4 (continued)

Site (Country)	Practice	Time span	Soil type	Fertilizer-N* (kg N ha^{-1})	N_2O Flux	Total N_2O emission	N lost (%)	Reference
New Delhi (India)	Irr. AFD, Tr.	105 d	Typic Ustochrept, Sandy loam	No N	2.53 μg m^{-2} h^{-1} (M)	0.038 kg ha^{-1}	–	Ghosh et al. (2003)
				Urea (120)	10.1 μg m^{-2} h^{-1} (M)	0.168 kg ha^{-1}	0.13	
				Am. Sul. (120)	11.03 μg m^{-2} h^{-1} (M)	0.152 kg ha^{-1}	0.12	
				Pot. Nitrate (120)	20.19 μg m^{-2} h^{-1} (M)	0.187 kg ha^{-1}	0.15	
				Urea+DCD (120)	4.9 μg m^{-2} h^{-1} (M)	0.079 kg ha^{-1}	0.06	
				Am. Sul.+DCD (120)	6.26 μg m^{-2} h^{-1} (M)	0.082 kg ha^{-1}	0.06	
				Am. Nitrate +DCD (120)	19.4 μg m^{-2} h^{-1} (M)	0.168 kg ha^{-1}	0.13	
New Delhi (India)	Irr., AFD and saturated(s), Tr.	105 d	Typic Ustochrept, Sandy loam	No N (s)	$\sim$0–15 g N ha^{-1} d^{-1} (M)	0.353 kg ha^{-1}	–	Pathak et al. (2003)
				Urea (120) (s)	$\sim<$1–17 g N ha^{-1} d^{-1} (M)	0.775 kg ha^{-1}	0.35	
				Urea (120)+DCD (10%) (s)	$\sim<$1–16 g N ha^{-1} d^{-1} (M)	0.601 kg ha^{-1}	0.20	
				Urea (60)+FYM (60) (s)	$\sim$1–18 g N ha^{-1} d^{-1} (M)	0.742 kg ha^{-1}	0.32	
				No N (AFD)	$\sim$0–15 g N ha^{-1} d^{-1} (M)	0.402 kg ha^{-1}	–	
				Urea (120) (AFD)	$\sim$1–16 g N ha^{-1} d^{-1} (M)	0.820 kg ha^{-1}	0.35	
				Urea(120)+ DCD(10%)(AFD)	$\sim$1–19 g N ha^{-1} d^{-1} (M)	0.666 kg ha^{-1}	0.21	
				Urea(60)+FYM (60) (AFD)	$\sim$1–18 g N ha^{-1} d^{-1} (M)	0.791 kg ha^{-1}	0.32	
Nanjeng, China	Single, Early and late, Irr., MSA, Tr.	149 d	Clay loam	Urea (300) in single rice	35.3 μg N m^{-2} h^{-1} (M)	1.26 kg ha^{-1}	0.42	Xing et al. (2002b)
		150 d		Urea (300) in single rice	115 μg N m^{-2} h^{-1} (M)	4.14 kg ha^{-1}	1.38	
		77 d		Urea (200) in early rice	118 μg N m^{-2} h^{-1} (M)	4.15 kg ha^{-1}	1.09	
		92 d		Urea (300) in late rice	68.6 μg N m^{-2} h^{-1} (M)	1.51 kg ha^{-1}	0.50	
Yingtan (China)	Early rice, Tr.; flooded+ MSA+alternate drying, Late Rice (Rest same)	70 d	Hydragric Anthrosol, Red clay	No N	10.8 μg N m^{-2} h^{-1} (M)	0.18 kg ha^{-1}	–	Xiong et al. (2002a)
				Urea (276)	21.0 μg N m^{-2} h^{-1} (M)	0.35 kg ha^{-1}	0.062	
				Milk Vetch (124)	17.8 μg N m^{-2} h^{-1} (M)	0.30 kg ha^{-1}	0.094	
				Urea (276)+Milk Vetch (124)	167.0 μg N m^{-2} h^{-1} (M)	2.81 kg ha^{-1}	0.66	
							–	
							0.019	
		98 d		No N	9.9 μg N m^{-2} h^{-1} (M)	0.23 kg ha^{-1}	NF	
				Urea (276)	12.1 μg N m^{-2} h^{-1} (M)	0.28 kg ha^{-1}	0.04	
				Milk Vetch (124)	11.1 μg N m^{-2} h^{-1} (M)	0.26 kg ha^{-1}		
				Urea (276)+Milk Vetch (124)	14.5 μg N m^{-2} h^{-1} (M)	0.34 kg ha^{-1}		

Table 8.4 (continued)

Site (Country)	Practice	Time span	Soil type	Fertilizer-N* (kg N ha^{-1})	N_2O Flux	Total N_2O emission	N lost (%)	Reference
Jiangshi	Irr., DS, Upland	125 d	Udic Ferrosol,	No N	15.9 μg N m^{-2} h^{-1} (M)	0.48 kg ha^{-1}	–	Xiong et al.
(China)	and aerobic	125 d	Red clay	Urea (103.5)	204 μg N m^{-2} h^{-1} (M)	0.61 kg ha^{-1}	0.13	(2002b)
			Red clay				–	
	Fallow-Upland	291 d		No N	NF	0.86 kg ha^{-1}	0.13	
		291 d		Urea (103.5)	NF	0.99 kg ha^{-1}		
Yangcheng	Irr., Aerobic	100 d	Aqu-alkalic	Urea (225 kg ha^{-1})+ plastic	0.252 mg m^{-2} h^{-1} (M)	12.6 Kg ha^{-1}	5.6	Xu et al.
(China)	and flooded, Tr.		Halosol, Sandy	mulch	0.197 mg m^{-2} h^{-1} (M)	11.1 Kg ha^{-1}	4.9	(2004)
			loam	Urea (225 kg ha^{-1})+ straw	0.034 mg m^{-2} h^{-1} (M)	2.15 Kg ha^{-1}	1.0	
				mulch				
				Waterlogged soil				
Ludhiana	Irr	120 d	Typic ustochrept,	No N	~15–~300 g N ha^{-1} d^{-1}	–	–	Aulakh et al.
(India)			sandy loam	Urea (120)	~15–~ 440 g N ha^{-1} d^{-1}	–	–	(2001)
				Urea (32)+GM (88)	~15–~ 440 g N ha^{-1} d^{-1}	–	–	
				GM (176)	~15–~ 450 g N ha^{-1} d^{-1}	–	–	
New Delhi	Irr	120 d	Typic ustochrept,	Urea (120)	~1–17 g N ha^{-1} d^{-1}	0.76 Kg ha^{-1}	0.63	Malla et al.
(India)			sandy laom	Urea+ hydrquinone (120)	~1–19 g N ha^{-1} d^{-1}	0.73 Kg a^{-1}	0.60	(2005)
				Urea+neem cake (108 + 12)	~0.5–16 g N ha^{-1} d^{-1}	0.68 Kg ha^{-1}	0.56	
				Urea+ thiosulphate (120)	~1–17.5 g N ha^{-1} d^{-1}	0.50 Kg ha^{-1}	0.41	
				Urea+CCC (120)	~2–17 g N ha^{-1} d^{-1}	0.54 Kg ha^{-1}	0.45	
				Neem oil coated urea (120)	~0.5–13 g N ha^{-1} d^{-1}	0.60 Kg ha^{-1}	0.5	
				Urea+DCD (108+12)	~1.5–17 g N ha^{-1} d^{-1}	0.63 Kg ha^{-1}	0.53	

NF – Not found; CAF – Controlled availability fertilizers; CCC – Wax coated calcium carbide; MT – Mid tillering; PI – Panicle initiation; Tr. – Transplanted; MSA – Mid season aeration; Av. – Average; DS – Direct seeded; M – mean; Max. – Maximum; irr. – irrigated; s – saturated; AFD – Alternate flooding and drying; CF – Continuous flooding; Tr. – Transplanted; GM – green manure; * Approximate values, deduced from emission trend presented in graphs; # Calculated from N dose and percent N_2O-N loss.

Table 8.5 Expressions for calculating radiative forcing (IPCC 2001)

Trace Gas	Simplified expression radiative forcing, ΔF (Wm^{-2})	Constant
CO_2	$\Delta F = \alpha \ln (C/C_o)$	$\alpha = 5.35$
CH_4	$\Delta F = \beta (M^{1/2} - M_o{}^{1/2}) - [f(M, N_o) - f(M_o, N_o)]$	$\beta = 0.036$
N_2O	$\Delta F = \varepsilon (N^{1/2} - N_o{}^{1/2}) - [f(M_o, N) - f(M_o, N_o)]$	$\varepsilon = 0.12$
CFC-11	$\Delta F = \lambda (X - X_o)$	$\lambda = 0.25$
CFC-12	$\Delta F = \omega (X - X_o)$	$\omega = 0.32$

The subscript "0" denotes the unperturbed (1750) concentration; $f(M,N) = 0.47 \ln [1 + 2.01 \times 10^{-5} (MN)^{0.75} + 5.31 \times 10^{-15} M (MN)^{1.52}]$; C is CO_2 in ppm, M is CH_4 in ppb, N is N_2O in ppb, X is CFC in ppb; $C_o = 278$ ppm, $M_o = 700$ ppb, $N_o = 270$ ppb, $X_o = 0$.

et al. 1982). Most of the N_2O monitoring studies from crop fields have been carried out in Asia, United States, Canada and some parts of Europe. A summary of some of the available N_2O emission data from crops collected worldwide is presented in Table 8.4 (for rice) and Table 8.5 (all other crops together) separately, as rice cultivation is done under a totally different set of crop management practices. Fodder and forage crops have been kept out of scope of this summary. Emission data clearly indicate that N loss via N_2O from rice fields is generally less than 1% of added fertilizer N except a few cases where it reaches up to 5.6% (Xu et al. 2004). In other crops, N_2O-N loss is more pronounced than rice, but remains generally below 1% or sometimes in between 1 and 2%. N_2O flux differs from experiment to experiment mainly due to variation in water regime, N dose and fertilizer type and various other crop management and inherent soil factors. In agriculture, on an average, 1.25% of the N input (as fertilizer, manure or through biological N fixation) is emitted from the field as N_2O (Bouwman 1996), though there is a large variation ranging from 0.25 to 2.25% in most cases (IPCC 1996a). EPA (1995) uses a factor (mean value of N_2O flux) of 1.17% for N_2O-N loss from added N in agriculture.

8.11 Radiative Forcing of N_2O Emitted from Crop Fields

The Intergovernmental Panel on Climate Change (IPCC) defines climate forcing as "an externally imposed perturbation in the radiative energy budget of the earth climate system through changes in solar radiation, changes in the earth's albedo or changes in atmospheric gases and aerosol particles". Radiative forcing is the difference between the incoming radiation energy and the outgoing radiation energy in a given climate system. A positive forcing i.e. more incoming energy tends to warm the system, while a negative forcing or more outgoing energy tends to cool it. Greenhouse gases, responsible for greenhouse effect and consequently global warming, are standardized for their warming potetials by an index known as global warming potential (GWP). It is a relative scale which compares the concerned gas to the same mass of carbon dioxide (whose GWP is 1, by definition) over a specific time interval. The GWP has been defined as the ratio of the time-integrated radiative forcing from the instantaneous release of 1 kg of a trace substance relative to that of 1 kg of a reference gas, i.e. CO_2 (IPCC 1990).

Notably, GWP depends on the time span over which the potential is calculated. A gas, which is quickly removed from atmosphere may initially have a large effect, but not for longer time periods. Thus, N_2O has a GWP potential of 296 over 100 years, but 175 over 500 years (IPCC 2001) which is not precisely known and hence the values are considered only approximate. Higher the global warming potential (GWP) of a greenhouse gas, more will be its positive radiative forcing than the same amount of other greenhouse gases with lower GWPs. The perturbation to radiative climate forcing, which has the largest magnitude and the least scientific uncertainty, is the forcing related to changes in long-lived and well mixed greenhouse gases, e.g. carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O) and halogens (mainly CFCs).

National Oceanic and Atmospheric Agency (NOAA), USA, has been estimating atmospheric concentrations of global greenhouse gases and their radiative forcing since 1979. The change in the annual mean of total radiative forcing by all long-lived greenhouse gases since pre-industrial era (1750) is used to define the NOAA Annual Greenhouse Gas Index (AGGI), introduced in 2004 (Hofmann et al. 2006a) and updated in 2006 (Hofmann et al. 2006b). AGGI is the ratio of total radiative forcing due to long-lived greenhouse gases for any year for which adequate global measurements exist to that which was present in 1990, the baseline year for Kyoto Protocol. This index is a measure of the interannual changes in conditions that affect CO_2 emission and uptake, CH_4 and N_2O sources and sinks and the decline in the atmospheric abundance of ozone-depleting chemicals related to Montreal Protocol. For 2006, the AGGI was 1.23 (an increase in total radiative forcing of 23% since 1990). The increase in CO_2 forcing alone since 1990 was about 32%. The decrease in methane emissions and CFCs has possibly tampered the increase in net radiative forcing. The AGGI is updated each spring when air samples from all over the globe for the previous year have been obtained and analyzed.

To determine the total radiative forcing of the greenhouse gases for a current year, NOAA uses IPCC (IPCC 2001) recommended expressions to convert greenhouse gas changes, relative to 1750, chosen as the pre-industrial base year, to get instantaneous radiative forcing by using the direct forcing only (Table 8.6). These empirical expressions used for radiative forcing are derived from atmospheric radiative transfer models and generally have an uncertainty of about 10%. Model-dependent feedbacks, for example, due to water vapor and O_3 depletion, are not included. Other spatially heterogeneous, short-lived, climate forcing agents having uncertain global magnitudes, such as aerosols and tropospheric O_3 are also not included here in order to maintain accuracy. Figure 8.2, prepared from NOAA's calculations (http://www.esrl.noaa.gov/gmd/aggi/), shows the radiative forcing for the major gases and a set of 10 minor long-lived halogen gases including CFC-113, CCl_4, CH_3CCl_3, HCFCs 22, 141b and 142b, HFC134a, SF_6 and halons 1211 and 1301. CO_2 dominates the total forcing with CH_4 and the CFCs becoming relatively smaller contributors to the total forcing over time. The five major greenhouse gases account for about 97% of the direct radiative forcing by long-lived greenhouse gas increases since 1750. The remaining 3% is contributed by the 10 minor halogen

Table 8.6 Summary of N_2O emissions from irrigated crop fields (other than rice) at several locations worldwide

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
Iowa (USA)	120 d	Typic haplaquolls, Webster series,	Soybean (cut during study)	No N	* ~0–0.1 kg ha^{-1}	NF	–	Bremner et al. (1981)
				Anhydrous NH_3 (250)	* ~0–1.3 kg ha^{-1}	NF	–	
	120 d	Typic haplaquolls, Canisteo Series	Soybean (cut during study)	No N	* ~0–0.09 kg ha^{-1}	NF	–	
				Anhydrous NH_3 (250)	* ~0–0.68 kg ha^{-1}	NF	–	
	120 d	Typic calciaquolls Harps series	Soybean (cut during study)	No N	* ~0–0.07 kg ha^{-1}	NF	–	
				Anhydrous NH_3 (250)	* ~0–0.68 kg ha^{-1}	NF	–	
Berthoul (USA)	150 d	Mesic aridic argiustoll, Nunn clay loam	Maize	Anhydrous NH_3 (200)	0.3–550 g ha^{-1} d^{-1}	2.5 kg ha^{-1}	1.25	Mosier and Hutchinson (1981)
	150 d	Mesic ustollic haplargids, Wiley fine silty	Sugerbeet	Anhydrous NH_3 (75)	0.3–8.2 g ha^{-1} d^{-1}	NF	–	
Colorado (USA)	180 d	Ustic torriorthens, Calcareous, sandy loam	Barley	No N	3.4 g ha^{-1} d^{-1}	0.54 kg ha^{-1}	–	Mosier et al. (1982)
				Am. Nitrate (56)	5.9 g ha^{-1} d^{-1}	0.93 kg ha^{-1}	0.7 0.4	
				Am. Nitrate (112)	6.7 g ha^{-1} d^{-1}	1.04 kg ha^{-1}	0.4 0.8	
				Am. Nitrate (224)	9.2 g ha^{-1} d^{-1}	1.43 kg ha^{-1}	1.0	
				Sewage-sludge (71)	7.0 g ha^{-1} d^{-1}	1.09 kg ha^{-1}		
				Sewage-sludge (356)	27.0 g ha^{-1} d^{-1}	4.19 kg ha^{-1}		
Maizeell (USA)	90 d	Gloschoric hapladalfs, Silt loam	Maize	No N	* ~0–25 g $ha^{-1}d^{-1}$	0.3 kg ha^{-1}	–	Duxbury and Mcconnaughey (1986)
				No N+C_2H_2	* ~0–125 g $ha^{-1}d^{-1}$	1.6 kg ha^{-1}	–	
				Calcium nitrate (120)	* ~0–15 g $ha^{-1}d^{-1}$	0.3 kg ha^{-1}	0.00	
				Calcium nitrate (120)+C_2H_2	* ~0–280 g $ha^{-1}d^{-1}$	2.0 kg ha^{-1}	0.33	
				Urea (120)	* ~0–175 g $ha^{-1}d^{-1}$	2.5 kg ha^{-1}	1.83	
				Urea+C_2H_2	* ~0–330 g $ha^{-1}d^{-1}$	2.4 kg ha^{-1}	0.66	
Wisconsin (USA)	330 d	Mesic typic hapludalf, Kidder loam	Maize	Cattle manure+Am. Nitrate (168 + 13)	* ~1–48 g ha^{-1} d^{-1}	–	– –	Cates and Keeney (1987)
				Cattle manure+Am. Nitrate (168 + 56)	* ~1–60 g ha^{-1} d^{-1}	–		

Table 8.6 (continued)

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
Colorado (USA)	NF	Mesic ustollic haplargid, Sandy clay laom	Wheat+ fallow	NF	* ~0–2.7 g ha^{-1}d^{-1} * ~0–3.7 g ha^{-1}d^{-1}	0.101 kg ha^{-1} 0.112 kg ha^{-1}	– –	Bronson and Mosier (1993)
	300 d	Aridic arginstoll, Clay	Wheat	No N	–	0.440 kg ha^{-1}	–	
				Urea (50,100,150)	* ~0–37 g ha^{-1}d^{-1}	0.929 kg ha^{-1}	–	
				Urea (50,100,150) +ECC (20 kg CaC_2 ha^{-1})	* ~0–23 g ha^{-1}d^{-1}	0.509 kg ha^{-1}	–	
				Urea (50,100,150)+ DCD(10% N)	* ~ 0–18 g ha^{-1}d^{-1}	0.437 kg ha^{-1}	–	
	150 d	Mesic aridic anguistoll, Clay loam	Maize	No N	–	0.108 kg ha^{-1}	–	
				Urea (218)	* ~ 0–249 kg g ha^{-1}	1.651 kg ha^{-1}	0.7	
				Urea+ECC (20 kg Ca C_2 ha^{-1})	* ~0–35 kg g ha^{-1}	0.483 kg ha^{-1}	0.17	
				Urea+Nitrapyrin (0.5 L ai ha^{-1})	* ~0–115 kg g ha^{-1}	0.980 kg ha^{-1}	0.4	
Rothamsted (UK)	540 d	Batcombe series, Loamy	Wheat	Am. Nitrate (350)	* ~–4–45 ng m^{-2} s^{-1}	1.4 kg ha^{-1}	<1	Yamulki et al. (1995)
Melle (Belgium)	312 d	Sandy laom	Maize	Inorganic N (75)	* ~0–76 g ha^{-1} d^{-1}	2.7 kg ha^{-1}	3.6	Vermoesen et al. (1996)
Ontario (Canada)	240 d	Grey brown luvisol, Silty loam	Alfa alfa	NF	* ~34.2–193.7 ng m^{-2}s^{-1}	NF	–	Burton et al. (1997)
Ontario (Canada)	420 d	Melanic brunisol, Conestogo silt loam	Maize	Am. Nitrate (100)	* ~0– 0.75 kg ha^{-1}	NF	–	Wagner Riddle and Thurtell (1998)
	420 d	Melanic brunisol, Conestogo silt loam	Canola	Am. Nitrate (100)	* ~0–1.25 kg ha^{-1}	NF	–	
	420 d	Melanic brunisol, Conestogo silt loam	Barley	Am. Nitrate (75)	* ~0–2.1 kg ha^{-1}	NF	–	

Table 8.6 (continued)

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
	420 d	Melanic brunisol, Conestogo silt loam	Soybean	Liquid dairy cattle manure (60)	* ~0–3.3 kg ha^{-1}	NF	–	
Canada	180 d	Humic gleysol , Ste Rosalie heavy clay	Maize	No N	* ~0–1.8 g ha^{-1} h^{-1}	NF	–	MacKenzie et al. (1997)
				Urea+Am. Nitrate (170)	* ~0–5.5 g ha^{-1} h^{-1}	NF	–	
				Urea+Am. Nitrate (285)	* ~0–2.9 g ha^{-1} h^{-1}	NF	–	
				Urea+Am. Nitrate (400)	* ~0–6.7 g ha^{-1} h^{-1}	NF	–	
Quebec (Canada)	180 d	Grey brown poodzol , Chicot sandy loam	Maize	No N	* ~0–5 g ha^{-1} h^{-1}	NF	–	MacKenzie et al. (1997)
				Urea+Am. Nitrate (170)	* ~0–7 g ha^{-1} h^{-1}	NF	–	
				Urea+Am. Nitrate (285)	* ~0–13 g ha^{-1} h^{-1}	NF	–	
				Urea+Am. Nitrate (400)	* ~0–19 g ha^{-1} h^{-1}	NF	–	
Ormstown (Canada)	180 d	Humic gleysol, Ormstown silty clay loam	Soybean	No N	* ~0–5.9 g ha^{-1} h^{-1}	NF	–	
				Urea+Am. Nitrate (20)				
				Urea+Am. Nitrate (40)				
	180 d	Humic gleysol, Ormstown silty clay loam	Maize	No N	* ~0–7.2 g ha^{-1} h^{-1}	NF	–	
				Urea+Am. Nitrate (90)				
				Urea+Am. Nitrate (180)				
Ile Parrot (Canada)	180 d	Ste Rosalie clay	Soybean	No N	* ~0–2.4 g ha^{-1} h^{-1}	NF	–	
				Urea+Am. Nitrate (20)				
				Urea+Am. Nitrate (40)				
	180 d	Humic gleysol, Ormstown silty clay loam	Maize	No N	* ~0–2.0 g ha^{-1} h^{-1}	NF	–	
				Urea+Am. Nitrate (90)				
				Urea+Am. Nitrate (180)				
Edinburgh (UK)	60 d	Sandy loam	Spring barley	No N	NF	0.3 kg ha^{-1}	–	McTaggart et al. (1997)
				Urea (120)	NF	0.6 kg ha^{-1}	0.5	
				Urea (120)+DCD (12.5)	NF	0.36 kg ha^{-1}	0.05	
				Am. nitrate (120)	NF	0.39 kg ha^{-1}	0.08	
				Am. nitrate (120)+DCD (12.5)	NF	0.42 kg ha^{-1}	0.1	

Table 8.6 (continued)

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
Ohio (USA)	150 d	Fluventic hapludoll, Fine silty	Maize	Ammonia (180)	* ~5–43 g ha^{-1} d^{-1}	NF	–	Jacinthe and Dick (1997)
	150 d	Fluventic hapludoll, Fine silty	Maize-Soybean	Ammonia (150)	* ~5–90 g ha^{-1} d^{-1}	NF	–	
	150 d	Fluventic hapludoll, Fine silty	Maize-Soybean-wheat-Vetch	Vetch biomass+dairy manure+urea+Am. nitrate (230)	* ~5–47 g ha^{-1} d^{-1}	NF	– –	
El Pinnel (Spain)	75 d	Mesic calcixerollic xerochrept, Bellcaire clay	Maize	Am. Sulphate (45.5)	–40–389 g ha^{-1} d^{-1}	0.360 kg ha^{-1}	0.8	Teira-Esmatges et al. (1998)
Barbens (Spain)	75 d	Mesic typic xerofluvent, Comelles silty clay	Maize	Pig slurry (133)	−3–144 g ha^{-1} d^{-1}	0.497 kg ha^{-1}	0.37	Teira-Esmatges et al. (1998)
Bellpuig (Span)	75 d	Mesic xerollic paleorthid, Seana clay loam	Maize	Pig slurry (112)	1.5–63 g ha^{-1} d^{-1}	0.426 kg ha^{-1}	0.38	
SE Scotland	330 d	Gleysol, Sandy clay loam	Potato	Am. Nitrate+urea (140)	NF	1.2 kg ha^{-1}	0.86	Smith et al. (1998)
	330 d	Gleysol, Sandy clay loam	Spring barley	Am. Nitrate (120)	NF	0.8 kg ha^{-1}	0.67	
	330 d	Gleysol, Sandy clay loam	Wheat	Am. Nitrate (180)	NF	0.3 kg ha^{-1}	0.17	
	330 d	Gleysol, Sandy clay loam	Potato	Am. Nitrate (170) Urea (170)	NF NF	3.1 kg ha^{-1} 3.2 kg ha^{-1}	1.8 1.9	
Munich (Germany)	270 d	Dystric Eutrochrept, Fine loamy	Wheat	Am. Nitrate+urea (100)	NF	1.54 kg ha^{-1}	1.54	Kemp et al. (1998)

Table 8.6 (continued)

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
Ontario (Canada)	120 d	Silty loam	Barley	Am. Nitate (75)	8–18.7 ng m^{-2} s^{-1}	0.828 kg ha^{-1}	1.104	Wagner Riddle and Thurtell (1998)
	120 d	Silty loam	Soybean	–	1.3–11.4 ng m^{-2} s^{-1}	0.197 kg ha^{-1}	–	
	120 d	Silty loam	Canola	Am. Nitrate (100)	–3.3– 10.9 ng m^{-2} s^{-1}	0.585 kg ha^{-1}	0.585	
	120 d	Silty loam	Maize	Am. Nitrate (100)	2.5–22.8	0.523 kg ha^{-1}	0.523	
Jokionen (Finland)	360 d	Loamy sand	Barley	Am. Nitrate (100)	0–230 g ha^{-1} d^{-1}	3.7 kg ha^{-1}	3.7	Syvasalo et al. (1998)
	360 d	Clay	Barley	Am. Nitrate (100)	0–60 g ha^{-1} d^{-1}	3.7 kg ha^{-1}	3.7	
Saxony (Germany)	360 d	Gleyic luvisol, silty loam	Wheat	No N	* ~0–340 μg m^{-2} h^{-1}	0.184 kg ha^{-1}	–	Rover et al. (1998)
				Am. Nitrate+urea soln. (220)	~0–490 μg m^{-2} h^{-1}	349.5 kg ha^{-1}	0.158	
Faisalabad (Pakistan)	90 d	Sandy clay loam	Maize	Urea (100)	−0.94–1.53 g ha^{-1} h^{-1}	0.16 kg ha^{-1}	0.16	Mahmood et al. (1998)
			Wheat	Urea (100)	0.05–0.39 g ha^{-1}h^{-1}	0.49 kg ha^{-1}	0.49	
Schleswig-Holstein (Germany)	360 d	Podzoli-cambic Arenosol	Maize	Cattle slurry (333 kg ha^{-1})	* ~0–99 g ha^{-1} d^{-1}	2.1 kg ha^{-1}	0.6	Mogge et al. (1999)
Ingham (Australia)	104 d	Clay	Sugarcane	Am. Sulphate (160)	* ~0–150 g ha^{-1} d^{-1}	1.082 kg ha^{-1}	0.676	Weier (1999)
				Urea split (60 + 60)	* ~0–240 g ha^{-1} d^{-1}	1.934 kg ha^{-1}	1.61	
				Urea (160)	* ~0–50 g ha^{-1} d^{-1}	0.415 kg ha^{-1}	0.259	
				Am. Sulphate (160)	* ~0–90 g ha^{-1} d^{-1}	0.951 kg ha^{-1}	0.594	
				Urea split (60 + 60)	* ~0–250 g ha^{-1} d^{-1}	1.562 kg ha^{-1}	1.30	
				Urea (160)	* ~0–150 g ha^{-1} d^{-1}	1.143 kg ha^{-1}	0.714	
Italy	150 d	Vertic ustochrept, silty clay	Maize	No N	* ~0–0.5 mg m^{-2} d^{-1}	0.65 kg ha^{-1}	–	Arcara et al. (1999)
				Pig slurry (225)	* ~0–3.0 mg m^{-2} d^{-1}	1.28 kg ha^{-1}	0.28	
				Pig slurry +urea (225 + 225)	* ~0–4.5 mg m^{-2} d^{-1}	1.30 kg ha^{-1}	0.118	
				Urea (225)	* ~0–3.0 mg m^{-2} d^{-1}	1.84 kg ha^{-1}	0.528	

Table 8.6 (continued)

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
Midwest USA	NF	NF	Maize-wheat-soybean	No N (Conventional Tillage)	* ~ 3.2 g ha^{-1} d^{-1}	NF	–	Robertson et al. (2000)
				No N (No Tillage)	* ~ 3.4 g ha^{-1} d^{-1}	NF	–	
				No N (Low chemical input)	* ~ 3.5 g ha^{-1} d^{-1}	NF	–	
			Alfa alfa	No N (Organic)	* ~ 3.4 g ha^{-1} d^{-1}	NF	–	
Shenyang (China)	270 d	Meadow brown	Maize	Urea+stamening (345)	-19.9–557.2 μg m^{-2} h^{-1}	4.52 kg ha^{-1}	1.3 5.6	Chen et al. (2000)
			Soy-bean	Urea (35)	-20.3–217.6 μg m^{-2} h^{-1}	1.99 kg ha^{-1}		
(Thailand)	120 d	Typic calciustalfs	Maize	No N	* ~ 0–8 μg m^{-2} h^{-1}	0.1 kg ha^{-1}	–	Watanabe et al. (2000)
				Am. Sulphate (62.5)	* ~ 1–61 μg m^{-2} h^{-1}	0.403 kg ha^{-1}	0.48	
	120 d	Typic haplustoxs	Maize	No N	* ~ 0–11 μg m^{-2} h^{-1}	0.122 kg ha^{-1}	–	
				Am. Sulphate (62.5)	* ~ 1–63 μg m^{-2} h^{-1}	0.351 kg ha^{-1}	0.366	
	120 d	T. tropaqualfs	Maize	No N	* ~ 0–15 μg m^{-2} h^{-1}	0.109 kg ha^{-1}	–	
				Am. Sulphate (62.5)	* ~ 1–58 μg m^{-2} h^{-1}	0.289 kg ha^{-1}	0.288	
	120 d	T. ochraqualfs	Maize	No N	* ~ 0–25 μg m^{-2} h^{-1}	0.119 kg ha^{-1}	–	
				Am. Sulphate (62.5)	* ~ 5–30 μg m^{-2} h^{-1}	0.196 kg ha^{-1}	0.123	
Giesseen (Germany)	150 d	Clay loam, fluvisol	Barley	Am. Sulphate nitrate (90)	~ 0–32 g ha^{-1} d^{-1}	0.132 kg ha^{-1}	0.146	Weiske et al. (2001)
				Am. Sulphate nitrate (90)+DMPP (1.6%)	* ~ 0–18 g ha^{-1} d^{-1}	0.093 kg ha^{-1}	0.103	
				Am. Sulphate nitrate (90)+DCD (16%)	* ~ 0–22 g ha^{-1} d^{-1}	0.079 kg ha^{-1}	0.087	
	150 d	Clay loam, fluvisol	Maize	Am. Sulphate nitrate (160)	* ~ 0–37 g ha^{-1} d^{-1}	0.340 kg ha^{-1}	0.212	Weiske et al. (2001)
				Am. Sulphate nitrate (90)+DMPP (1.6%)	* ~ 0–22 g ha^{-1} d^{-1}	0.267 kg ha^{-1}	0.296	
				Am. Sulphate nitrate (90)+DCD (16%)	* ~ 0–31 g ha^{-1} d^{-1}	0.181 kg ha^{-1}	0.201	

Table 8.6 (continued)

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
Giesseen (Germany)	150 d	Clay loam, fluvisol	Wheat	Am. Sulphate nitrate (180)	* ~0–43 g ha^{-1} d^{-1}	0.353 kg ha^{-1}	0.196	Weiske et al. (2001)
				Am. Sulphate nitrate (180)+DMPP (1.6%)	* ~0–48 g ha^{-1} d^{-1}	0.252 kg ha^{-1}	0.140	
				Am. Sulphate nitrate (180)+DCD (16%)	* ~0–24 g ha^{-1} d^{-1}	0.167 kg ha^{-1}	0.092	
Bavaria (Germany)	120 d	Brown earth, silty loam	Wheat	Am. Sulphate nitrate (160)	~0.2–1.5 g ha^{-1} d^{-1}	NF	–	Linzmeier et al. (2001)
				Am. Sulphate nitrate (160)+DCD (16%)	~0.2–~0.7 g ha^{-1} d^{-1}	NF	–	
				Am. Sulphate nitrate (160)+DMPP (1.6%)	~0.15–0.9 g ha^{-1} d^{-1}	NF	–	
				Calcium Am. Nitrate (160)	~3–38 g ha^{-1} d^{-1}	NF	–	
Munich (Germany)	730 d	Fine loamy, Dystric Eutrochrept	Potato	CAN+UAN (50)	* ~0–260 μg m^{-2} h^{-1}	NF	–	Ruser et al. (2001)
				CAN+UAN (150)	* ~0–500 μg m^{-2} h^{-1}	NF	–	
	730 d	Fine loamy, Dystric Eutrochrept	Wheat	CAN+UAN (90)	* ~0–490 μg m^{-2} h^{-1}	NF	–	
				CAN+UAN (180)	* ~0–943 μg m^{-2} h^{-1}	NF	–	
	730 d	Fine loamy, Dystric Eutrochrept	Maize	CAN+UAN (65)	* ~0–190 μg m^{-2} h^{-1}	NF	–	
				CAN+UAN (130)	* ~0–200 μg m^{-2} h^{-1}	NF	–	
Tsukuba (Japan)	120 d	Hydrichapludand, clay loam	Maize	No N	* ~0–20 μg m^{-2} h^{-1}	0.125 kg ha^{-1}	–	Yan et al. (2001)
				Urea (150)	* ~5–125 μg m^{-2} h^{-1}	0.544 kg ha^{-1}	0.28	
				Polyeolefin coated urea (150)	* ~0–115 μg m^{-2} h^{-1}	0.537 kg ha^{-1}	0.27	
				Urea (250)	* ~5–45 μg m^{-2} h^{-1}	0506 kg ha^{-1}	0.15	
Gyrstinge (Denmark)	330 d	Sandy loam	Barley	No N	* ~0–240 μg m^{-2} h^{-1}	NF	–	Ambus et al. (2001)
				Sewage sludge (81)	* ~0–330 μg m^{-2} h^{-1}	NF	–	

Table 8.6 (continued)

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
Yingtan (China)	150 d	Red clay, Udic ferrosol	Peanut	No N	* ~5–55 μg m^{-2}h^{-1}	0.52 kg ha^{-1}	–	Xiong et al. (2002b)
				Urea (103)	* ~3–140 μg m^{-2}h^{-1}	0.86 kg ha^{-1}	0.34	
	150 d	Red clay, Udic ferrosol	Pea	No N	* ~3–16 μg m^{-2}h^{-1}	0.29 kg ha^{-1}	–	
				Urea (52)	* ~4–30 μg m^{-2}h^{-1}	0.42 kg ha^{-1}	0.25	
	150 d	Red clay, Udic ferrosol	Soybean	No N	* ~2–50 μg m^{-2}h^{-1}	0.64 kg ha^{-1}	–	
				Urea (34)	* ~5–68 μg m^{-2}h^{-1}	0.86 kg ha^{-1}	0.64	
	150 d	Red clay, Udic ferrosol	Rapeseed	No N	* ~3–48 μg m^{-2}h^{-1}	0.49 kg ha^{-1}	–	
				Urea (103)	* ~5–50 μg m^{-2}h^{-1}	0.58 kg ha^{-1}	0.094	
New Delhi (India)	90 d	Sandy laom, Typic ustochrept	Wheat	No N	* ~0–11.5 mg m^{-2} d^{-1}	0.75 kg ha^{-1}	–	Majumdar et al. (2002)
				Urea (120)	* ~1–17.5 mg m^{-2} d^{-1}	1.43 kg ha^{-1}	0.56	
				Urea+DCD (10% N)	* ~1–12.5 mg m^{-2} d^{-1}	1.09 kg ha^{-1}	0.28	
				Urea+Am. Thiosulphate (15%N)	* ~0–14.2 mg m^{-2} d^{-1}	1.0 kg ha^{-1}	0.20	
				Neem coated urea	* ~0–15.5 mg m^{-2} d^{-1}	1.36 kg ha^{-1}	0.50	
				Nimin coated urea	* ~0–13.0 mg m^{-2} d^{-1}	1.19 kg ha^{-1}	0.36	
Shenyang (China)	120 d	Meadow brown, silt loam	Soybean	No N	* ~13–29 μg m^{-2} h^{-1}	NF	–	Chen et al. (2002)
				Manure (559)	* ~0–88 μg m^{-2} h^{-1}	NF	–	
	120 d	Meadow brown, silt loam	Maize	No N	* ~0–40 μg m^{-2} h^{-1}	NF	–	
				Manure (150)	* ~40–300 μg m^{-2} h^{-1}	NF	–	
				Manure (250)	* ~40–625 μg m^{-2} h^{-1}	NF	–	
Turitea (New Zealand)	120 d	Typic andoaqualf, silt loam	Maize+cont. tillage (CT)	Nitrophoska+urea (44)	* ~0.08–316 mg m^{-2}h^{-1}	NF	–	Chaudhary et al. (2001)
	120 d		Maize+normal tillage (NT)	Nitrophoska+urea (44)	* ~0.1–50.12 mg m^{-2}h^{-1}	NF	–	
	150 d		Oat+CT	Nitrophoska+urea (54)	12.59–100 mg m^{-2}h^{-1}	NF	–	
	150 d		Oat+NT	Nitrophoska+urea (54)	1.25–1258 mg m^{-2}h^{-1}	NF	–	

Table 8.6 (continued)

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
Carolina (USA)	13 d	Sandy loam	Soybean	Swine effluent (274)	0–12.5 kg ha^{-1}d^{-1}	4.1 kg ha^{-1}	1.5	Sharpe and Harper (2002)
New Delhi (India)	105 d	Typic ustochrept, Sandy loam	Wheat	No N (s)	* ~2–11.5 kg ha^{-1}d^{-1}	0.370 kg ha^{-1}	–	Pathak et al. (2002)
				Urea (120) (s)	* ~4–16 kg ha^{-1}d^{-1}	0.766 kg ha^{-1}	0.33	
				Urea+FYM (60 + 60) (s)	* ~2–16 kg ha^{-1}d^{-1}	0.796 kg ha^{-1}	0.36	
				Urea+DCD (108 + 12) (s)	* ~1.5–14 kg ha^{-1}d^{-1}	0.619 kg ha^{-1}	021	
	105 d	Typic ustochrept, Sandy loam	Wheat	No N (ID)	* ~1–13 kg ha^{-1}d^{-1}	0.308 kg ha^{-1}	–	
				Urea (120) (ID)	* ~4–21 kg ha^{-1}d^{-1}	0.711 kg ha^{-1}	0.34	
				Urea+FYM (60 + 60) (ID)	* ~4–22 kg ha^{-1}d^{-1}	0.725 kg ha^{-1}	0.35	
				Urea+DCD (108 + 12) (ID)	* ~3–16.5 kg ha^{-1}d^{-1}	0.550 kg ha^{-1}	0.20	
New Delhi (India)	105 d	Typic ustochrept, sandy loam	Wheat	No N (s)	* ~0–12 g ha^{-1} d^{-1}	0.394 kg ha^{-1}	–	Pathak et al. (2003)
				Urea (120) (s)	* ~0–14 g ha^{-1} d^{-1}	0.857 kg ha^{-1}	0.39	
				Urea+FYM (60 + 60) (s)	* ~0–16 g ha^{-1} d^{-1}	0.846 kg ha^{-1}	0.38	
				Urea+DCD (108 + 12) (s)	* ~0–13 g ha^{-1} d^{-1}	0.639 kg ha^{-1}	0.20	
	105 d	Typic ustochrept, sandy loam	Wheat	No N (ID)	* ~0–11 g ha^{-1} d^{-1}	0.322 kg ha^{-1}	–	
				Urea (120) (ID)	* ~0–13 g ha^{-1} d^{-1}	0.753 kg ha^{-1}	0.36	
				Urea+FYM (60 + 60) (ID)	* ~0–12.5 g ha^{-1} d^{-1}	0.696 kg ha^{-1}	0.31	
				Urea+DCD (108 + 12) (ID)	* ~0–11 g ha^{-1} d^{-1}	0.566 kg ha^{-1}	0.20	
Shanxii (China)	120 d	Orthic anthrosol	Wheat	Unspecified (150)	* ~0–47 μg m^{-2} h^{-1}	NF	–	Bai et al. (2003)
Tsukuba (Japan)	360 d	Andisol	Pac Choi	Poultry manure (150)	* ~0–370 μg m^{-2} h^{-1}	1.84 kg ha^{-1}	1.23	Akiyama and Tsuruta (2003)
				Swine manure (150)	* ~0–185 μg m^{-2} h^{-1}	0.62 kg ha^{-1}	041	
				Urea (150)	* ~0–200 μg m^{-2} h^{-1}	045 kg ha^{-1}	0.30	
Hidalgo (Mexico)	120 d	Loamy	Maize	Urea (80)	–484–385 μg m^{-2} h^{-1}	NF	–	Longoria-Ramirez et al. (2003)
			wheat	Am. Sul (100)	–63.82–86.24 μg m^{-2} h^{-1}	NF	–	
Saint Marcsur-Riechelieu (Canada)	120 d	Orthic humic gleysol, clay loam	Maize	Anhydrous NH_3 (150)	* ~0–3.5 mg N m^{-2} h^{-1}	NF	–	Rochette et al. (2004a)
				Anhydrous NH_3 (250)	* ~0–4.5 mg N m^{-2} h^{-1}	NF	–	

Table 8.6 (continued)

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
Jokionen (Finland)	360 d	Loamy sand	Barley	Am. Nitrate (100)	0–230 g ha^{-1} d^{-1}	3.7 kg ha^{-1}	3.7	Syvasalo et al. (2004)
	360 d	Clay	Barley	Am. nitrate (100)	0–60	3.7 kg ha^{-1}	3.7	
Waeningen (The Netherlands)	180 d	Typic endoaquoll, Sandy	Maize	No N	NF	0.14 kg ha^{-1}	–	van Groenigen et al. (2004)
				CAN (75)	NF	0.017 kg ha^{-1}	0.04	
				CAN (113)	NF	0.29 kg ha^{-1}	0.13	
				CAN (150)	NF	0.24 kg ha^{-1}	0.07	
	180 d	Typic endoaquoll, Sandy	Maize	Cattle slurry (104)	0–29 g ha^{-1} d^{-1}	0.47 kg ha^{-1}	0.31	
				Cattle slurry (156)	0–85 g ha^{-1} d^{-1}	1.28 kg ha^{-1}	0.73	
				Cattle slurry (209)	0–105 g ha^{-1} d^{-1}	1.7 kg ha^{-1}	0.75	
				Cattle slurry (261)	0–110 g ha^{-1} d^{-1}	1.92 kg ha^{-1}	0.68	
	180 d	Typic endoaquoll, Sandy	Maize	CAN+ Cattle slurry (38 + 156)	0–29 g ha^{-1} d^{-1}	0.7 kg ha^{-1}	0.21	
				CAN+ Cattle slurry (75 + 104)	0–28 g ha^{-1} d^{-1}	0.52 kg ha^{-1}	0.29	
				CAN+ Cattle slurry (113 + 52)	0–30 g ha^{-1} d^{-1}	0.59 kg ha^{-1}	0.27	
	180 d	Typic endoequept, heavy clay	Maize	CAN+ Cattle slurry (38 + 147)	0–55 g ha^{-1} d^{-1}	3.47 kg ha^{-1}	1.01	
				CAN+ Cattle slurry (75 + 98)	* ∼0–72 g ha^{-1} d^{-1}	3.24 kg ha^{-1}	0.93	
				CAN+ Cattle slurry (1138 + 49)	0–165 g ha^{-1} d^{-1}	6.63 kg ha^{-1}	3.11	
Quebec (Canada)	360 d	Aeric haplaquept, Fine loamy	Maize	Am nitrate (150)	* ∼0–0.04 mg m^{-2} h^{-1}	1.74 kg ha^{-1}	1.16	Rochtte et al. (2004b)
				Pig slurry (200	* ∼0–3.5 mg m^{-2} h^{-1}	9.22 kg ha^{-1}	4.61	
Kannus (Finland)	360 d	Peaty organic	Barley	Inorganic N (200)	* ∼0–29 mg m^{-2} d^{-1}	8.48 kg ha^{-1}	4.24	Maljanen et al. (2003)

Table 8.6 (continued)

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
Harlaka (Canada)	360 d	Typic humaquept Clay Sombric fragiorthod, Loam	Alfa alfa Timothy Soybean Alfa alfa Timothy Soybean	No N	* ~0–0.58 mg m^{-2} h^{-1}	1.35 kg ha^{-1}	–	Rochette et al. (2004b)
	120 d			No N	* ~0–0.63 mg m^{-2} h^{-1}	1.03 kg ha^{-1}	–	
				No N	* ~0–0.78 mg m^{-2} h^{-1}	3.08 kg ha^{-1}	–	
				No N	* ~0–0.12 mg m^{-2} h^{-1}	0.67 kg ha^{-1}	–	
				No N	* ~0–0.1 mg m^{-2} h^{-1}	0.8 kg ha^{-1}	–	
				No N	* ~0–0.15 mg m^{-2} h^{-1}	0.66 kg ha^{-1}	–	
New Delhi (India)	120 d	Typic ustochrept, Sandy loam	Wheat	Urea (120)	* ~3.7–13 g ha^{-1} d^{-1}	0.66 kg ha^{-1}	0.55	Malla et al. (2005)
				Urea+hydrquinone (120)	* ~3.5–14.8 g ha^{-1} d^{-1}	0.62 kg ha^{-1}	0.51	
				Urea+neem cake (108 + 12)	* ~3–13.5 g ha^{-1} d^{-1}	0.52 kg ha^{-1}	0.43	
				Urea+thiosulphate (120)	* ~2.–12.8 g ha^{-1} d^{-1}	0.48 kg ha^{-1}	0.40	
				Urea+CCC (120)	* ~3–13 g ha^{-1} d^{-1}	0.58 kg ha^{-1}	0.48	
				Neem oil coated urea (120)	* ~3–12.7 g ha^{-1} d^{-1}	0.56 kg ha^{-1}	0.46	
				Urea+DCD (108 + 12)	* ~2–12.8 g ha^{-1} d^{-1}	0.47 kg ha^{-1}	0.39	
Lucknow (India)	120 d	Sandy loam	Wheat Mustard Chickpea Faba bean	Cowdung+Inorganic N (150 + 20)	* ~2.5–16 mg m^{-2} d^{-1}	NF	–	Verma et al. (2006)
				Cowdung+Inorganic N (150 + 20)	* ~1–22 mg m^{-2} d^{-1}	NF	–	
				Cowdung+Inorganic N (150 + 20)	* ~1.5–22.5 mg m^{-2} d^{-1}	NF	–	
				Cowdung+Inorganic N (150 + 20)	* ~1–17 mg m^{-2} d^{-1}	NF	–	

Table 8.6 (continued)

Site (Country)	Time span	Soil type	Crop	Fertilizer-N* (kg N ha^{-1})	N_2O-Flux	Total N_2O-N emission	N lost (%)	Reference
Vihti, Finland	150 d	Vertic cambisol	Barley	Cattle slurry (157)	* ~0–66 g ha^{-1} d^{-1}	1.1 kg ha^{-1}	0.70	Perala et al. (2006)
				Inj. cattle slurry+Am. Nitrate (129)	* ~0–40 g ha^{-1} d^{-1}	0.66 kg ha^{-1}	0.51	
				Cattle slurry+Am. Nitrate (129)	* ~0–20 g ha^{-1} d^{-1}	0.44 kg ha^{-1}	0.31	
				Am. Nitrate (100)	* ~0–10 g ha^{-1} d^{-1}	0.29 kg ha^{-1}	0.29	
Arganda del Ray (Spain)	150 d	Typic xerofluvent, sandy loam	Maize	No N	* ~0–7.5 mg m^{-2} d^{-1}	5.98 kg ha^{-1}	–	Meijide et al. (2007)
				Urea (175)	* ~0–12.5 mg m^{-2} d^{-1}	8.57 kg ha^{-1}	1.49	
				Pig slurry (175)	* ~2–10 mg m^{-2} d^{-1}	8.27 kg ha^{-1}	1.3	
				Digested pig slurry (175)	* ~1–8 mg m^{-2} d^{-1}	7.7 kg ha^{-1}	0.9	
				Pig slurry+DCD (175)	* ~1–9 mg m^{-2} d^{-1}	6.7 kg ha^{-1}	0.41	
				Composted pig slurry+urea (300)	* ~2–12.5 mg m^{-2} d^{-1}	9.28 kg ha^{-1}	1.88	
				municipal solid waste+urea (300)	* ~2–9 mg m^{-2} d^{-1}	8.57 kg ha^{-1}	0.66	
Fengqiu (China)	120 d	Aqyic inceptisol, sandy loam	Maize	No N	* ~0–80 μg m^{-2} h^{-1}	0.437 kg ha^{-1}	–	Ding et al. (2007)
				Urea (150)	* ~20–820 μg m^{-2} h^{-1}	2.008 kg ha^{-1}	1.05	
				Urea (250)	* ~10–1480 μg m^{-2} h^{-1}	3.780 kg ha^{-1}	1.33	
	120 d	Aqyic inceptisol, sandy loam	Wheat	No N	* ~–10–20 μg m^{-2} h^{-1}	0.157 kg ha^{-1}	–	
				Urea (150)	* ~–10–65 μg m^{-2} h^{-1}	0.394 kg ha^{-1}	0.26	
				Urea (250)	* ~–5–73 μg m^{-2} h^{-1}	0.602 kg ha^{-1}	0.24	
Madrid (Spain)	200 d	Typic xerofluvent, sandy loam	Maize	No N	* ~1–4 mg m^{-2} d^{-1}	4.66 kg ha^{-1}	–	Lopez-Fernandez et al. (2007)
				Surface pig manure (170)	* ~1–7 mg m^{-2} d^{-1}	5.08 kg ha^{-1}	0.24	
				Incorporated pig manure (170)	* ~1–9 mg m^{-2} d^{-1}	3.71 kg ha^{-1}	–	
				Sheep manure (170)	* ~1–7 mg m^{-2} d^{-1}	3.77 kg ha^{-1}	–	
				MSW (170)	* ~0–9 mg m^{-2} d^{-1}	5.9 kg ha^{-1}	0.72	
				Urea (170)	* ~2–23 mg m^{-2} d^{-1}	2.9 kg ha^{-1}	–	

NF – Not found; CAF – Controlled availability fertilizers; CCC – Wax coated calcium carbide; ECC – Encapsulated calcium carbide; MSW – Municipal solid waste; CAN – Calcium ammonium nitrate; UAN – Urea ammonium nitrate; s – saturated; ID – Intermittent drainage; * Approximate values, deduced from emission trend presented in graphs.

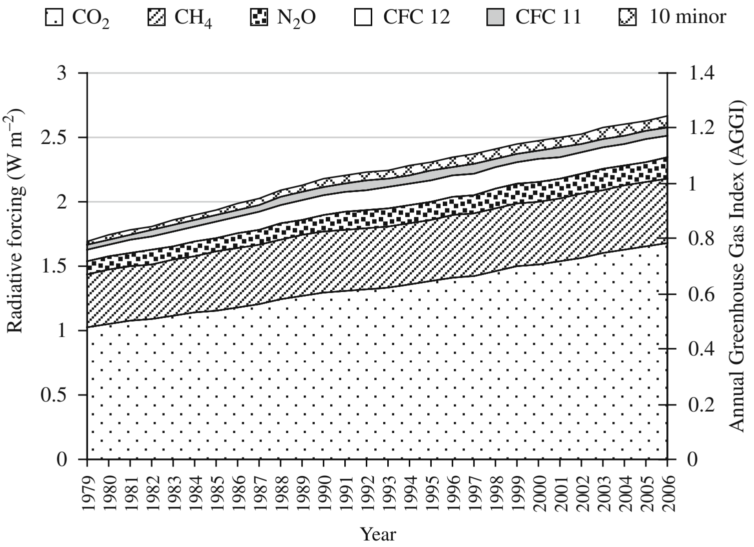

Fig. 8.2 Radiative forcing of all the long-lived greenhouse gases relative to 1750

gases. Of the five long-lived greenhouse gases that contribute 97% to radiative climate forcing, CO_2 and N_2O are the only ones that continue to increase at a regular rate. The contribution to radiative forcing by methane and CFCs has been nearly constant or declining, respectively, in recent years. While the radiative forcing of the long-lived, well-mixed greenhouse gases increased by about 22% from 1990 to 2006 ($\sim$0.50 watts m^{-2}), CO_2 has accounted for about 80% of this increase ($\sim$0.40 watts m^{-2}). Had the ozone-depleting gases not been regulated by the Montreal Protocol, it is estimated that climate forcing would have been as much as 0.2 watt m^{-2} higher (Velders et al. 2007), or about one-half of the increase in radiative forcing due to CO_2 alone since 1990. According to IPCC estimates, nitrous oxide has caused an atmospheric radiative forcing of $0.15\,Wm^{-2}$ or 6% of the enhanced radiative forcing by well-mixed greenhouse gases from pre-industrial to present times. To total radiative foring of $2.43\,W\,m^{-2}$, other contributions are 1.46, 0.48 and $0.34\,W\,m^{-2}$ by CO_2, CH_4 and halocarbons, respectively (IPCC 2001).

Radiative forcing by N_2O emitted from crop fields would vary widely from experiment to experiment due to wide variability in the emissions itself. Even though N_2O is much more potent than CH_4, another major greenhouse gas contributed by crop fields, seasonal emission per unit area of the former is lesser under rice, for example, and so is its radiative forcing, calculated as kg-CO_2 equivalent (Ghosh et al. 2003; Malla et al. 2005; Bhatia et al. 2005). When radiative forcing is calculated from the data of Ghosh et al. (2003), CH_4 had much higher global warming potential than N_2O on all time scales on seasonal basis but when calculated from Cai et al. (1999), N_2O had higher radiative forcing on hourly emission basis than CH_4 on 100 and 500-year time scales, both under rice. But, the combined seasonal radiative forcing of CH_4 and N_2O is quite significant and considering the large area under crop cultivation worldwide, the contribution of crop cultivation to global warming may be appreciable. Yan et al. (2003) have found that total GWP of CH_4 and N_2O emissions in the rice-growing season, under the condition of increasing organic matter application (such as that of 4.5 t hm^{-2}), with drainage only 60% of that of permanent flooding. Thus, it can be concluded that intermittent drainage may be an effective strategy for the minimization of total radiative forcing from emissions of N_2O and CH_4 from rice fields. On the other hand, N_2O emissions from crops grown under aerobic soil condition are often more than CH_4 emissions, latter even showing negative emissions sometimes (Bronson and Mosier 1993). In many occasions, N_2O emissions have been found to be more from wheat fields than rice, two of the most important crops worldwide, implying that CO_2-equivalent emissions and radiative forcing of N_2O could be higher from wheat cultivation under a given set of climate and soil conditions (Bhatia et al. 2005). Since global crop production needs to be increased to feed world's burgeoning population and it implies that high nitrogen input to crop cultivation remains unabated and hence positive radiative forcing of atmosphere will continue unabated. Efforts are to be made to optimize CH_4 and N_2O emission trade-off from crop fields, so that their combined radiative forcing remains at a minimum.

8.12 Strategies for N_2O Mitigation from Crop Fields

The underlying principle of N_2O mitigation from agriculture is increasing fertilizer N use efficiency. Several practices have been followed for the last three-four decades to increase N use efficiency in field crops even before the days of N_2O monitoring, since N use efficiency has always been of prime importance in crop production (Katyal et al. 1985; DeDatta 1995). According to IPCC (1995), by better matching of N supply to crop demand and more closely integrating animal waste and crop-residue management with crop production, N_2O emission could be reduced by about 0.38 Mt N_2O-N from agriculture, while by using improved techniques, like controlled release fertilizers, nitrification inhibitors, timing and water management, additional 0.30 Mt N_2O-N can be reduced (Table 8.7). Regional or country wise mitigation strategies have also been mooted by several research groups e.g. Follett et al. (2005) for USA and Gregorich et al. (2005) for Canada.

Several strategies have been also formulated to mitigate N_2O emissions from rice cultivation (Beauchamp 1997; Mosier et al. 1996). Majumdar (2003) has proposed several N_2O mitigation strategies for irrigated rice. Exclusive rice agronomic practices, like water management, suitable fertilizer placements (Schnier 1995) and common practices, like coated urea (Majumdar et al. 2000), nitrification inhibitors (Kumar et al. 2000; Ghosh et al. 2003; Mosier et al. 1994), have been tried to increase N-use efficiency in rice, many of which were able to increase N-use efficiency and simultaneously mitigate N_2O emissions.

Table 8.7 A list of practices to improve use efficiency of synthetic fertilizer and manure N in agriculture and expected reduction of N_2O emissions assuming global application of mitigation practices (Mt N yr^{-1}) (IPCC 1995)

Practices followed	Estimated decrease in N_2O emissions
Match N supply with crop demand	0.24^a
Use soil/plant testing to determine fertilizer N needs	
Minimize fallow periods to limit mineral N accumulation	
Optimize split application schemes	
Match N application to reduced production goals in regions of crop over-production	
Tighten N flow cycles	0.14^b
Integrate animal and crop production systems in terms of manure reuse in plant production	
Maintain plant residue N on the production site	
Use advanced fertilization techniques	0.15^c
Controlled release fertilizers	
Place fertilizers below the soil surface	
Foliar application of fertilizers	
Use nitrification inhibitors	
Match fertilizer type to seasonal precipitation	
Optimize tillage, irrigation and drainage	0.15^d
Total	**0.68**

[a] Assumed that fertilizer N use efficiency can be increased to save 20% of N applied in North America, Europe and FSU (CAST 1992; Doerge et al. 1991; Iserman 1994; Peoples et al. 1995).

[b] Tightening N cycles may decrease the need for 20% of the N that is used currently in North America, Europe and FSU, thus saving 20% of fertilizer and reducing N_2O from manure by the same amount where applicable (Buresh et al. 1993; Iserman 1994).

[c] Controlled release fertilizers (Minami 1994), nitrification inhibitors (Bronson et al. 1992; Keerthisinghe et al. 1993; McTaggart et al. 1994; Minami 1994) and matching fertilizer type with seasonal precipitation can decrease N_2O emissions in the range of 40–90%. We assume that 10% of all fertilizer-derived N_2O production can be decreased by 50%.

[d] There is little published data to confirm this assumption (Granli and Bockman 1994). A conservative assumption of a 5% decrease, that can be achieved globally, is used.

8.13 Concluding Remarks and Future Research Needs

We are in the fourth decade of nitrous oxide monitoring from crop fields and this period has witnessed an evolution of N_2O monitoring methods and detectors, tapping of possibly all pathways of N_2O loss from crop fields, development of N_2O mitigation strategies, development of N_2O simulation models and streamlining of methodologies and calculations for regional and global N_2O budgets. There are several doubts and uncertainties still, but no doubt, monitoring, quantification and prediction are better than ever before. This is a welcome development keeping in view the gloomy future under the shadows of enhanced global warming predicted by IPCC (http://www.ipcc.ch). After appreciating the importance and magnitude of N_2O emissions from crop fields, primary emphasis has been laid on N_2O mitigation management and practices that ensure no yield reduction, negligible environmental damage and no burden on financial resources. A reduction in application of N fertilizers and organic manures would surely reduce N_2O emissions from crops, but would also reduce crop production unless either acreage or yield potential of crop is increased. Though nitrification inhibitors and slow release N-fertilizers can mitigate N_2O emission, they are not popularly used in farmers' fields due to lack of publicity, non-availability, high price etc. Farmers need to be enlightened about the benefits of nitrification inhibitors and slow release N-fertilizers in N-use efficiency and crop yield through proper extension activities and policy changes in favour of their use could lead to a larger scale application.

Simultaneous monitoring of all the greenhouse gases (e.g. N_2O, CH_4 and CO_2) emitted from crop fields and their control to keep the total CO_2 equivalent emissions at minimum should be the target. Since several of the control strategies of these gases are conflicting i.e. they minimize one but favour another, it puts an onus on the researchers to formulate special control measures to keep the overall CO_2 equivalent emissions at minimum. Further, mitigation practices should be targeted for the entire cropping system, by adjustments, like no fallowing, various other post-harvest land management practices, no excessive fertilization and more use of nitrification and urease inhibitors and slow release/controlled availability N-fertilizers for better N use efficiency and crop yield. The strategies should be effective, easily applicable, technically feasible, remunerative, less time taking and at the same time easily understood and accepted by farmers. Labor requirement, effects on crop yield and soil fertility and short and long term environmental sustainability are other important considerations. More research impetus and funding on the greenhouse gas mitigation have to be generated. Even IPCC (IPCC 1996b) has recognized that only a little national and international funding is available for extensive research in this area, which is needed to make realistic regional and global N_2O budgets.

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Chapter 9
Quantifying Direct N_2O Emissions from Paddy Fields During Rice Growing Season in Mainland China in 1980s and 1990s

Jianwen Zou, Yao Huang and Yanyu Lu

9.1 Introduction

Nitrous oxide (N_2O) is one of key greenhouse gases that cause global warming. It continues to rise at a rate of approximate 0.26% per year and has reached a concentration of 319 ppb (10^{-9} mol mol^{-1}) in 2005 (IPCC 2007a). Agriculture accounts for about 60% of global anthropogenic N_2O emissions. Globally, agricultural N_2O emissions have increased by nearly 17% from 1990 to 2005 (IPCC 2007b), and are projected to increase by 35–60% up to 2030 due to increased nitrogen fertilizer use and increased animal manure production (FAO 2003). The emissions of N_2O that result from anthropogenic N inputs, occur through a direct pathway (i.e. directly from soils to which the N is added), and through two indirect pathways: volatilization of compounds, such as NH_3 and NO_X and subsequent redeposition, and through leaching and runoff. Relative to the indirect pathways, the direct emission contributes most to the agricultural N_2O sources (Zheng et al. 2004). Thus, a good estimate of direct N_2O emission from agricultural fields will help assess its global source strength.

The United Nations Framework Convention on Climate Change (UNFCCC) obligates all signatory parties to periodically provide national inventories on emissions and/or removals of greenhouse gases that are not controlled by the Montreal Protocol, such as N_2O releases from crop production. Accordingly, the Intergovernmental Panel on Climate Change (IPCC) developed *Revised 1996 IPCC guidelines for national and Greenhouse Gases Inventories* (IPCC 1997) and *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000). The present IPCC guidelines for estimating direct N_2O emission from fields include a default emission factor (EF) of 1.25% (0.25–2.25%) for fertilizer-induced emission plus a background emission (B) rate of 1 kg N_2O-N ha^{-1} yr^{-1} (IPCC 1997). Most global extrapolations are based on the IPCC default. Nevertheless, cropping-specific and country-specific emission factors should be used where

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possible, in order to reflect the specific conditions of the country and the agricultural practices involved (IPCC 2000).

China is the most important rice producing country in the world. Rough rice production in China contributes ~30% to the world total (International Rice Research Institute (IRRI 2004)). Its planting area accounts for about 20% of the world total and 23% of all cultivated land in China (Frolking et al. 2002). Of which, ~93% is irrigated rice paddies; ~5% is distributed in rain fed lowlands and ~2% in uplands (IRRI 2004). Various water management regimes are currently practiced in China's rice paddies, such as seasonal continuous flooding (F), flooding-midseason drainage-frequent water logging with intermittent irrigation (F-D-F), and flooding-midseason drainage-reflooding-moist intermittent irrigation but without water logging (F-D-F-M) (Gao and Li 1992; Huang et al. 2004). An episode of mid-season drainage for 7–10 days rather than continuous flooding is commonly employed in China to inhibit ineffective tillers, remove toxic substances and improve roots activities.

Water regime often incurs a sensitive change in N_2O emission in rice paddies (Akiyama et al. 2005). It is well documented that mid-season drainage in rice paddies triggers substantial N_2O emission in contrast with continuous flooding (Cai et al. 1997; Zheng et al. 2000; Jiang et al. 2003). In addition, N_2O fluxes during intermittent irrigation periods depend strongly on whether or not water logging is present in paddy fields, which often begets a significant difference in seasonal total of N_2O emissions between the water regimes of F-D-F and F-D-F-M (Zou et al. 2005a).

Some studies have gone into quantifying fertilizer-induced N_2O emission and its background emission from rice paddies at the regional and global scales (Yan et al. 2003; Zheng et al. 2004; Akiyama et al. 2005). Yan et al. (2003) estimated N_2O emission factors and background emissions in irrigated paddy fields during the rice growing season, but they did not differentiate N_2O emissions under different water regimes. In contrast, Akiyama et al. (2005) recently reported that the EFs averaged 0.22% for the continuous flooding paddies and 0.37% for the fertilized paddies with mid-season drainage. In the data set employed to estimate N_2O emission factors by Akiyama et al. (2005), only 5 field studies were carried out in China (Cai et al. 1997; Chen et al. 1997; Hou et al. 2000; Zheng et al. 2000; Xiong et al. 2002). In addition, some N_2O measurements from rice paddies under the F-D-F-M water regime (Xing and Zhu 1997; Zheng et al. 2000) were treated as statistical outliers, and thus they were excluded by Akiyama et al. (2005) as well.

Consequently, the estimates of emission factor and background N_2O emission in previous studies may not truly reflect N_2O emission from rice paddies in China where various water regimes are practiced. Here, we compiled and statistically analyzed available field measurements of N_2O from rice paddies in China. County-level agricultural survey data for China in the 1980s and 1990s were acquired from the database of National Greenhouse Gases Inventories of Agriculture. The objective of this study was to quantify the cropping-specific direct emission factor and background emission for N_2O during the rice growing season, and thereafter, to estimate direct N_2O emissions from paddy rice production in mainland China in the 1980s–1990s.

9.2 Material and Methods

9.2.1 N_2O Data Collection

We compiled measurements of direct N_2O emission from rice field studies that were published to date in peer-reviewed Chinese and English journals. Data published in Chinese were gathered from the Chinese Journal Net (CJN) full-text database and those in English from the Science Citation Index (SCI) database. Measurements of N_2O flux taken from field studies over an abnormally shorter period (Xu et al. 1995), from pot experiments (Xiong et al. 2003a) or from incubation studies (Yan et al. 2000) were not considered in this study. We did not adopt an obviously abnormal result that N_2O fluxes were not detectable when pig manure was incorporated at the rate of 164 kg N ha^{-1} under the water regime of F-D-F (Yang et al. 1996). Measurements from aerobic rice fields (Xu et al. 2004), and from those in which the controlled release fertilizers were applied (Li et al. 2004a) or the amount of organic amendments was not presented for nitrogen treatments (Khalil et al. 1998) were excluded as well.

At the end, we employed 71 field measurements from 17 studies to estimate the emission factor and background emission of N_2O during the paddy rice growing season under different water regimes (Tables 9.1, 9.2 and 9.3). These studies were performed at 8 field sites that were located in Beijing, Guangdong, Henan, Jiangsu, Jiangxi, Liaoning, and Sichuan provinces, China over the period of 1992–

Table 9.1 Direct N_2O emissions from continuous flooding paddy fields during rice growing season

		Chemical fertilizer		Organic amendment			
Location	Year	Type[a]	Amount kg N ha^{-1}	Type[b]	Amount kg N ha^{-1}	N_2O emission kg N ha^{-1}	Reference
Shenyang,	1992	no	0	no	0	0	Chen et al. (1995)
41°32′N,	1992	U	374	no	0	0	Chen et al. (1995)
122°23′E	1996	U	374	FM	42	0.04	Hou et al. (2000)
Nanjing,	2000	CF + Urea	277	no	0	0.06	Zou et al. (2005a)
32°00′N,	2000	CF + Urea	277	WR	18	0.03	Zou et al. (2005a)
118°48′E							
Guangzhou,	1995	U	306	-	0	0.16	Lu et al. (1997)
23°26′N,							
113°30′E							

[a] Type of chemical fertilizer: CF, compound fertilizer; U, urea.

[b] Type of organic amendment: FM, farmyard manure; WR, winter wheat residue.

Table 9.2 Direct N_2O emissions from paddy fields under the water regime of flooding-midseason-reflooding (F-D-F) during rice growing season

		Chemical fertilizer		Organic amendment			
Location	Year	Type[a]	Amount kg N ha^{-1}	Type[b]	Amount kg N ha^{-1}	N_2O emission Kg N ha^{-1}	Reference
Nanjing, 32°00′N, 118°48′E	1994	No	0	No	0	0.14	Cai et al. (1997)
	1994	U	100	No	0	0.17	Cai et al. (1997)
	1994	AS	100	No	0	0.17	Cai et al. (1997)
	1994	U	300	No	0	0.62	Cai et al. (1997)
	1994	AS	300	No	0	0.98	Cai et al. (1997)
	2000	U	277	No	0	1.55	Zou et al. (2005a)
	2000	U	277	No	18	1.43	Zou et al. (2005a)
Jurong, 31°58′N, 119°10′E	1995	No	0	No	0	0.62	Cao et al. (1999)
	1995	U	100	No	0	0.86	Cao et al. (1999)
	1995	U	200	No	0	0.82	Cao et al. (1999)
	1995	U	200	No	0	0.74	Cao et al. (1999)
	1995	U	300	No	0	0.93	Cao et al. (1999)
Yingtan, 28°15′N, 116°55′E	2000	No	0	MV	124	0.26	Xiong et al. (2002, 2003b)
	2000	No	0	MV	124	0.30	Xiong et al. (2002, 2003b)
	2000	No	0	No	0	0.18	Xiong et al. (2002, 2003b)
	2000	No	0	No	0	0.23	Xiong et al. (2002, 2003b)
	2000	U	276	No	0	0.35	Xiong et al. (2002, 2003b)
	2000	U	276	No	0	0.28	Xiong et al. (2002, 2003b)
	2000	U	276	MV	0	0.34	Xiong et al. (2002, 2003b)
	2000	U	276	MV	124	2.81	Xiong et al. (2002, 2003b)
Guangzhou, 23°26′N, 113°30′E	1994	U	162	PM	64	0.49	Yang et al. (1997)
	1994	U	287	No	0	3.14	Yang et al. (1997)
	1995	U	140	MR	10	0.24	Yang et al. (1996)
	1995	U	140	PM	82	0.40	Yang et al. (1996)
	1995	U	280	No	0	3.14	Yang et al. (1996)
	1995	U	306	No	0	0.28	Lu et al. (1997)
	1995	U	306	No	0	1.32	Lu et al. (1997)

[a] Type of chemical fertilizer: AS, ammonium sulphate; U, urea.

[b] Type of organic amendment: MR, mushroom residue; MV, milk vetch residue; PM, pig manure.

2002. These provinces are distributed in the 5 crop-zone regions in mainland China (Fig. 9.1). For each field study, we documented the seasonal N_2O emission, the type and amount of organic amendment and fertilizer nitrogen application, the water management regime, the drainage duration, the field location and cropping season. If seasonal N_2O amounts were not originally presented, they were calculated from average fluxes of N_2O over the entire rice growing season.

Major types of chemical N fertilizer in agriculture in China, such as urea, compound fertilizer, ammonium sulphate, and ammonium bicarbonate, were included in this study (Tables 9.1, 9.2 and 9.3). For the fertilized fields, seasonal chemical N fertilizer was, on average, applied at the rate of 247 kg N ha^{-1}, ranging from 100 to 450 kg N ha^{-1} (Tables 9.1, 9.2 and 9.3). Besides, chemical nitrogen fertilizer application, organic manure or crop residue amendment was used as basal fertilizer in some studies. Organic amendments comprised farmyard manure and crop residue, such as winter wheat, milk vetch, rice straw, mushroom and rapeseed cake. Due

Table 9.3 Direct N_2O emissions from paddy fields under the water regime of flooding-midseason-reflooding-moist intermittent irrigation but without water logging (F-D-F-M) during rice growing season

		Chemical fertilizer		Organic amendment			
Location	Year	Type[a]	Amount kg N ha^{-1}	Type[b]	Amount kg N ha^{-1}	N_2O emission Kg N ha^{-1}	Reference
Beijing, 40°30′N, 116°24′E	1992	AB	125	No	0	1.32	Khalil et al. (1998)
	1993	AB	125	No	0	1.21	Khalil et al. (1998)
Fengqiu, 35°04′N, 113°10′E	1994	AB + U	364.5	PM	67	4.42	Cai et al. (1999)
	1994	AB + U	364.5	PM	67	2.01	Cai et al. (1999)
	1994	AB + U	364.5	PM	67	1.71	Cai et al. (1999)
Nanjing, 32°00′N, 118°48′E	2001	CF + AB	333	CM	29	3.26	Zou et al. (2004)
	2001	CF + AB	333	PM	50	3.38	Zou et al. (2004)
	2001	CF + AB	333	No	0	4.11	Zou et al. (2005a)
	2001	CF + AB	333	RC	146	4.83	Zou et al. (2005a)
	2001	CF + AB	333	WR	18	3.33	Zou et al. (2005a)
	2002	No	0	No	0	1.38	Zou et al. (2005b)
	2002	U	150	No	0	2.67	Zou et al. (2005b)
	2002	U	150	WR	36	2.97	Zou et al. (2005b)
	2002	U	225	WR	18	3.79	Zou et al. (2005b)
	2002	U	300	No	0	4.44	Zou et al. (2005b)
	2002	U	450	No	0	6.17	Zou et al. (2005b)
Wuxi, 31°37′N, 120°28′E	2001	U	150	No	0	1.50	Zheng et al. (2004)
	2001	U	250	No	0	2.31	Zheng et al. (2004)
	2001	U	250	No	0	1.21	Zheng et al. (2004)
	2002	U	0	No	0	0.90	Zheng et al. (2004)
	2002	U	150	No	0	1.71	Zheng et al. (2004)
	2002	U	250	No	0	1.99	Zheng et al. (2004)
	2002	U	250	No	0	2.99	Zheng et al. (2004)
Shuzhou, 31°16′N, 120°38′E	1993	No	0	No	0	0.86	Xing and Zhu (1997)
	1993	U	210	No	0	2.57	Xing and Zhu (1997)
	1993	AS	220	No	0	3.27	Xing and Zhu (1997)

Table 9.3 (continued)

Location	Year	Chemical fertilizer Type[a]	Chemical fertilizer Amount kg N ha^{-1}	Organic amendment Type[b]	Organic amendment Amount kg N ha^{-1}	N_2O emission Kg N ha^{-1}	Reference
	1993	U	210	PM	68	3.01	Xing and Zhu (1997)
	1993	U	310	No	0	2.82	Xing and Zhu (1997)
	1994	No	0	NoNo	0	0.46	Zheng et al. (2000)
	1994	AB	191	No	0	1.24	Zheng et al. (2000)
	1994	AB	191	No	0	1.72	Zheng et al. (2000)
	1994	AB	191	No	0	1.52	Zheng et al. (2000)
	1996	No	0	No	0	0.50	Zheng et al. (2000)
	1996	U	161	RS	30	1.01	Zheng et al. (2000)
	1996	AB	191	No	0	3.45	Zheng et al. (2000)
	1996	U	191	No	0	1.92	Zheng et al. (2000)
Guangzhou, 23°26′N, 113°30′E	2002	No	0	No	0	0.93	Li et al. (2004a)
	2002	U	180	No	0	2.45	Li et al. (2004a)

[a] Type of chemical fertilizer: AB, ammonium bicarbonate; AS, ammonium sulphate; CF, compound fertilizer; U, urea.

[b] Type of organic amendment: CM, cow manure; PM, pig manure; RC, rapeseed cake; RS, rice straw; WR, winter wheat residue.

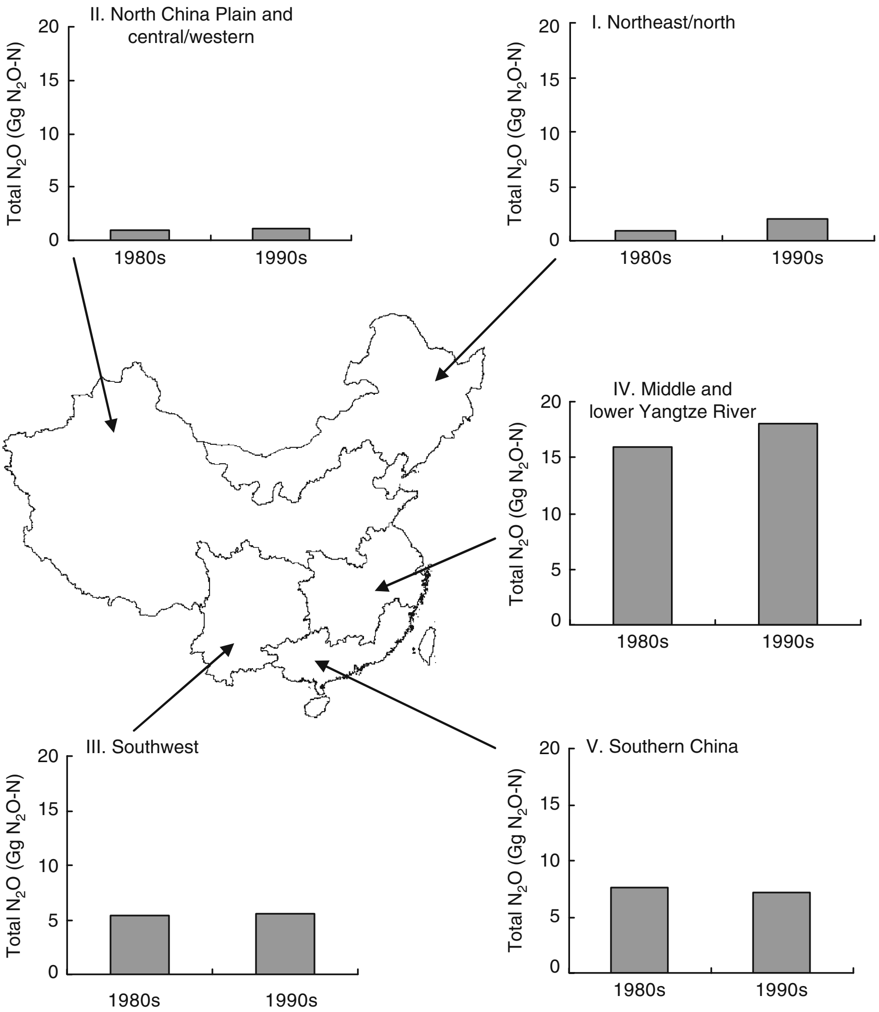

Fig. 9.1 Spatial distribution of seasonal total N_2O emissions from paddy fields during the rice growing season in mainland China in 1980s and 1990s. The rice paddies assigned to the 5 crop-zone regions: I, North and Northeast, covering Heilongjiang and Inner Mongolia, Liaoning and Jilin provinces; II, North China Plain and Central and west, covering Henan, Hebei, Tianjin, Beijing, Ningxia, Shaanxi, Shanxi, Shandong, Xingjiang and Gansu provinces; III, Southwest, covering Guizhou, Sichuan, Chongqing and Yunnan provinces; IV, Middle and lower Yangtze River, covering Anhui, Hubei, Hunan, Jiangsu, Jiangxi, Shanghai and Zhejiang provinces; V, Southern China, covering Fujian, Guangdong, Guangxi and Hainan provinces

to lack of available data, we could not distinguish between organic- and inorganic nitrogen fertilizer-induced N_2O emissions. Thus, seasonal total nitrogen input was used to quantify the fertilizer-induced emission factor of N_2O in this study.

In all employed measurements, N_2O fluxes were measured in situ by static chamber-gas chromatograph method. They were measured by manual sampling system, except that an automatic system was used to continuously measure N_2O fluxes in one study in Shuzhou (Zheng et al. 2000). In general, gas samples were taken twice weekly (Cai et al. 1997; Cao et al. 1999; Chen et al. 1999; Zou et al. 2005b) or once a week (Yang et al. 1996; Cai et al. 1999; Xiong et al. 2002) over the entire rice growing season, except for once a day during the mid-season drainage (Xiong et al. 2003b; Zou et al. 2005a).

9.2.2 Rice Production Data Collection

County-level agricultural survey data for China in the 1980s and 1990s were acquired from the database of National Greenhouse Gases Inventories of Agriculture. The database contains county statistics on paddy rice planting area and yield, water regime, percentage of crop residue retained, manure and synthetic nitrogen fertilizer use during the rice growing season for counties in mainland China. No data were available for Taiwan, Hong Kong or Macao. The paddy rice yield reported by IRRI was used to check the quality of the data set since only paddy rice yield can be obtained at the IRRI dataset and no nitrogen input data were available at national or international statistics dataset. Provincial rice yield derived from this surveyed (Y_S) county-scale data set is comparable to that officially reported (Y_I) by the IRRI ($Y_S = 1.046Y_I, r^2 = 0.635$, $P < 0.0001$). This suggests that water regime and nitrogen input information in the data set could also have represented the actual status of paddy rice production in mainland China. In order to estimate the nitrogen input of crop residue and organic manure, we assumed that harvest biomass of crop residue is $4.5\,\mathrm{t\,ha^{-1}}$, and the nitrogen content of crop residue and organic manure was derived from the IPCC guidelines, with an average of 0.45 and 0.71%, respectively (IPCC 2000).

9.2.3 Statistical Analysis

The relationship between N application and N_2O emission established by Eichner (1990) and Bouwman (1996) motivated the concept of fertilizer-induced emission factor (EF). EF is defined as N_2O emission from nitrogen fertilizer plots minus the emission from unfertilized control plots (all other conditions being equal to those of the fertilized plots) expressed as a percentage of N applied. This IPCC methodology suggests that linear regression models could be used to quantify N_2O emissions. In the present study, we used a linear regression model ($N_2O\text{-}N = EF{\cdot}N + B + \varepsilon$) with the character of Ordinary Least Squares (OLS) to fit N_2O emissions (N_2O-N) by nitrogen inputs (N). In this model, ε denotes the error term; EF and B are the simulated parameters that represent N_2O emission factor and background emission, respectively. A t-test was used to examine statistic significance of the parameter estimates. An analysis of variance (ANOVA) F-test partitioned the total variation of N_2O-N into the linear relationship with N and the part not explained by the relationship. The model fitness to the data was examined by both residual distribution pattern and power analysis. We also used a one-way ANOVA to test whether seasonal N_2O amount depended on the water regime. The statistic analyses were conducted using JMP IN 5.1 (SAS INC. 2003).

9.2.4 Uncertainties in N_2O Estimate

National inventories of N_2O from soils would typically contain a wide range of emission estimates. Total uncertainty in emissions inventory is generally combined by uncertainties in emission factors and activity data. Similar to uncertainty estimate in the IPCC methodology (IPCC 2000), we use the error propagation equation to calculate the uncertainties in seasonal N_2O emissions from rice paddies under each water regime as follow:

$$U_C = \sqrt{U_A^2 + U_E^2} \tag{9.1}$$

Where: U_C is the combined uncertainty expressed as a percentage for each water regime; U_A and U_E are the percentage uncertainties for the activity data and emission factor, respectively. In this study, the activity data is the input data of the estimate method and U_A is principally determined by the reliability of nitrogen input data. The confidence interval of parameter estimates in the simulated model was used to calculate U_E. A confidence interval of 95% that is suggested by the IPCC guidelines represents a 95% probability of containing the unknown true value. U_E was expressed as half the 95% confidence interval divided by the mean.

Eventually, the total uncertainty in seasonal N_2O estimates during the rice growing season for each decade was calculated by the Eq. (9.2) (IPCC 2000):

$$U_{total} = \frac{\sqrt{(U_F \cdot x_F)^2 + (U_{F-D-F} \cdot x_{F-D-F})^2 + (U_{F-D-F-M} \cdot x_{F-D-F-M})^2}}{x_F + x_{F-D-F} + x_{F-D-F-M}} \tag{9.2}$$

Where: U_{total} is the total uncertainty expressed as a percentage for each decade; x_i and U_i (i represents the water regimes of F, F-D-F, and F-D-F-M) are the uncertain quantities and the percentage uncertainties (i.e. U_C in the Eq. (9.1)) associated with them under different water regimes, respectively.

9.3 Results

9.3.1 Interannual and Spatial Variations of N_2O Emissions

It is apparent that N_2O emissions varied interannually and spatially (Tables 9.2 and 9.3). Under an identical water regime of F-D-F, seasonal total N_2O in 1994 averaged 1.49 kg N_2O-N ha^{-1} when ammonium bicarbonate was applied at the rate of 191 kg N ha^{-1}, which was 57% lower than that in 1996 in Shuzhou (Table 9.2). In Wuxi, seasonal N_2O emission from plots with urea applied at the rate of 250 kg N ha^{-1}, on average, amounted to 2.49 kg N_2O-N ha^{-1} in the 2002 season, which was increased by 41% as compared to that in the 2001 season. In 2002, on the other hand, seasonal total of N_2O emissions from plots with urea applied at the rate of 150 kg N ha^{-1} was 2.67 kg N_2O-N ha^{-1} in Nanjing, while that from plots with the identical fertilizer application and water regime was only 1.71 kg N_2O-N ha^{-1} in Wuxi (Table 9.3). Although it varied interannually and spatially, seasonal N_2O emissions generally increased with nitrogen input under the water regime of F-D-F or the F-D-F-M (Fig. 9.1b, c).

9.3.2 Modeling Emission Factor and Background Emission for N_2O

A one-way ANOVA indicated that seasonal total N_2O emission from rice paddies significantly varied with water regime ($F_{2,70} = 21.7$, $P < 0.0001$), which suggests that the effect of water regime on seasonal pattern of N_2O fluxes has incurred a pronounced difference in seasonal N_2O amount. To accurately quantify seasonal N_2O emissions and minimize its uncertainties, the data set of N_2O was classified into three categories based on water regime (F, F-D-F, and F-D-F-M, Tables 9.1, 9.2 and 9.3) and seasonal N_2O emissions were separately modeled under different water regimes in this study.

Probably due to limited available data and low N_2O emissions, no pronounced relationship between N_2O emission and nitrogen input was found in the continuous flooding rice paddy fields (Fig. 9.2a, $F_{1,5} = 0.25$, $P = 0.65$). Mean and stand error of seasonal N_2O amounts were estimated to be 0.048 and 0.024 kg N_2O-N ha^{-1}, and those of N inputs to be 278 and 60 kg N ha^{-1} for the 6 field measurements, respectively (Table 9.1). Seasonal total of N_2O was, on average, equivalent to 0.02% of the N fertilizer applied under continuous flooding.

In contrast, a significant linear relationship between seasonal N_2O amount and N input in rice paddies was found under the water regime of F-D-F (Table 9.4, Fig. 9.2b). Based on simulated parameters of the model F-D-F-1 (Table 9.5), the fertilizer-induced emission factor for N_2O during the rice growing season averaged 0.42%, with a stand error of 0.13%. Seasonal background N_2O emission was estimated to be 0.009 kg N_2O-N ha^{-1}, with a high stand error (0.30 kg N_2O-N ha^{-1}), suggesting its large uncertainty. A t-test showed that this estimated value did not

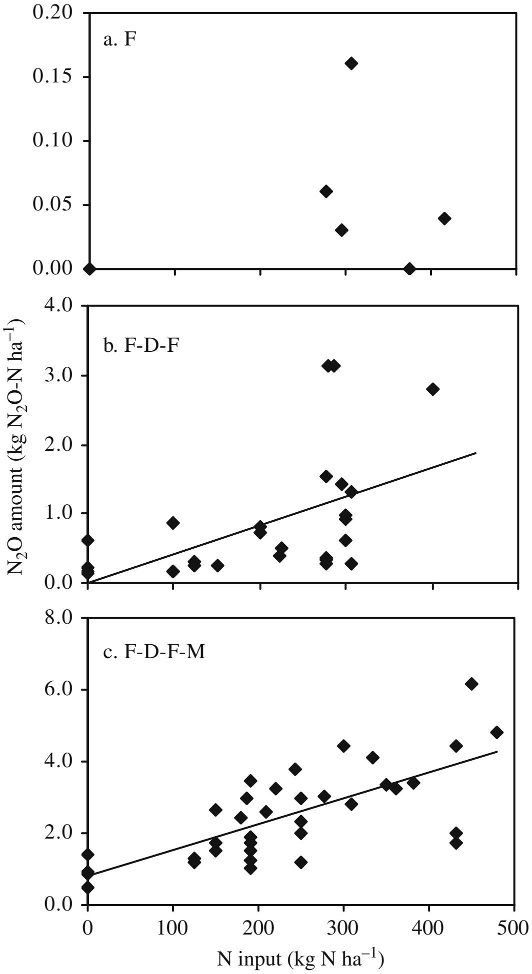

Fig. 9.2 Dependence of seasonal N_2O amount on nitrogen input in rice paddies under different water regimes. (**a**) Continuous flooding (F); (**b**) flooding-mid-season drainage-reflooding (F-D-F); and (**c**) flooding-midseason drainage-reflooding-moisture intermittent irrigation but without water logging (F-D-F-M)

Table 9.4 ANOVA F-tests for the simulated OLS models in rice paddies with mid-season drainage

Model	Term	DF	SS	*F*	*P*	Power	r^2
F-D-F-1	Regression	1	5.9	10.4	0.004	0.87	0.293
	Error	25	14.4	–	–	–	–
F-D-F-2	Regression	1	25.2	45.6	<0.0001	1.00	0.293
	Error	26	14.4	–	–	–	–
F-D-F-M	Regression	1	36.2	45.3	<0.0001	1.00	0.557
	Error	36	28.8	–	–	–	–
MSD-Akiyama[a]	Regression	1	13.1	18.8	<0.0001	0.99	0.28
	Error	44	30.5	–	–	–	–

[a] The MSD-Akiyama model simulated using the data that were shown in Table 9.2 in Akiyama et al.'s (2005) study.

Table 9.5 T-tests for parameter estimates in the simulated models in rice paddies with mid-season drainage

	EF				B			
Model	Estimate	Stand error	t-ratio	*P*	Estimate	Stand error	t-ratio	*P*
F-D-F-1	0.0042	0.0013	3.22	0.004	0.0087	0.3	0.03	0.98
F-D-F-2	0.0042	0.0006	6.75	<0.0001				
F-D-F-M	0.0073	0.0011	6.73	<0.0001	0.79	0.28	2.82	0.008
MSD-Akiyama[a]	0.0043	0.001	4.34	<0.0001	0.20	0.19	1.06	0.30

[a] The MSD-Akiyama model simulated using the data that were shown in Table 9.2 in Akiyama et al.'s (2005) study.

significantly differ with the assumed value of zero (term "B" for the model F-D-F-1, Table 9.5). Subsequently, we used a reduced model: $N_2O\text{-}N = EF \cdot N + \varepsilon$, in which the background emission (B) was omitted (model F-D-F-2, Table 9.4). The reduced model F-D-F-2 suggested that N_2O emission factor averaged 0.42%, with a stand error of 0.06% (Table 9.5). Compared with the model F-D-F-1, the model F-D-F-2 minimized substantially the uncertainty in estimates of direct N_2O emission.

Similarly, direct N_2O emissions depended greatly on nitrogen inputs in rice paddies under the water regime of F-D-F-M (model F-D-F-M, Table 9.4; Fig. 9.3c).

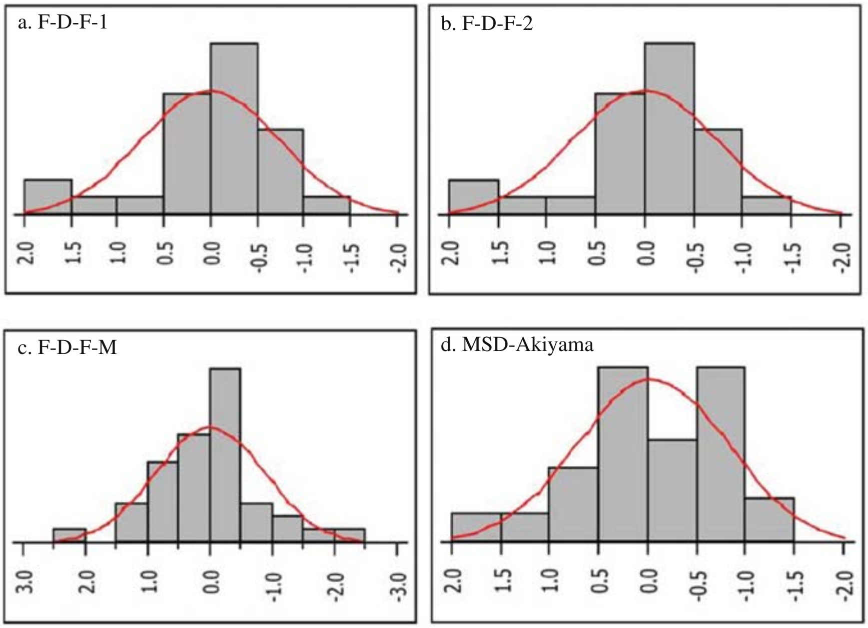

Fig. 9.3 Distribution pattern of residuals for the simulated Ordinary Least Square linear regression models. (**a**) Model F-D-F-1; (**b**) model F-D-F-2; (**c**) model F-D-F-M; and (**d**) model MSD-Akiyama. The MSD-Akiyama model simulated using the data that were shown in Table 9.2 in Akiyama et al.'s (2005) study. *Fit line* represents the normal distribution pattern

The model F-D-F-M yielded that N_2O emission factor and background emission averaged 0.73% and 0.79 kg N_2O-N ha^{-1}, respectively (Table 9.5). Stand errors of EF and background emission were estimated to be 0.11 and 0.28%, respectively. Results of t-tests suggested that both nitrogen application and background emission contributed significantly to seasonal N_2O total in rice paddies under the F-D-F-M water regime (Table 9.5).

9.3.3 Quantifying Seasonal Direct N_2O Emissions in 1980s and 1990s

Rice planted area in mainland China totals about 32.9 million hectare in the 1980s, which is 19% greater than in the 1990s (Table 9.6). The decrease in area mainly occurred in the IV (Middle and lower Yangtze River) and V (Southern China) crop-zone regions. In the 1980s, 18% of rice paddy is continuous flooding, 77% under the water regime of F-D-F, and 5% under the F-D-F-M, while their respective percentages have changed to be 12, 75 and 13%, respectively. The percentage of various water regime derived from the surveyed data set is generally consistent with previous estimates. Xing (1998) estimated that continuous flooding rice paddies account for 10% of the total in 1995. Li et al. (2002) reported that rice paddies with mid-season drainage contribute ~80% to the total in China in 2000. Yan et al. (2003) estimated that two-thirds of rice paddy is under intermittent irrigation or middle season drainage in China.

Chemical fertilizer application rate was, on average, estimated to be 168.21 kg N ha^{-1} during the rice growing season in the 1980s, which is 16% lower than in the 1990s (Table 9.6). Seasonal chemical N input was highest in the II and III crop-zone regions. In contrast, manure and crop residue nitrogen inputs were generally lower in the 1990s than in the 1980s. Seasonal organic manure nitrogen input rate, with an average of 40.65 kg N ha^{-1}, in the 1980s was almost twice as that in the 1990s. About 31% and 24% of the total above-ground crop biomass was retained in paddy fields in the 1980s and 1990s, respectively. Thereby, crop residue N retained during the rice growing season was, on average, estimated to be 6.31 kg N ha^{-1} in the 1980s and 4.85 kg N ha^{-1} in the 1990s for the 5 crop-zone regions in mainland China (Table 9.6).

The emission factors and background emissions simulated under different water regimes in the present study are assumed to be applicable for N_2O emissions from rice paddies in mainland China in the 1980s and 1990s. Thus, N_2O emissions during the rice growing season averaged 28.68 Gg N_2O-N in the 1980s and 32.26 Gg N_2O-N in the 1990s. Seasonal N_2O fluxes were estimated to be, on average, 0.88 kg N_2O-N ha^{-1} in the 1980s and 1.00 kg N_2O-N ha^{-1} in the 1990s, which was equivalent to 0.41–0.46% of the seasonal total N input (Table 9.6). Substantial N_2O emission occurred in the region of Middle and lower Yangtze River, contributing 52–53% to the national total N_2O emission during the rice growing season (Fig. 9.3).

Table 9.6 Rice planted area, yield, organic and chemical nitrogen inputs (Mean ± SE) and seasonal direct N_2O emissions during paddy rice growing season in 1980s and 1990s

Region	Area (10^6 ha)	Yield (t ha^{-1})	Crop residue (kg N ha^{-1})	Manure Gg N_2O-N	Chemical fertilizer (%)	Total N	N_2O flux	Total N_2O	Percent of N applied
1990s									
I	1.16	4.95±0.15	3.45±1.29	20.78±9.50	157.85±29.57	182.08±37.50	0.81	0.94	0.45
II	0.91	5.00±0.40	2.81±0.65	49.06±1.76	179.96±17.42	231.83±21.02	1.03	0.94	0.45
III	4.93	5.26±0.19	4.24±0.75	59.36±4.71	178.38±16.40	241.98±23.09	1.03	5.09	0.43
IV	17.62	4.28±0.19	7.39±0.57	43.96±4.36	167.70±6.00	219.05±7.04	0.83	14.57	0.38
V	8.29	5.64±0.36	8.63 ± 1.05	27.75±4.36	160.53±11.06	196.91±13.55	0.86	7.14	0.44
Total	**32.90**	**5.03±0.15**	**6.31±0.50**	**40.65±3.61**	**168.21± 5.32**	**215.16±6.98**	**0.88**	**28.68**	**0.41**
1990s									
I	1.96	6.77±0.64	3.87±1.52	14.81±4.29	194.25±33.09	212.94±32.82	0.92	1.81	0.47
II	1.11	6.79±0.48	2.20±0.84	18.23±7.46	227.00±20.41	247.43±24.90	1.08	1.19	0.40
III	5.28	5.76±0.28	3.32±0.86	21.10±6.84	206.13±21.08	230.55±19.97	1.03	5.46	0.43
IV	16.50	5.38±0.23	5.33±0.69	26.27±2.25	197.84±10.59	229.44±10.94	1.05	17.31	0.44
V	7.04	6.19±0.17	7.41±0.97	16.06±3.46	176.27±11.99	199.74±14.52	0.92	6.50	0.50
Total	**31.89**	**6.18±0.16**	**4.85±0.47**	**20.99±2.02**	**198.80±7.23**	**224.64±7.73**	**1.00**	**32.26**	**0.46**

9.3.4 Uncertainties in Seasonal N_2O Estimate

According to the methodology recommended by the IPCC (2000) for quantifying uncertainties (Eqs. (9.1) and (9.2)), the uncertainties in emission factor were estimated to be 28.0 and 29.5% for the models F-D-F and F-D-F-M, respectively. The uncertainty in seasonal total nitrogen input was estimated to be 22.2 and 24.3% in the 1980s and 1990s, respectively. Eventually, their combined total uncertainties in seasonal N_2O estimate were estimated to be 36.2% in the 1980s and 37.5% in the 1990s.

9.4 Discussion

9.4.1 Fitness of OLS Model

The assumptions of the OLS linear regression model strictly concern the error term (ε) that can be represented by the pattern of residuals. The residuals from the fitted model are important for checking whether the assumptions of linear regression analysis are met (Quinn and Keough 2004). The residuals of the OLS models tended to be symmetrically distributed and centered on zero, suggesting that these models were well fit for the data in rice paddies under the F-D-F and the F-D-F-M (Fig. 9.3a–c). Power analyses for the model F-D-F-2 and the model F-D-F-M also showed that these linear relationships were strong enough to model the N_2O data (Table 9.4). On the other hand, a stronger power for the model F-D-F-2 relative to the model F-D-F-1 suggested that it was better suited for the data, although both models had the similar determination coefficient (r^2) (Table 9.4). However, great cares should be taken in using the r^2 values for comparing the fitness of different models, such as the model F-D-F-1 and the reduced model F-D-F-2 in this study, since it is inappropriate for comparing models with different numbers of parameters (Scott and Wild 1991).

9.4.2 Emission Factor of Nitrogen for N_2O

The fertilizer-induced emission factor of N_2O in the present study was, on average, 0.42% for the water regime of F-D-F and 0.73% for the F-D-F-M. Obviously, these estimated emission factors of N_2O in rice paddies are significantly lower than the IPCC (1997) default factor of 1.25% or estimates in upland croplands in this area (Zheng et al. 2004). Yan et al. (2003) estimated that N_2O emission factors and background emissions averaged 0.25% and 0.26 kg N_2O-N ha^{-1} in the rice growing season, respectively. However, they did not distinguish N_2O emissions under different water regimes in rice paddies. In contrast, Akiyama et al. (2005) recently reported that the EFs averaged 0.22% for the continuous flooding rice paddies and 0.37% for the fertilized paddies with mid-season drainage. These estimates represent the mean of 16 and 23 emission factors directly measured from field studies in which both nitrogen and no-nitrogen treatments were designed, respectively. As the authors pointed out, seasonal total N_2O emissions were not significantly related to nitrogen input during the rice growing season over all water regimes or for continuous flooding (Akiyama et al. 2005). However, based on the data that were exhibited in Table 9.2 by Akiyama et al. (2005), a pronounced relationship between N_2O emission and N input during the rice growing season was found in the rice paddies with mid-season drainage. Using Akiyama et al. (2005) identical data (excluding measurements from nitrification inhibitors and controlled released fertilizers), the simulated OLS linear model MSD-Akiyama predicted that N_2O emission factor averaged 0.43%, with a background emission of 0.20 kg N_2O-N ha^{-1} (Table 9.5). This emission factor simulated by the OLS model is slightly higher than that obtained by Akiyama et al. (2005), which was based on the Maximum Likelihood (ML) estimate.

Indeed, the emission factor of N_2O estimated by Akiyama et al. (2005) refers to the value of EF that maximizes the likelihood of observing N_2O emission in measurements. Point and interval estimation using the ML model relies heavily on distributional assumptions that a sample of observations (response variables) has a normal distribution (Quinn and Keough 2004). However, the direct EF data in Akiyama et al. (2005) study had a log-normal distribution pattern. In contrast, EF estimated by the OLS model in this study represents the value with the least uncertainty. The OLS point estimates require no distributional assumptions for variables, but instead concentrate on residual distribution (Quinn and Keough 2004). As shown in Fig. 9.3d, the residuals of the model MSD-Akiyama were close to being normally distributed. A power analysis also showed that it was strong enough to model the data (Table 9.4). In order to minimize the uncertainty in estimates of emission factor for N_2O, presumably, the OLS model could be more appropriate for N_2O data than the ML model used by Akiyama et al. (2005).

9.4.3 Background N_2O Emission

In the present study, background emission of N_2O in rice paddies was pronounced only under the water regime of F-D-F-M (Table 9.5). Background N_2O emissions during the rice growing season were negligible for the continuous flooding paddy fields. Under the water regime of F-D-F, background N_2O emission was estimated to be 0.009 kg N_2O-N ha^{-1}, which was not significantly different from “0” (Table 9.5). This negligible background emission is partially due to water management that water logging dominated over the rice season except for a short-term episode of mid-season drainage. An intensive N_2O emission occurred only in the course of mid-season drainage under the F-D-F. In contrast, background N_2O emission was, on average, as high as 0.79 kg N_2O-N ha^{-1} under the F-D-F-M water regime. Based on seven measurements in the continuous flooding and intermittent irrigation rice paddies, Yan et al. (2003) estimated background N_2O emission, on average, to be

0.26 kg N_2O-N ha^{-1} in the paddy rice season. However, as the authors acknowledged, there was much uncertainty with respect to the background emission estimate in paddy fields. Indeed, the background emission has become one of the most sensitive factors for developing an inventory of agricultural N_2O emissions (Bouwman et al. 2002; Yan et al. 2003; Akiyama et al. 2005).

9.4.4 Effect of Water Regime on N_2O Emissions

Different water regimes caused a sensitive change in N_2O emission in rice paddies. Under continuous flooding, N_2O emissions were generally pronounced only when fields were drained before rice harvesting (Chen et al. 1995; Lu et al. 1997). In contrast to continuous flooding, mid-season drainage triggered substantial N_2O emission from rice paddies under the F-D-F water regime (Xiong et al. 2002; Jiang et al. 2003; Zou et al. 2005a). Based on the results of this study, we predict that seasonal N_2O emissions will amount to 0.03 kg N_2O-N ha^{-1} when nitrogen is applied at the rate of 150 kg N ha^{-1} in the continuous flooding rice paddies, which is similar to the results of earlier studies at other regions (Smith et al. 1982; Granli and Bøckman 1994). However, seasonal total N_2O will be, on average, up to 1.87 kg N_2O-N ha^{-1} under the water regime of F-D-F-M, which is threefold as much as that under the water regime of F-D-F.

Primarily, N_2O is produced in soils via the biogeochemical processes of nitrification and denitrification that are greatly influenced by soil water status. In contrast to paddies with the water regime of F-D-F, or the seasonal continuous flooding paddies, the dry-wet alteration after mid-term drainage created a favorable soil environment to both nitrification and denitrification processes, which contributed greatly to higher N_2O emissions under the water regime of F-D-F-M. Under continuous flooding, a large proportion of N_2O produced from denitrification would be further reduced to N_2 before leaving the soil (Firestone and Davidson 1989). On the other hand, water regime might influence the availability of nitrogen, labile C compounds, and O_2 in paddy soils that are key factors to N_2O production in general denitrification models (Firestone and Davidson 1989). The mid-season drainage and dry-wet alteration are able to improve root activities and accelerate soil organic C decomposition, which might produce more available C and N for soil microbes, and thereby favor N_2O emissions.

Water management pattern in rice production has been greatly changed in mainland China. Since the 1980s, mid-season drainage has been commonly adopted to increase rice productivity. Due to water resources scarcity and cultivation technique development, the water regime of F-D-F-M as a water-saving irrigation technology has been increasingly practiced in China's rice production. For example, water is especially scarce in the North China Plain that contains 26% of the China's cultivated land, 30% of its irrigated land, and 24% of its total grain production (Geng et al. 2001). The water regime of F-D-F-M and aerobic rice paddies, instead of anaerobic paddies, have been suggested as potential options for rice production in this area. However, a process-based model estimated that shifting water management from continuous flooding to mid-season drainage increased N_2O emissions from Chinese rice paddies by 0.13–0.20 Tg N_2O-N yr^{-1} (Li et al. 2004b) or 0.15 Tg N_2O-N yr^{-1} (Li et al. 2005). In addition, N_2O emissions have been shown to be extremely higher from aerobic rice paddies compared to anaerobic paddies (Xu et al. 2004). Therefore, these options would increase N_2O emissions from rice production in China. Indeed, how to reconcile increasing N_2O emissions and scarcity of water resources with the development of rice production, has become a real challenge in mainland China.

9.4.5 Contribution of Rice Production to Total N_2O Emission from Croplands

Using a precipitation-rectified emission factor model and the IPCC uncertainty estimate methodology, we estimated EF of N_2O in China's uplands to be 1.14% in 1997 with an uncertainty of 29% (Lu et al. 2006). Xing (1998) reported that direct N_2O emissions from paddy fields totaled 88 Gg N_2O-N, consisting of 35 Gg N_2O-N emitted during the rice growing season and 53 Gg N_2O-N during the upland crop seasons. Direct N_2O emission from croplands in China was estimated to be 275 Gg N_2O-N yr^{-1} in the 1990s by Zheng et al. (2004), or 398 Gg N_2O-N in 1995 by Xing (1998). These estimates suggest that rice production occurring on 23% of the cultivated land accounts for 7–11% of the total N_2O emission from croplands in China. Therefore, due to rice planting area increase in the past decades and lower emission factor, paddy rice relative to upland crop production could have greatly contributed to mitigating N_2O emissions from agriculture in China.

9.4.6 Uncertainties in Quantifying Direct N_2O Emission

In the present study, we did not find a significant relationship between N_2O emission and nitrogen input in the continuous flooding rice paddy fields. Besides, the scanty measurements and low N_2O emission may be another important cause. In contrast to continuous flooding, N_2O emissions were significantly higher in paddy fields with mid-season drainage and thereby relationship between N_2O emission and nitrogen input became pronounced. Under the water regime of F-D-F, fertilizer input and background emission in the simulated regression model can only explain 29% of the variability in the 27 observed seasonal average N_2O fluxes. Under the water regime of F-D-F-M, however, up to 56% of the variability in the 38 observed N_2O measurements can be explained by fertilizer together with background emission in the simulated regression model (Table 9.4).

Obviously, some factors other than water regime may also be important to N_2O emission factor in rice paddy fields. Besides fertilizer amount, fertilizer type has been recognized as another factor influencing N_2O emissions in agricultural fields

(Bouwman et al. 2002). Although seasonal N_2O emissions generally increased with fertilizer input, it varied with the type of fertilizer as well in rice paddy fields. Compared with urea, application of ammonium sulphate or ammonium bicarbonate induced higher N_2O emission under an identical water regime of F-D-F or F-D-F-M (Cai et al. 1997; Zheng et al. 2000). In contrast to pure chemical fertilizer application, organic manure and crop residue amendments increased seasonal N_2O emissions in some studies (Zheng et al. 2000; Zou et al. 2004, 2005b), while they decreased N_2O emissions in other studies (Xiong et al. 2003b).

Difference in frequency of N_2O measurements may also contribute to its estimate for uncertainty. Ideally, N_2O emissions should be measured frequently enough to capture its peak fluxes. Sharp peaks of N_2O fluxes in paddy fields were observed in a study using an automated monitor system (Zheng et al. 2000). Relative to measurements once a week, measurements twice weekly showed more peak fluxes of N_2O, particularly after nitrogen fertilizer was applied in rice paddies (Zou et al. 2005a). However, most studies in the data set measured N_2O flux only once a week. As a consequence, some flux peaks might have been missed and seasonal N_2O emissions could have been underestimated in these studies.

It is note worthy that this study only estimated direct N_2O emissions during the rice growing season, but did not count those during the following non-rice seasons in paddy fields. Although water regime has distinguished N_2O emissions in rice paddies from upland crops, some agricultural practices, such as water management and organic incorporation during the rice growing season, may have a substantial effect on the following seasonal N_2O emissions (Zou et al. 2005b). Results of our previous study in a paddy rice-winter wheat rotation system indicated that compared with the water regime of F-D-F, continuous flooding in the rice season significantly increased N_2O emissions from the winter wheat growing season. As well, wheat residue incorporation before rice transplanting had a far-lasting effect on N_2O emissions during the winter wheat growing season (Zou et al. 2003). Therefore, annual total of N_2O emissions in rice paddies would be underestimated by extrapolating N_2O data during the rice growing season.

9.5 Conclusions

During rice growing season, N_2O emissions depended significantly on water regime in paddy fields. Seasonal total N_2O was, on average, equivalent to 0.02% of the nitrogen input in the continuous flooding rice paddies. The emission factor of fertilizer for N_2O averaged 0.42% and 0.73% under the F-D-F and the F-D-F-M water regimes, respectively. N_2O background emission during the rice growing season was not pronounced under the water regime of F-D-F, but it amounted to 0.79 kg N_2O-N ha^{-1} under the F-D-F-M water regime. Seasonal N_2O emissions amounted to 32.26 Gg N_2O-N in the 1990s and 28.68 Gg N_2O-N in the 1980s, accounting for 7–11% increase of the reported estimates of annual total emission from croplands in mainland China. Relative to upland crop production, paddy rice development in the past decades could have greatly contributed to mitigating N_2O emissions from agriculture in China.

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Chapter 10
Impacts of Ground-Level Ozone on Crop Production in a Changing Climate

K. Vandermeiren, H. Harmens, G. Mills and L. De Temmerman

10.1 Introduction

Ozone (O_3) is a naturally occurring chemical present in both the stratosphere (the 'ozone layer', 10–40 km above the earth) and in the troposphere (0–10 km above the earth). While stratospheric O_3 protects the Earth's surface from solar UV radiation, tropospheric O_3 is the third most important greenhouse gas (after CO_2 and CH_4) (Denman et al. 2007; Solomon et al. 2007). It contributes to greenhouse radiative forcing, causing a change in the balance between incoming solar radiation and outgoing infrared radiation within the atmosphere that controls the Earth's surface temperature. Besides its role as a direct greenhouse gas, O_3 has been identified as one of the major phytotoxic air pollutants. The adverse effects of O_3 on plants were first identified in the 1950s (Hill et al. 1961), and it is now recognized as the most important rural air pollutant, affecting human health and materials, as well as vegetation (WGE 2004).

Comparisons of the mean global tropospheric O_3 concentrations with those measured over a century ago indicate that current levels have increased by approximately two times due to enhanced emissions associated with fossil fuel and biomass burning (Gauss et al. 2006; Denman et al. 2007). Long-distance and even intercontinental transport has resulted in a steady increase in O_3 concentration in rural areas hundreds and thousands of kilometers from the original sources of pollution (Prather et al. 2003). Nearly one-quarter of the Earth's surface is currently at risk from the mean tropospheric O_3 in excess of $60\,\mathrm{nl\,l^{-1}}$ during mid-summer with even greater local concentrations occurring (Fowler et al. 1999a, b). This is well above the mean concentration of $40\,\mathrm{nl\,l^{-1}}$ that has been determined for damage to sensitive plant species (Fuhrer et al. 1997; Mills et al. 2000; LRTAP Convention 2007). Several scenarios indicate that concentrations of tropospheric O_3 might further increase throughout the 21st century (Gauss et al. 2003); simulations for the period 2015 through 2050, project increases in tropospheric O_3 of 20–25% (Meehl et al. 2007).

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The global patterns of exposure of vegetation to O_3 are also changing. A prediction of the differences in annual global mean surface O_3 concentrations from the 1990s to 2020s has recently been modelled by Dentener et al. (2005), showing increases in all major agricultural areas of the northern hemisphere with large spatial variation. Control measures on emissions of nitrogen oxides (NO_x) and volatile organic compounds (VOCs) applied in North America and western Europe, where the impacts of O_3 on crop production and forest vitality have been well established, are expected to lead to reductions in peak O_3 concentrations (Gardner and Dorling 2000). However, the effect of these changes may be offset by the predicted increases in global background tropospheric concentrations, in particular as a result of increased global emissions of NO_x (NEGTAP 2001). Furthermore, in parts of Asia, Latin America and Africa, these increases in background concentrations are combined with trends of increased emissions of O_3 precursors, suggesting that current and future impacts of O_3 on crops and forests in these areas may be very significant.

Climate change can further affect tropospheric O_3 e.g. by modifying emissions of precursors (European Commission 2003), such as biogenic VOC emissions (e.g. isoprene) that may be highly sensitive to climate change. Although these emissions increase with increasing temperature, certain studies concur that climate-driven changes in vegetation types unfavourable to isoprene emissions (notably the recession of tropical forests) would partly compensate for the effect of warming in terms of O_3 generation (Lathière et al. 2005). Of course, changes in temperature, humidity, UV radiation intensity and atmospheric circulation brought about by climate change could affect production, transport and removal of O_3 significantly and increases in regional O_3 pollution are expected due to higher temperatures and weaker circulation. Other, more indirect effects of climate change may cause either an increase or a decrease in background tropospheric O_3, due to competing effects of higher water vapour and higher stratospheric input (Denman et al. 2007).

Many studies have been conducted on the impacts of O_3 pollution on vegetation, ranging from effects at the cellular level to predicting impacts on a regional and international scale (EPA 1996). O_3 damage to plant tissues includes visible leaf injury, decreased photosynthesis and increased senescence, which has significant repercussions on the yield of major agricultural crops, biodiversity and forest health. There is no doubt that predicted increases in tropospheric O_3 will impact on future agro-ecosystems and their management. Nevertheless, the major current projections of global food production under atmospheric change scenarios do not account for the damaging effect of rising O_3 and current risk assessment tools do not sufficiently consider its interaction with other climatic changes (Long et al. 2005; Easterling et al. 2007). In addition, many coupled climate-carbon models have currently neglected the impacts of changing ground-level O_3 concentrations on carbon cycling (Sitch et al. 2007). The aim of this chapter is to provide an overview of the impacts of O_3 on crops in a changing climate.

10.2 Ground Level Ozone as a Component of Climate Change

The level of O_3 in the troposphere is controlled by a complex set of photochemical reactions involving NO_x, carbon monoxide and VOCs (Penkett 1991; United Kingdom Photochemical Oxidants Review Group 1993; Crutzen et al. 1999). Natural sources of NO_x (e.g. from soils, lightning and transport from the stratosphere) and VOCs (e.g. from soils and vegetation) ensure that there is always a background concentration of O_3 in the troposphere. There is also a contribution from incursions of O_3 from the stratosphere, although this is of minor importance in the global budget (Denman et al. 2007). Anthropogenic emissions of large quantities of O_3 precursors due to fossil fuel combustion and biomass burning, have substantially increased the amount of O_3 since the pre-industrial era. Recent evaluations of surface measurements in the 19th and early 20th century in Europe (Volz and Kley 1988; Harris et al. 1997) clearly indicate much lower O_3 concentrations than today. Since O_3 is relatively short-lived, lasting for a few days to weeks in the atmosphere, ground-level distributions are highly variable and tied to the abundance of its forerunner compounds, water vapour and sunlight. Trends in anthropogenic emissions of O_3 precursors (1990–2000) show reductions in industrialised regions like the USA and Organisation for Economic Co-operation and Development (OECD) Europe, while regions dominated by developing countries show significant growth in emissions (Forster et al. 2007) which has repercussions on the global O_3 distribution. The boundary layer O_3 concentrations show strong diurnal and seasonal cycles with daytime and summertime maxima. O_3 episodes are associated with hot sunny weather and occur over wide areas; peak concentrations occur mainly during the afternoon, when photochemical O_3 production is most active. At night time, however, in the absence of significant O_3 formation, the O_3 concentration may fall due to reactions with NO and through deposition to the surface, which is the major sink for O_3 at rural sites. Moreover, sites that experience the same regional O_3 distribution, may have different exposure levels due to different local features, such as elevation, wind speed, roughness of the earth's surface (Derwent and Kay 1988).

As global O_3 exposures increase over this century, direct and indirect interactions with climate change and elevated CO_2 will modify plant dynamics (Fiscus et al. 2005) and as such, it is vital to evaluate the impact of O_3 on vegetation within a framework of future climatic conditions (Ashmore and Bell 1991). Continued emissions of the most important greenhouse gases, carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) at or above current rates will cause further warming and induce many changes in the global climate system during the 21st century that would very likely be larger than those observed during the 20th century (Meehl et al. 2007). For the next two decades, a warming of about 0.2°C per decade is projected for a range of emission scenarios. Even if the concentrations of all greenhouse gases and aerosols had been kept constant at year 2000 levels, a further warming of about 0.1°C per decade would be expected (IPCC 2007). However, there is unanimous agreement among the models that future climate change will reduce the efficiency of the land and ocean carbon cycle to absorb anthropogenic CO_2, essentially owing to a reduction in land carbon uptake. The latter is driven by a combination of reduced net primary productivity and increased soil respiration of CO_2 under a warmer climate. This positive feedback will lead to atmospheric CO_2 concentrations between 730 and 1020 ppm by 2100 and an additional warming of between 0.1 and 1.5°C (Meehl et al. 2007). Globally averaged mean water vapour and evaporation are projected to increase. Increases in the amount of precipitation are very likely in high latitudes, while decreases are predicted in most subtropical land regions. A warmer future climate will also imply fewer frost days and increased summer dryness with greater risk of drought especially in the mid-continental areas. These projected climatic changes will have an impact on the response of plants to O_3 (Tausz et al. 2007). But the opposite also applies: O_3 itself can modify the response of plants to a range of naturally occurring environmental stresses, such as drought (Bell 1987; Heggestad et al. 1985). Other important interactions may arise from the fact that O_3 alters the performance of herbivorous insect pests and plant pathogens, which will themselves be influenced by the climate change, e.g. as a result of greater survival under milder winter conditions.

10.3 Ozone Uptake and Plant Responses

O_3 is one of the most powerful, highly reactive oxidants and its potential to damage vegetation has been known for over 30 years. One of the first confirmed reports of widespread foliar injury which could be attributed to O_3 was the so-called "weather fleck" of tobacco in the eastern United States (Heggestad and Middleton 1959). Leaves were damaged by straw-coloured flecks making the tobacco unusable. In the early 1960s, another economically important disorder of potatoes called "speckle leaf" was observed in USA (Hooker et al. 1973), which was later associated with the concomitant occurrence of elevated tropospheric O_3 concentrations. Research in recent years has advanced our understanding of the mechanisms underlying O_3 effects on agricultural crops, trees and native plant species and detailed compendia have been produced to illustrate the range of symptoms produced on different species (Krupa et al. 1998).

The key process in relating O_3 exposure to biochemical, physiological and final yield responses is the ease with which O_3 gets access into the stomata. This depends firstly on atmospheric processes above the plant canopy that control the transfer of ambient O_3 towards the vicinity of the leaf surface (Fig. 10.1). This transfer is mainly governed by wind turbulence and the roughness of the terrestrial landscape, including altitude and type of vegetation. At the leaf surface, the thickness and resistance of the boundary layer depend primarily on the wind speed and leaf characteristics, such as orientation, size, shape and hairiness. The actual diffusion of gasses through the stomata, expressed by the stomatal conductance (g_s), is proportional to the atmospheric concentration of O_3, but is also strongly controlled by the

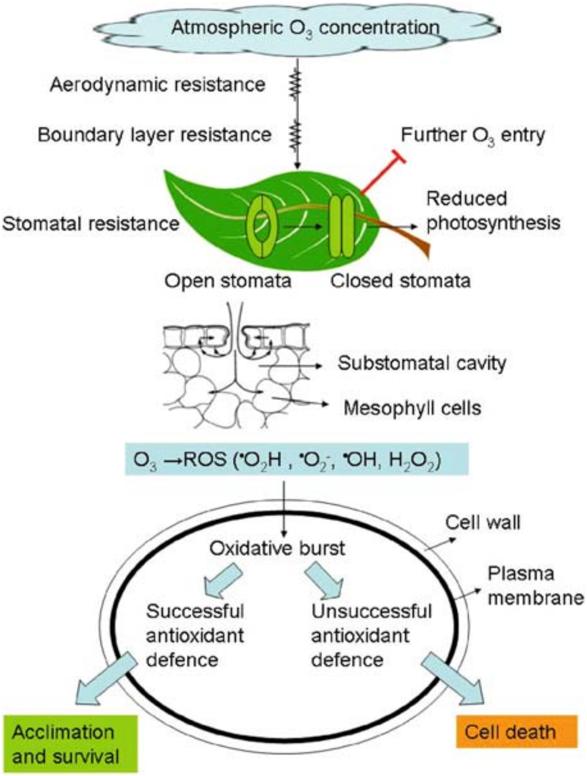

Fig. 10.1 Simplified scheme of O_3 transfer, plant uptake and cell response. For details: see text

stomatal aperture. The importance of these successive resistance factors is not to be ignored. High O_3 concentrations often tend to coincide with the weather conditions that limit the dose of O_3 absorbed by the plant: stagnant weather situations limit the O_3 transfer across the atmospheric boundary layer to the vegetation and high vapour pressure deficits (VPD) (= low relative air humidity) lead to low values of stomatal conductance (Grünhage et al. 1997).

As illustrated in Fig. 10.1, once O_3 has penetrated the leaf via the stomata and substomatal air space, the majority will be absorbed into the aqueous phase of the mesophyllic cell wall matrix where it reacts with water and solutes to form free radicals, such as hydroperoxide ($\cdot O_2H$), superoxide ($\cdot O_2^-$) and the hydroxyl radical ($\cdot OH$), which can further lead to the formation of hydrogen peroxide (H_2O_2) (Long and Naidu 2002). There is now an increasing evidence that these short lived, highly reactive oxygen species (ROS) are involved in O_3-mediated injury (Polle 1998). Although ascorbate and other scavenging systems will partly remove O_3 and ROS, a fraction may still reach and damage the outer proteins and lipids of the plasmalemma, causing cell leakage and loss of solutes. In response to O_3 stress, reactive radicals are also generated inside the cell. Although ROS are considered deleterious and harmful, recent studies indicate that such an "oxidative burst" may be important to stimulate signal transduction pathways that promote plant defence responses and programmed cell death to a wide variety of stimuli, such as high light, heavy metals, mechanical and physical stresses, drought, UV radiation and pathogens (Sandermann et al. 1998; Rao et al. 2000; Van Breusegem et al. 2001). Whether the defence responses are successful or not depends on the concentration of O_3, the duration of the exposure, the plant age, genotype and pre-conditioning.

Loss of photosynthetic capacity is an early phenomenon of O_3 exposure and sometimes the only physiological symptom of damage during chronic exposure. This may be also attributed partly to a decrease in the amount and activity of the CO_2 fixating enzyme, ribulosebiphosphate carboxylase (Rubisco) (Lehnherr et al. 1988; McKee et al. 1995) and partly due to accelerated senescence, with a down-regulation of photosynthetic genes and an up-regulation of genes involved in programmed cell death and/or tissue senescence. Inhibition of CO_2 assimilation can also result from direct or indirect inhibition of stomatal opening that reduces uptake (Saurer et al. 1991; Torsethaugen et al. 1999). These biochemical and physiological changes determine the final effect on plant vitality and productivity, although all the processes by which O_3 leads to reductions in agricultural yield are complex and not always fully understood. For example, the same O_3 episode may have different effects on crop yield depending on when it occurs (Vandermeiren et al. 1995; Soja et al. 2000). This is related to the relative sink and source strength and hence, on partitioning priorities at the time of exposure.

Due to changes in pool size of metabolites, the effects on crop quality can be significant. Importantly, the altered biochemical state, including increases in antioxidant scavenging systems within the tissue, may change the response of the plant to existing environmental conditions and other stresses. This "cross-induction" suggests that distinct stresses may activate the same, or at least overlapping, signal transduction pathways (Sharma and Davis 1994). Rao et al. (2000) indeed showed that pre-exposure to O_3 induced resistance to subsequent pathogen infection.

As such, there are two modes of O_3-induced injury patterns currently categorized by Long and Naidu (2002):

1) Acute injury due to exposure to high concentrations, $120–500\,\mathrm{nl\,l^{-1}}$, for hours, may occur at the most polluted sites. Distinct visible injury like water-logging is detected on the leaves in several hours, the area of which then turns into typical O_3 symptoms (necrotic stippling, bronzing or chlorosis) within a few days. Under these exposure conditions, toxic reactions of O_3 with cellular components exceed the tolerance level of the cells, resulting in cell death.
2) Chronic injury is a consequence of exposure to an elevated background concentration with peak daily concentrations in the range of $40–120\,\mathrm{nl\,l^{-1}}$ over several days in the growing season. This type of injury is more subtle and, depending on plant species, may include symptoms such as chlorosis and premature senescence, resulting in earlier leaf abscission and flowering (Pell et al. 1997). Often no visible injury is observed, but lower rates of photosynthesis do indicate adverse effects on plant vitality. Decreased photosynthetic capacity and accelerated loss of leaf area will depress plant productivity, leading to losses in crop yield.

10.4 Impacts of Ozone on Crop Production and Quality

Whilst extreme acute O_3 exposure damages the plasmalemma to the extent that the cell is unable to maintain its ionic balance and cell death follows, less acute and chronic exposure diminishes whole plant productivity which may affect the final marketable yield prior to the occurrence, if any, of lesion formation (Heagle 1989; Heath and Taylor 1997). Acute injury can cause obvious and immediate loss of economic value, if the market value of the species depends on its visible appearance, such as many horticultural crops. Severe damage has been observed on irrigated crops in the Mediterranean region (Fumigalli et al. 2001). For example, an O_3 episode north of Athens in 1998 caused severe reddening and necrosis on lettuce (*Lactuca sativa*) and chicory (*Chicorium indybus* L. *and endivia* L.) with 100% loss of the commercial value of the products (Velissariou 1999). There have also been several reports of visible O_3 injury in North America and Europe (Emberson et al. 2003), and also in a range of horticultural crops in Taiwan (Sheu and Liu 2003).

It is well established that there are pronounced differences in O_3 sensitivity between species, but also between varieties within species. Many of the published lists of sensitive species are based on visible injury induced by acute O_3 exposure (Mills et al. 2000; Table 10.1), but these rankings may not be related to relative sensitivity in terms of yield or physiology responses to longer-term exposures (Ashmore 2002). When crops were ranked in sensitivity to O_3 by determining the AOT40[1] associated with a 5% reduction in yield, wheat, pulses, cotton and soybean were the most sensitive of the agricultural crops (Mills et al. 2007; Table 10.2). Several horticultural crops, such as tomato and lettuce were of comparable sensitivity. Crops, such as potato and sugar beet, that have green foliage throughout the summer

Table 10.1 List of commercial agricultural and horticultural crops injured by ambient O_3 episode in Europe (from Mills et al. 2000)

Agricultural crops		Horticultural crops	
Bean	*Phaseolus vulgaris*	Courgette	*Cucurbita pepo*
Clover	*Trifolium repens*	Chicory	*Chicorium endiva*
Corn	*Zea mays*	Lettuce	*Lactuca sativa*
Grape-vine	*Vitis vinifera*	Muskmelon	*Cucumis melo*
Peanut	*Arachis hypogea*	Onion	*Allium cepa*
Potato	*Solanum tuberosum*	Parsley	*Petroselinum sativum*
Soybean	*Glycine max*	Peach	*Prunus persica*
Tobacco	*Nicotiana tabacum*	Pepper	*Capiscum anuum*
Wheat	*Triticum aestivum*	Radish	*Raphanus sativus*
	Triticum durum	Red beetroot	*Beta vulgaris*
		Spinach	*Spinacea oleracea*
		Tomato	*Lycopersicon esculentum*
		Watermelon	*Citrullus lanatus*

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[1] AOT40 is the sum of the differences between the hourly mean O_3 concentration (in $\mathrm{nl\,l^{-1}}$) and $40\,\mathrm{nl\,l^{-1}}$ for each hour when the concentration exceeds $40\,\mathrm{nl\,l^{-1}}$, accumulated during daylight hours. This value is expressed in units of $\mathrm{nl\,l^{-1}\,h}$.

Table 10.2 The range of sensitivity of agricultural and horticultural crops to O_3 (see Mills et al. 2007 for response functions and definitions of sensitivities)

Sensitive	Moderately sensitive	Moderately resistant	Insensitive
Cotton, Lettuce, Pulses, Soybean, Salad onion, Tomato, Turnip, Watermelon, Wheat	Potato, Rapeseed, Sugarbeet, Tobacco	Broccoli, Grape, Maize, Rice	Barley, Fruit (plum and strawberry)

months, were classified as moderately sensitive to O_3. In contrast, important cereal crops, such as maize and barley, can be considered to be moderately resistant and insensitive to O_3, respectively.

The reasons for inter- and intra-specific differences in O_3 sensitivity are not yet fully understood. It is clear that stomatal conductance, which determines O_3 uptake, is often related to sensitivity (Reich 1987), although other factors, such as antioxidant levels or the evolution of stress ethylene, can also be related to O_3 sensitivity. There is an evidence for a genetic basis of such difference in sensitivity and inadvertent selection in breeding processes may have affected O_3 tolerance. There are, however, no indications that modern crop cultivars, bred under higher O_3 concentrations than decades ago, are more tolerant to O_3. Indeed, modern Greek cultivars of wheat are actually more sensitive to O_3 than older cultivars (Barnes et al. 1990), suggesting that in wheat, selection for higher yield led to selection for characteristics associated with lower O_3 resistance.

O_3 may also have direct or indirect (through reduced carbon allocation) effects on the reproductive capacity, such as pollen germination, tube growth, fertilization, and the abscission or abortion of flowers, pods, ovules or seeds (Black et al. 2000). Due to changes in metabolite pools, increasing O_3 concentrations can have significant impact on crop quality. Soja et al. (1997) showed that O_3 exposure over 2 years caused a large decrease in sugar content of grape (*Vitis vinifera*). Juice quality was more sensitive to ozone exposure than grape yield (Soja et al. 2004). In oil seed rape (*Brassica napus*), both seed yield and oil content were reduced, which represents an additional economic loss (Ollerenshaw et al. 1999). The CHIP study (Changing Climate and Potential Impacts on Potato Yield and Quality) on potato at seven different sites across Europe, reported both beneficial and detrimental effects of season-long O_3 exposure on tuber quality (Vandermeiren et al. 2005). Vitamin C content was increased, whereas reducing sugar and starch content were reduced (Vorne et al. 2002). Decrease in forage quality of grasslands has been demonstrated both in North America (Powell et al. 2003) and Europe (Fuhrer et al. 1994), which has economic implications for their use by ruminant herbivores.

Such qualitative and nutritional characteristics may become increasingly important from a commercial and industrial viewpoint, and hence more interesting for future research, especially in those countries where demand for food is stable or increasing slowly. On the other hand, in regions where there are problems in maintaining food supplies, the economic and social implications of widespread loss of yield could be very serious in the face of rapidly increasing populations and loss of productive land (Ashmore and Marshall 1999). A 20% increase in surface O_3 by 2050 would result in yield losses relative to today's yields of 5, 4, 9 and 22% for maize, rice, wheat and soybean, respectively, and approximately double these losses by the end of the century (Long et al. 2005). Wang and Mauzerall (2004) project that with the very large increases in surface O_3 projected for east-central China, crop losses for maize, rice and soybean will each exceed 30% by 2020. Although the actual economic costs of O_3-induced crop losses are difficult to assess, the total benefits resulting from various regulatory scenarios, mostly involving reductions of current ambient levels, ranged from about 0.1–2.5 billion US $ in 1980 in the United States (Adams and Horst 2003). Additionally, in 1996, the US Environmental Protection Agency estimated annual national level losses to major crops to be in excess of 1 billion US $ in 1990 (US EPA 1996). Recently, Holland et al. (2006) quantified the range of O_3-induced yield losses for 23 crops in 47 countries in Europe to be €4.4 to 9.3 billion per year for year 2000 emissions, with a best estimate of €6.7 billion per year. The core estimate represents losses equal to 2% of arable agricultural production in Europe. These estimates, however, do not account for damage via visible injury, changes in crop quality, or interactions with pests.

10.5 Modelling Ozone Uptake and Crop Yield Responses

Effects-based research has resulted in the establishment of critical levels[2] of O_3 for vegetation (LRTAP Convention 2007). Historically, critical levels of O_3 for vegetation were based on the concentration of O_3 in the atmosphere, but it has long been recognised that plant responses to O_3 are more closely related to the internal O_3 dose in the leaf, or the instantaneous flux of O_3 through the stomata, than the ambient O_3 exposure (Lefohn and Runeckles 1987; Fuhrer 2000). This approach requires mathematical modelling of the pathway of O_3 into the leaf including atmospheric, boundary layer and stomatal resistances. The influence of plant phenology, irradiance, temperature, vapour pressure deficit and soil moisture on stomatal O_3 uptake is incorporated in the models through their interference with stomatal conductance (Jarvis 1976). The differences between a concentration or flux-based approach can have very important implications for risk assessments across Europe (Simpson et al. 2007). While the highest AOT40 exposures are in central and southern Europe, considerable O_3 fluxes have also been predicted in parts of northern and western Europe. Calculations of cumulative stomatal O_3 uptake over the growing season for wheat and beech for four grid squares (in Sweden, UK, Czech Republik and Spain), which experience quite different AOT40 values, showed very little difference in modelled cumulative stomatal dose, primarily because of the effects differences in phenology and of modelled vapour pressure deficit (Emberson et al. 2000).

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[2] Critical level = concentration above which direct adverse effects on receptors, such as plants, ecosystems or materials, may occur according to current knowledge (UNECE 1988).

Crops and cultivars with equal stomatal conductance may have different tolerances to the same O_3 concentrations. Therefore, Massman et al. (2000) and Tausz et al. (2007) proposed extending the flux approach linking estimates of O_3 dose through stomata to the capacity of defence mechanisms to detoxify the incoming O_3 flux. Both the absolute level of antioxidant enzymes (mainly superoxide dismutase, ascorbate peroxidase, glutathione reductase) and non-enzymatic low-molecular weight antioxidant molecules (ascorbate, glutathione, α-tocopherol), as

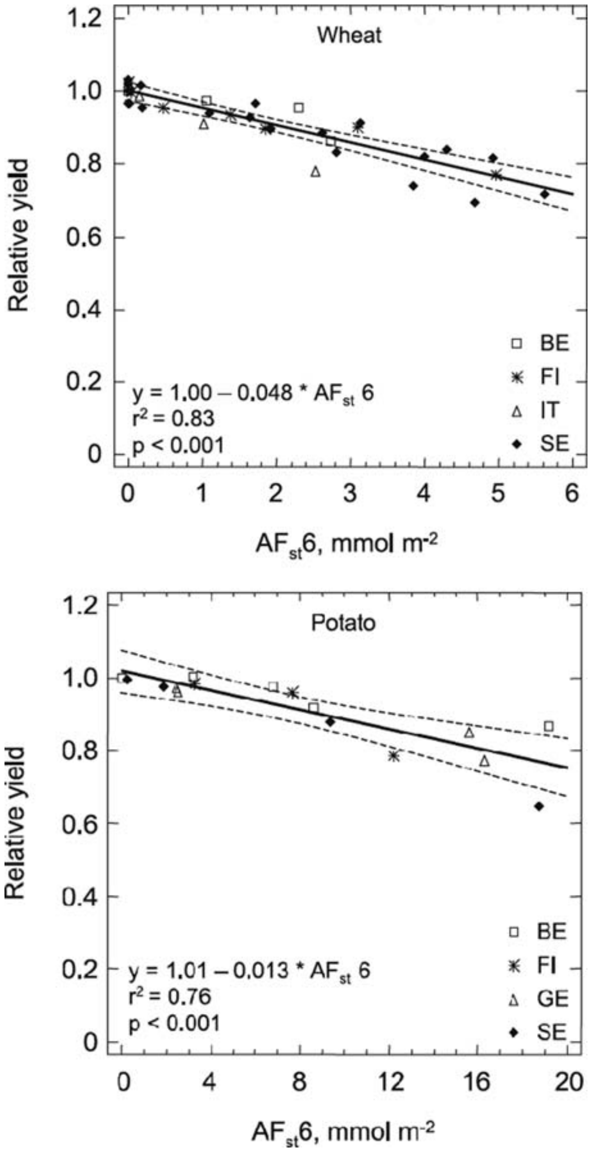

Fig. 10.2 The relationship between the relative yield of wheat and $AF_{st}6$ for the wheat flag leaf based on five wheat cultivars and for sunlit leaves of potato cv Bintje from four European countries. The effective temperature sum was used to describe phenology (BE – Belgium, FI – Finland, IT – Italy, GE – Germany, SE – Sweden) (LRTAP Convention 2007)

well as the capacity to enhance the anti-oxidative potential in response to O_3, might contribute to protection of photosynthetic machinery and membrane functions from oxidative stress. Thus, it has been suggested that O_3 flux models should include a coefficient for O_3 detoxification capacity, such as a threshold O_3 flux (see Fig. 10.2), which clearly improved the performance of flux-response relationships (Pleijel et al. 2004). In another approach, the defence response is regarded as an indirect function of photosynthesis, since photosynthate provides the raw material and energy for the defence and repair processes. Musselman and Massman (1999) defined the "effective flux" as the balance between O_3 uptake into the leaf at a given time and the defence response at that time. The defence response factor is proportional to the effect of O_3 on gross photosynthesis. Martin et al. (2000) proposed to determine a threshold coefficient based on the effects of O_3 on the maximum carboxylation rate of Rubisco. More detailed models are available to describe the extent to which incoming O_3 flux can be detoxified by reactions with ascorbate in the apoplast (Plöchl et al. 2000).

Recently, stomatal flux-based critical levels of O_3 were developed for crops (Pleijel et al. 2007; LRTAP Convention 2007) based on the relationship between the relative yield and modelled stomatal O_3 flux for wheat and potato (Fig. 10.2). The flux-response relationships with the highest correlation were obtained using the exposure index $AF_{st}6$[3].

In combination with the dry deposition model DO$_3$SE (Deposition of Ozone and Stomatal Exchange) that allows the calculation of both stomatal and non-stomatal O_3 deposition for a variety of land covers across Europe (Simpson et al. 2003), such flux-response models allow an estimation of the risk of O_3 impact to vegetation across Europe under different policy scenarios. Further, cost-benefit analysis then allows policy-makers to devise cost-effective emission control programmes.

10.6 Influence of Global Change on Stomatal Flux and Detoxification of Ozone

The flux of O_3 into the stomata and its subsequent detoxification are key determinants of the ultimate response of the plant to the pollutant. Since both are highly dependant on climatic conditions, there is a significant potential for the predicted changes in the climate to influence the response to O_3 through an effect on rates of flux and detoxification. Effects can be direct – e.g. temperature, CO_2 and humidity effects on stomatal conductance or indirect via an influence on soil water potential (SWP) and plant development.

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[3] $AF_{st}6 =$ accumulated stomatal flux of O_3 above a threshold of 6 nmol O_3 m^{-2} projected sunlit leaf area, based on hourly values of O_3 flux.

10.7 Direct Impacts on Stomatal Flux of Ozone

10.7.1 Increased Temperature

The stomatal response to leaf temperature is described by a parabolic function (Fig. 10.3) with a minimum and maximum temperature at which stomatal opening occurs and an optimum temperature for stomatal conductance (g_s) (Emberson et al. 2000). Each plant species will have its own optimum temperature for stomatal conductance and the impact of climate warming on g_s will depend on which part of the temperature response function corresponds with the current ambient temperature. In temperate moist climates, an increase in temperature is likely to result in an increase of g_s and therefore, an increase in the stomatal uptake of O_3, possibly resulting in enhancement of leaf damage. For those plants already at their optimum temperature for g_s, warming is likely to result in a decrease of g_s and stomatal uptake of O_3, possibly resulting in a reduction of leaf damage. The changes described can be predicted for sunlit leaves at the top of the canopy, however, the consequences for the stomatal flux into the whole canopy are yet unknown.

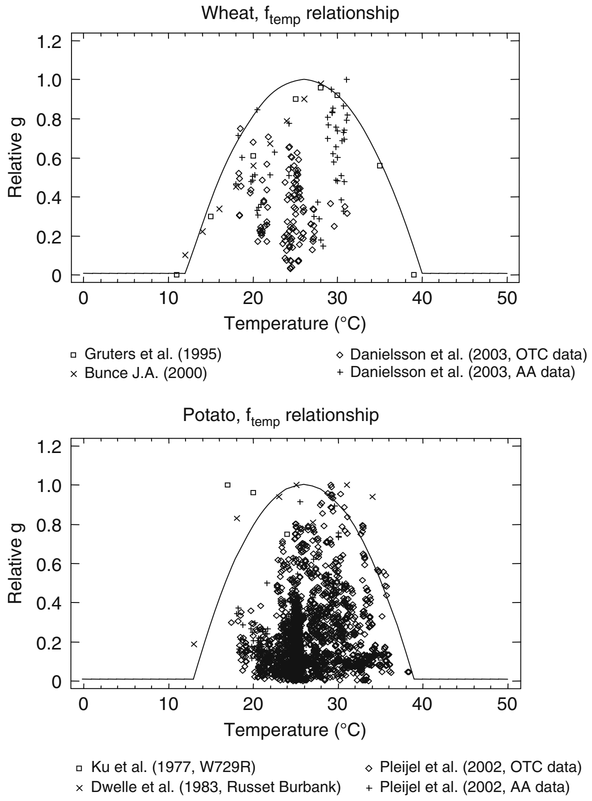

Fig. 10.3 Parameterisation of wheat and potato stomatal conductance models. The function shown (f_{temp}) describes the dependence of the relative stomatal conductance (g) on temperature (see Chap. 3 of the Mapping Manual (LRTAP Convention 2007) for details)

10.7.2 Increased CO_2 Concentration

Many short- and long-term studies have shown that plants grown in an elevated CO_2 environment have reduced g_s (Allen 1990; Drake et al. 1997). In potato, this CO_2-induced reduction in g_s was not affected by exposure to O_3 per se (Vandermeiren et al. 2002), and thus increasing CO_2 concentrations are likely to decrease O_3 uptake. Indeed, in most studies, elevated CO_2 leads to a reduction of O_3 induced yield losses (e.g. Fiscus et al. 1997; Volin et al. 1998; Craigon et al. 2002).

10.7.3 Changes in Precipitation Patterns

In simplest terms, effects of changes in precipitation patterns are likely to be mediated directly through (i) effects of vapour pressure deficit (VPD) on g_s with increasing VPD causing a decrease in flux and (ii) changes in soil water potential (SWP), with decreasing SWP resulting in decreased stomatal flux and vice versa (Fig. 10.4). Interactions between O_3 stress and plant water use efficiency (WUE) are important due to co-occurrence of high O_3 levels, combined with reduced soil moisture during warming. A reduction in soil moisture below a threshold will reduce the stomatal

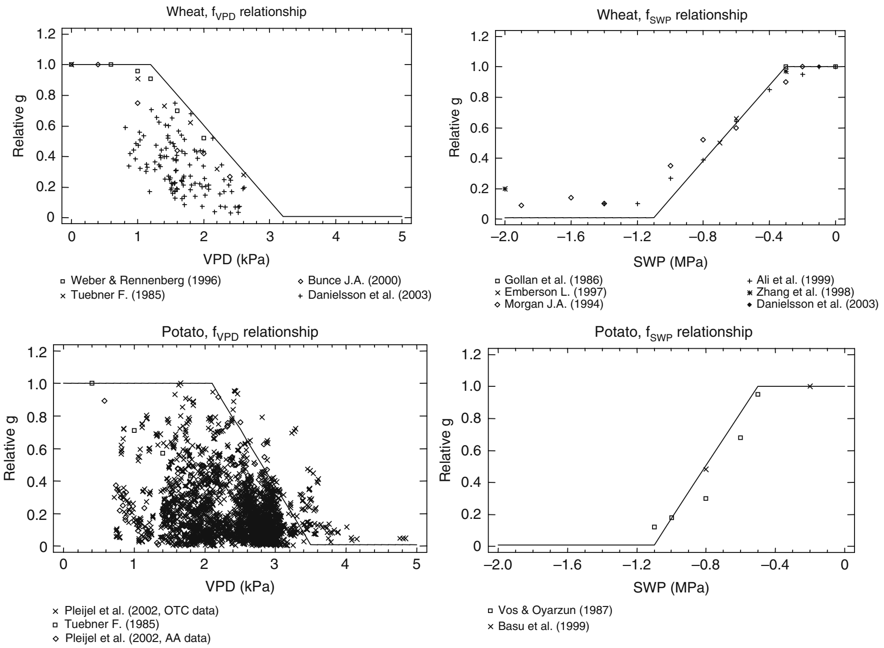

Fig. 10.4 Parameterisation of wheat and potato stomatal conductance models. The functions shown (f_{VPD} and f_{SWP}) describe the dependence of the relative stomatal conductance (g) on vapour pressure deficit (VPD) and soil water potential (SWP) (see Chap. 3 of the Mapping Manual (LRTAP Convention 2007) for details)

flux of O_3, resulting in less O_3 uptake and damage in plants during peak levels of O_3. However, during periods of drought, many crops are often irrigated, which reduces the limitations of soil moisture deficit on the stomatal uptake of O_3, resulting in high stomatal O_3 fluxes. For example, in Greece, no symptoms of O_3 injury were observed on non-irrigated chicory plots, whilst irrigated plots on the same farm suffered severe leaf damage (Velissariou 1999).

10.7.4 Effects of Combined Changes

The summarised information presented in the previous paragraphs assumes that the predicted changes are occurring in isolation. However, the situation is far more complex as the various changes associated with climate change will act together to influence stomatal uptake. Some examples are provided to illustrate the complexity of the interactions.

10.7.4.1 Elevated CO_2 Increases Leaf Temperature

Whilst a rise in CO_2 levels would be expected to reduce stomatal opening and thus reduce uptake, the reduced g_s will increase leaf temperature due to a reduction in leaf evapotranspiration, which adds to an enhanced effect of temperature on g_s. In free-air concentration enrichment (FACE[4]) systems, rises in canopy temperatures between 0.6 and 1.1°C have been reported at elevated CO_2 concentrations and such increases should be added to those predicted for global warming (Kimball et al. 2002). The resultant increase in leaf temperature would either increase or decrease conductance depending on whether the optimum temperature for g_s has been reached. Potentially, the effects of elevated CO_2 and temperature on the g_s of leaves could cancel out each other.

10.7.4.2 Changes in Soil Moisture at Elevated CO_2

In general, decreases in g_s with elevated CO_2 are exacerbated by drought stress which also tends to close stomata (Kang et al. 2002). Partial closure of stomata results in increased WUE, indicating that plants may be able to withstand more moderate water stress. However, improved WUE detected at the leaf level may not cause the expected reduction in whole plant water consumption as whole canopies may still consume equal amounts or even more water due to an elevated CO_2-induced increase in total leaf area and leaf temperature (Riedo et al. 1999; Hui et al. 2001; Kimball et al. 2002). Elevated CO_2 tends to increase soil moisture by reducing plant water uptake (Owensby et al. 1999; Kimball et al. 2002), which could potentially result in a lower reduction in both g_s and uptake of O_3. In general, elevated CO_2 will reduce the leaf uptake of O_3, but the impact of CO_2 enrichment on O_3 uptake at

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[4] FACE = a circular or octagonal system of pipes that release treatment gas, or air enriched with the treatment gas, just above the crop canopy.

the canopy level will also be affected by elevated CO_2-induced changes in leaf area index and soil moisture.

Interactions between O_3 stress and WUE are important due to the co-occurrence of high O_3 levels, combined with reduced soil moisture during warm weather. Reduced leaf WUE in response to O_3 was found in wheat (Saurer et al. 1991) and soybean (Vozzo et al. 1995). This effect may be linked to direct negative effects of O_3 on stomatal functioning (Leipner et al. 2001; McAinsh et al. 2002) or the stronger sensitivity of photosynthetic CO_2 fixation relative to the g_s of water vapour (Saurer et al. 1991). But because of a concurrent reduction in crop biomass, O_3 is not likely to be having an effect on total crop water consumption.

10.7.5 Indirect Impacts on Stomatal Uptake

The complexity of the interactions between the factors involved in climate change is well illustrated by consideration of the impacts of global warming on the canopy uptake of O_3. When considered as a single factor, increased temperature is likely to increase stomatal uptake of O_3, provided the optimum for g_s has not been reached (Fig. 10.3). However, the response to warming will also be affected by the following indirect effects:

- Providing the precursors are present, an increase in warming would increase the rate of tropospheric O_3 formation with a consequent increase in O_3 concentration surrounding the leaves and available for stomatal uptake;
- Warming will result in a increase in VPD and decrease in SWP (soils will dry out faster due to enhanced soil evaporation and enhanced canopy evapotranspiration), which will generally result in a decrease in the stomatal flux of O_3 into leaves due to a reduction in g_s (Emberson et al. 2000). Semi-arid regions are likely to be most sensitive to warming as the potential evapotranspiration increases by about 2–3% for each 1°C rise in temperature (Fuhrer and Booker 2003);
- Warming will enhance plant development, which will reduce the stomatal flux of O_3 into leaves at a later stage of development (Emberson et al. 2000).

Thus, the overall impact of warming on the canopy flux of O_3 is difficult to predict and will depend on the severity and timing (e.g. summer or winter) of warming, future changes in SWP and the phenological stage of the vegetation, as well as any changes in seasonal pattern in the occurrence of peak episodes of O_3. In a first simple modelling approach for winter wheat, Harmens et al. (2007) showed that in a future climate (including increases in background O_3 concentration, temperature and CO_2, changes in VPD and precipitation), the exceedance of the flux-based critical level of O_3 might be reduced over Europe. In contrast, under the same scenario, the exceedance of the concentration-based critical level would increase. Izrael (2002), on the other hand, predicts that the projected warming accompanied by a 30% increase in tropospheric O_3 and 20% decline in humidity would decrease the grain and fodder productions by 26 and 9%, respectively in North Asia.

10.8 Detoxification of Ozone in Plants in a Changing Climate

Of all factors involved in global climate change, the influence of elevated CO_2 on O_3 sensitivity has been the subject of the majority of experimental studies involving O_3 impacts on plants. In contrast with the general reduction of uptake of O_3 by plants due to a reduction in g_s at elevated compared to ambient CO_2, impacts of CO_2-enrichment on the antioxidant status of leaves are not consistent. In barley, primary leaves developed early senescence at elevated compared to ambient CO_2, which was associated with a decline in antioxidant capacity, resulting in oxidative stress in this otherwise O_3-insensitive species (Robinson and Sicher 2004). These results suggest that protection of aging leaves from O_3 damage might be reduced at elevated CO_2. In a FACE experiment with deciduous trees, it was found that although elevated CO_2 decreased g_s, the relative decrease in dry matter production and photosynthesis caused by elevated O_3 was the same at ambient and elevated CO_2, suggesting that elevated CO_2 grown tissue was metabolically less tolerant of O_3 (Karnosky et al. 2005). A study on wheat has also suggested that CO_2 enrichment will render plants more susceptible to O_3 at the cellular level (Barnes et al. 1995). In contrast, Rao et al. (1995) indicated that elevated CO_2 might protect wheat leaves against O_3-induced oxidative stress by prolonged enhancement of the antioxidant status, whereas McKee et al. (1997b) concluded that stomatal exclusion rather than the antioxidant status plays a major role in the protective effect of elevated CO_2 against O_3 damage in wheat. Recent results showed that leaf antioxidant enzyme activity (e.g. superoxide dismutase) was involved in conferring CO_2-induced tolerance to O_3 stress in soybean (Lee 2000). Both high light and elevated carbohydrate levels favoured the maintenance of high total ascorbic acid in leaves, which often correlates with O_3 tolerance. For the same crops, Booker and Fiscus (2005) found that equivalent O_3 fluxes, that suppressed net photosynthesis, growth and yield at ambient concentrations of CO_2, were generally much less detrimental to plants treated concurrently with elevated CO_2.

In summary, a general ameliorating effect of CO_2-enrichment on O_3-induced oxidative stress via changes in the antioxidant status of leaves has not been proven as experimental data are inconclusive. Predictions cannot yet be made for impacts of small increases in temperature on the antioxidant status of leaves, since temperature effects have mainly been reported in relation to chilling or heat stress rather than the effects of a few degrees rise in temperature.

10.9 Impact of Elevated O_3 and CO_2 on Visible Injury, Crop Yield and Quality

Since both O_3 and CO_2 have strong effects on photosynthesis and crop production, knowledge of crop responses to a combination of elevated 'greenhouse' concentrations of both gases is one of the most important issues in view of future climate changes. While elevated CO_2 generally has a growth stimulating effect due to an increase in photosynthesis, O_3 tends to have the opposite effect. Moreover, both components can directly affect biochemical and physiological processes, such as plant senescence, that might influence plant response to other biotic or abiotic stresses (Barnes and Davison 1988).

To date, studies that have examined the interactive effects of CO_2 and O_3 have shown a variety of plant responses (Polle and Pell 1999; Olszyk et al. 2000). The nature of the interaction may be influenced by the characteristics of the O_3 exposure pattern (timing in relation to phenological development, chronic or acute exposure), plant species, water availability, and other climatic parameters, but it will also depend upon the kind of effect that is considered i.e. total biomass or economic yield, photosynthesis, visible injury etc. In general, elevated CO_2 reduces O_3-induced leaf damage and yield losses, primarily through O_3 exclusion via a reduction in g_s, but also to a certain extent due to an increased detoxification capacity.

10.9.1 Visible Injury

In a number of crop species, O_3 injury to leaves was reduced substantially by elevated CO_2, e.g. tobacco (Heck and Dunning 1967), spring wheat (Mortensen 1990; McKee et al. 1995; Rao et al. 1995; Fangmeier et al. 1996; Mulholland et al. 1997; Cardoso-Vilhena et al. 1998), radish (Barnes and Pfirrmann 1992), barley (Fangmeier et al. 1996), snap bean (Cardoso-Vilhena et al. 1998) and potato (De Temmerman et al. 2002). For wheat, CO_2 exposure influenced the severity of visible leaf damage and protected against O_3 induced premature senescence during early vegetative growth (Mulholland et al. 1997). However, elevated CO_2 had only a limited protective effect (reduced foliar injury) for a sensitive clone of white clover (Heagle et al. 1993) and had no effect on O_3 injury to leaves of *Phaseolus vulgaris* (Heck and Dunning 1967). Exposure of C_3 and C_4 grass species to O_3 increased leaf dark respiration and decreased photosynthesis (Volin et al. 1998), which was not the case in an elevated CO_2 environment. As repair processes on the cellular level depend primarily on the dark respiration, the cost of repair is lower in the elevated CO_2 environment. The length of the dark period is also very important for the plant to recover from the O_3 exposure during the day. As De Temmerman et al. (2002) pointed out, crops, such as potato, show visible symptoms at much lower O_3 concentrations during the long days in summer, when the dark period becomes too short for repair.

10.9.2 Yield and Quality

The relative yield stimulation by elevated CO_2 tends to be larger in an atmosphere with elevated levels of O_3, or vice versa, in a CO_2-enriched atmosphere, negative effects of O_3 are less than at ambient CO_2. In determinate crops, such as cereals, grain yield not only depends on photosynthesis, but also on the length of the active phase of leaf photosynthesis and the sink capacity of the grains. In wheat, elevated CO_2 fully protects against the detrimental effects of O_3 on biomass, but not yield (McKee et al. 1997a). Similar results have been reported with soybean (Fiscus et al. 1997), cotton (Heagle 1989) and tomato (Reinert et al. 1997). On the other hand, Pleijel et al. (2000) found that the grain yield in wheat was negatively affected by O_3 at ambient CO_2 but unaffected by O_3 at elevated CO_2. Responses of wheat to elevated O_3 and CO_2 appear to be cultivar-dependant, as some cultivars do not respond significantly to elevated O_3 levels and for those cultivars, no significant interactions between O_3 and CO_2 were observed (Bender et al. 1999). Although elevated CO_2 did not prevent O_3 induced yield losses in potato, the yield increases in response to CO_2 fertilisation far exceeded O_3-induced losses (Craigon et al. 2002). In potato, significant interactions between O_3 and CO_2 were observed regarding the glucose and reducing sugar content in tubers (Vorne et al. 2002). Despite the beneficial impact of CO_2 enrichment on growth and yield of C_3 cereal crops, declines in flour quality due to reduced N content are likely in a CO_2-enriched world (Fangmeier et al. 1999), thereby counteracting the positive effect of O_3 on flour quality (Vandermeiren et al. 1992; Pleijel et al. 1999).

It has been indicated that the maximum benefits for wheat production in response to CO_2 enrichment will not be accomplished under concomitant increases in tropospheric O_3 concentration (Rudorff et al. 1996). This implies that predictive models based simply on the impacts of elevated CO_2 will result in an overestimation of the likely effects of atmospheric change on plant productivity (Barnes and Wellburn 1998). Importantly, Long et al. (2005) pointed out that chamber studies, which have been the main mechanistic basis for crop yield models, overestimate the yield gain by elevated CO_2 compared to what was observed under fully open-air conditions in the field, as provided by FACE systems. Based on chamber experiments, average yield stimulation for C_3 crops with a doubling of CO_2 has been estimated at 30%, whilst estimates based on results from field-scale experiments under more realistic conditions (including varying water availability) were lower. According to a review by Kimball et al. (2002) on responses of agricultural crops in FACE systems, elevated CO_2 stimulates biomass in C_3 grasses by an average of 12%, grain yield in wheat and rice (*Oryza sativa* L.) by 10–15%, and tuber yield in potato by 28%. Yield stimulation in C_4 crops is much lower. Some environmental differences between chamber and open air micro-climate also have an influence on the plant interaction with O_3 uptake and detoxification (Morgan et al. 2003). Morgan et al. (2006) observed in an open-air study that the effects of season-long elevation of O_3 induced substantially greater grain losses in soybean compared to chamber experiments. If this is representative of other major crops and growing areas, then yield losses due to rising O_3 will even outweigh any gains due to rising CO_2 (Long et al. 2005).

While leaf-level responses to elevated CO_2 and O_3 are well documented, a few studies have addressed canopy level responses to increases in these pollutants. In SoyFACE, the results show a decrease in evapotranspiration of soybean for all three treatments ($+CO_2$, $+O_3$, $+CO_2$ and O_3), with the largest decrease observed for growth in elevated O_3 (Bernacchi et al. 2006). When integrated over the season, plants grown in elevated CO_2 and O_3 used 10 and 18% less water, respectively.

While the directional response of soybean exposed to increases in CO_2 and in O_3 were similar, the mechanisms for these responses differ. Growth in elevated O_3 resulted in a decrease in leaf area compared with the control. It is likely that the O_3-induced damage to the plant canopy, responsible for the lower biomass and leaf area, is responsible for the lower evapotranspiration in soybean. On the other hand, soybean grown in elevated CO_2 demonstrated higher leaf area while showing a reduction in evapotranspiration suggesting that a decrease in g_s was sufficient to more than offset the increase in leaf area (Bernacchi et al. 2007). These results imply that future atmospheric change may influence soybean response to drought conditions and may have feedback effects on atmospheric moisture, potentially altering regional precipitation patterns.

10.10 Impacts of Increased Ground-Level Ozone on Weeds, Pests and Diseases

The occurrence of plant pests (weeds, insects or microbial pathogens) is an important constraint with global average yield losses estimated at about 40% (Oerke et al. 1994), and production costs significantly dependent on the extent of measures necessary for plant protection. Consequently, changes in the occurrence of pests due to increased O_3 and its interaction with other climatic changes are of economical importance. Because insect and plant species show individual responses to climate change, it is expected that climate change will affect the temporal and spatial association between species interacting at different trophic levels (Harrington et al. 1999). Changes in foliar chemistry and surface characteristics by O_3 may have an effect on the incidence of viral and fungal diseases and the impacts of insect pests, although not much experimental information is available on this subject in relation to crops because these are mostly grown under conditions which prevent the occurrence of pests and diseases.

10.10.1 Weeds

Virtually nothing is known about effects of elevated O_3 on crop-weed interactions (Fuhrer and Booker 2003), but O_3 may potentially affect the ability of weeds and crops to compete for common resources. In the case of aggressive weed species of tropical or subtropical origin, future climatic conditions may lead to their expansion into temperate regions. The generalized prediction that in a CO_2-rich world, major crops will compete more successfully with the worst agricultural weeds, which are mostly C_4 species (Dukes and Mooney 1999), may not be accurate due to regional differences. For instance, in US, 9 out of 15 worst weeds in the most important crops are C_3, and a substantial fraction of crops are C_4 (Bunce and Ziska 2000). The interactions between weeds and crops or grassland species in relation to climate change are complex and further experiments with different crop-weed systems under a range of atmospheric and edaphic conditions, are needed for accurate predictions.

10.10.2 Insect Pests

It is generally expected that with climate warming in mid- to high-latitude regions, insect pests become more abundant. The main effect of climate warming in the temperate zone is believed to be a change in winter survival, while in the northern latitude shifts in phenology in terms of growth and reproduction, may be of prime importance (Bale et al. 2002). Shifts in climate are also likely to affect the geographical distribution of insects. Species-specific responses of pests to increasing O_3 concentrations were observed (see Fuhrer 2003).

Most studies have related changes in insect performance to changes in foliar concentrations of nitrogen, carbohydrate and phenolics. In contrast to elevated CO_2, O_3 tends to increase the N concentration in leaves (Pleijel et al. 1999). However, the relevant plant nutritive factor in the response of herbivores to O_3 might be the carbohydrate rather than the N concentration in leaves, as O_3 stress increases herbivore development as well as foliar sugar and starch in some host-plant systems (Heagle et al. 1994; Hummel et al. 1998). Glucosinolate content of rape seed oil has been reported to be influenced by O_3 exposure (Gielen et al. 2006). These nitrogen- and sulfur-containing secondary plant metabolites are primarily involved in plant-insect interactions, but in high concentrations, they may become toxic for animal feed, whereas moderate consumption has beneficial effects on human health (Stoewsand 1995). Elevated CO_2 and O_3 may alter the performance of insects through changes in bottom-up (plant) and top-down (natural enemy) control. In summary, in an atmosphere containing higher levels of both O_3 and CO_2, increased populations of some insect pests can be expected.

10.10.3 Plant Diseases

Impacts of climate change on specific host-pathogen systems are variable (Coakley et al. 1999; Chakraborty et al. 2000). In addition, O_3 effects on plants lead to altered disease susceptibility, but the effect is inconsistent. In wheat, leaf rust disease was strongly inhibited by O_3, but largely unaffected by elevated CO_2, both in the presence and absence of O_3 stress (Von Tiedemann and Firsching 2000). O_3 damage to leaves was strongly affected by CO_2 concentration and infection. On infected plants, O_3 lesions appeared 2–4 weeks earlier and were up to fourfold more severe compared to non-infected plants. Elevated CO_2 did not delay the onset of lesions, but it significantly reduced the severity of leaf damage. The relative increases in growth and yield induced by CO_2 were much larger on O_3-stressed than on non-stressed plants. Both, O_3 and fungal infection reduced biomass formation and grain yield. However, adverse effects of leaf rust infection were more severe. Elevated

CO_2 compensated for the negative effects of O_3, but not for the detrimental effects of fungal infection. Therefore, the impact of O_3 in the field should not be estimated without considering the predisposing effects deriving from fungal infections and the compensating effects deriving from elevated CO_2. The interaction between O_3 and pathogens may be determined primarily by the timing of O_3 exposure relative to the presence of inoculation. The outcome of plant-pathogen interactions may strongly vary with timing, stage of plant development, predisposing factors, and environmental conditions (Fuhrer 2003).

10.11 Conclusions, Policy Implications and Research Needs

This review has revealed that there is currently much uncertainty in predicting the effects of O_3 in the changing climate of the 21st century. When the changes in mean global O_3 concentration with changing profile (decreased peak with increasing background concentrations) are also brought into the predictions for the 21st century, the scope for generalising responses is rather limited. Furthermore, feedback mechanisms will influence the magnitude of climate change and therefore the effects on crops. For example, a reduction in the stomatal O_3 flux in a future climate might lead to an increase in ambient O_3 concentrations due to a reduction in stomatal deposition to vegetation and therefore result in enhanced radiative forcing. O_3-induced productivity losses would continue to affect the global carbon cycle by reduced sequestration (Sitch et al. 2007).

The flux-based approach for the risk assessment of O_3 effects on vegetation offers a possibility to include the modifying influence of the environment on plant responses to O_3. As such, it becomes possible to predict with higher certainty how plants will respond to the cocktail of environmental conditions which is of particular relevance for evaluation of the impacts of global climate change and for the modelling of pollutant effects across continents. However, there are also disadvantages that need to be considered. These include the higher requirements of measured and modelled meteorological data to drive the models, the more complicated calculations to make risk assessments, and the challenge of communicating results which are based on more complex models than those purely using pollutant concentrations. These disadvantages are, however, inevitable to permit the generalisation of results, for instance, between years with different weather conditions, sites with different climate, estimations of future impacts of climate change and the quantification of economic consequences of O_3 pollution (Pleijel 2005).

10.11.1 Policy Implications

When considering the policy implications of the effects of O_3 in a changing climate, the following sources of uncertainty need to be taken into account:

- Plant responses to climate change (including elevated O_3 concentrations) are species-specific and sometimes even cultivar-specific, and are influenced by other factors, such as local climate, soil characteristics and management.
- Vegetation responses to changes in single drivers of climate change cannot simply be scaled up to predict responses to changes in multiple drivers.
- Results from field release (FACE) systems provide the most reliable information on plant responses to combinations of drivers. Although FACE studies generally substantiate predictions based on chamber studies, some inconsistencies between the results of chamber and FACE studies have been reported. Results from chamber-based experiments might over- or underestimate crop responses to environmental changes.

10.11.2 Conclusions

Taking into account the uncertainties described above and the current information available the following conclusions on the plant responses to O_3 in a changing climate can be tentatively drawn:

- There is a tendency for O_3 effects to be less pronounced at elevated CO_2. This effect has been detected in several crop species. Primarily, CO_2-induced reduction in the stomatal uptake of O_3 is believed to be responsible for this beneficial effect.
- Global warming may reduce the ameliorating effect of elevated CO_2 on the plant response to O_3. Under conditions of no water stress, at a temperature below the optimum for the species concerned, warming is likely to increase O_3 uptake, while a rise in elevated CO_2 is likely to decrease uptake.
- The combined drivers of climate change will influence the flux of O_3 into leaves and will thus influence the magnitude of the effects for a given atmospheric concentration. The magnitude and direction of impact on flux will depend on the combined response to temperature, CO_2 concentration, vapour pressure deficit, soil moisture content, O_3 concentration and plant development stage at any moment in time.
- There is no conclusive evidence that elevated CO_2 will increase the rate of O_3 detoxification within plants. Responses appear to be species- specific. More studies show no effect than others showing an increase in detoxification of O_3. The generalized conclusion is that the beneficial effects of CO_2 in reducing O_3 effects are mediated by stomatal closure rather than detoxification.
- Responses of insects and pathogens to the combined effects of elevated CO_2 and O_3 are species-specific. By altering leaf chemistry, both O_3 and CO_2 can enhance insect and pathogen attack in some species.
- Elevated O_3 at relatively low concentrations can significantly reduce the growth enhancement by elevated CO_2 and therefore reduce C sequestration. This may mean that worldwide vegetation growth stimulations will not be as great as previously predicted from elevated CO_2 studies. Thus, it is important to bring an

understanding of O_3 as a moderator of CO_2 responses in global models of terrestrial net primary productivity and C sequestration. The situation is even more complicated when plant-herbivore interactions or feedbacks operating through the soil are considered.

10.11.3 Research Recommendations

The following research recommendations are being made to improve scientific knowledge on the impacts of ozone on crops in a changing climate:

- The influence of climate change should be taken into account when predicting the future effects of O_3 on vegetation and vice versa.
- Further improvements linking the actual O_3 uptake and detoxification processes, to carbon assimilation and allocation will be needed to provide an even more reliable mechanistic basis for future risk assessments (Ashmore 2005; Musselman et al. 2006). Furthermore, the flux approach needs to be scaled to the canopy level to be generally applicable (Fiscus et al. 2005)
- In the longer-term, it may be necessary to develop alternative modelling procedures since the current method is based on the stomatal responses to climatic and plant factors considered in isolation rather than in combination.
- There is a clear need for multi-factorial experiments to provide information for O_3-effect modelling. Because of the high cost involved with FACE systems, this may only be possible using enclosed or semi-enclosed chamber systems, especially where warming and CO_2 are considered as factors.
- There is a need for a wider and longer-term use of FACE for crops to expand the range of species for which data exists.
- Much more work is needed before unified mechanistic models of O_3 response can be developed that apply to all main agricultural and horticultural crops.

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Chapter 11
Ozone-Induced Changes in Plant Secondary Metabolism

Marcello Iriti and Franco Faoro

11.1 Introduction

In their ecosystem, plants have to cope with a plethora of potentially unfavourable conditions. Stress factors affecting plant's fitness not only derive from natural sources, such as adverse temperature fluctuations (heating, chilling and freezing), high irradiance (photoinhibition, photooxidation), osmotic imbalance (salinity and drought), hypoxia/anoxia (flooding), mineral (macro- and micronutrient) deficiency, wounding, phytophagy and pathogen attack, but also from anthropogenic activities. The latter include xenobiotics employed in agriculture (herbicide, pesticides and fungicides), environmental (air, soil and water) pollutants and increased UV radiations. Particularly, many atmospheric pollutants, belonging to greenhouse gases, may increase the greenhouse effect, a natural warming process that prevents heat from diffusing to the outer atmosphere, thus balancing Earth cooling processes. Without the natural greenhouse effect, temperature on Earth would be much lower than it is now, and the existence of life would have not been possible. However, the rising emissions of greenhouse gases due to anthropogenic activities, namely carbon dioxide (CO_2), chlorofluorocarbons (CFCs), nitrous oxide (N_2O), tropospheric ozone (O_3) and water vapour, may cause a short-term increase of the mean global temperature on the planet surface with consequent changes in precipitation patterns (Krupa and Kickert 1989). In this scenario, life on the earth depended from the co-evolution between atmosphere and biosphere, because the gradual and long-term climate changes enabled living organism adaptation to the new temperatures, precipitation patterns and other climate conditions (Voronin and Black 2007).

Regardless of natural or anthropogenic stress factors, plants have to cope with their stressors. From a pathophysiological point of view, a plant may avoid or adapt to a particular stress, with a dose-dependent mechanism (Fig. 11.1). Under a certain threshold, a mild stress may be compensated by the plant, whereas, at higher

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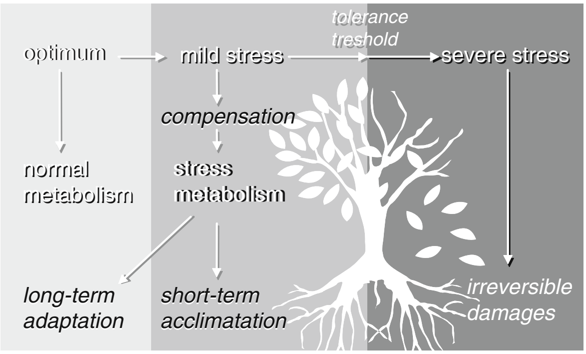


Fig. 11.1 Stressor dose-stress effect relationships in plants

concentration, the detrimental effects of a severe stress may cause irreversible damages, according to the stressor dose-stress effect relationship (Lichtenthaler 1998). Furthermore, the stress tolerance threshold depends not only on the type of stressor and exposure time, but also on the plant stress-coping capacity. Anyway, the shift between normal and stress metabolism represents a fundamental trait in plant acclimation (short-term) and adaptation (long-term) strategies, although it is almost impossible to define exactly the threshold between them (Heiser and Elstner 1998).

In this chapter, we deal with the phytotoxic potential of tropospheric ozone, due to partially reduced oxygen intermediates produced by the pollutant in biological systems. These intermediates are more reactive than molecular oxygen in its ground state and include both radical (superoxide anion, $\cdot O_2^-$ and hydroxyl radical, $OH\cdot$) and non-radical (hydrogen peroxide, H_2O_2) forms, collectively termed as reactive oxygen species (ROS) (Halliwell 2006). Plant tolerance mechanisms will also be discussed with emphasis on ozone-induced metabolic fluxes between primary (normal) and secondary (stress) metabolism, due to acute or chronic pollutant exposure. Besides, ozone is a greenhouse gas, although it plays a minor role in regulating the air temperature and in contributing to the warming effect (Wang et al. 1995).

11.2 The Ozone in the Atmosphere

Ozone is an important constituent of the atmosphere, although present in trace amounts. As a matter of fact, two different pools of O_3 exists, the beneficial and the detrimental one (Fig. 11.2). In the stratosphere (the higher atmosphere, ranging approximately from 15 to 40 km in altitude), the ozone layer absorbs the harmful UV-B and UV-C radiations, thus saving the living organisms (Dutsch 1978; Kerr and McElroy 1993). In the past decades, emission of ozone-depleting chemicals led to the reduction of the ozone shield (commonly referred to as 'ozone hole') against UV radiation, worsening its harmful effects on animals and plants (Platt and Hönninger 2003). Otherwise, in the troposphere (the lower part of the atmosphere, approximately from the earth surface to 10–12 km in altitude), that is to say the layer

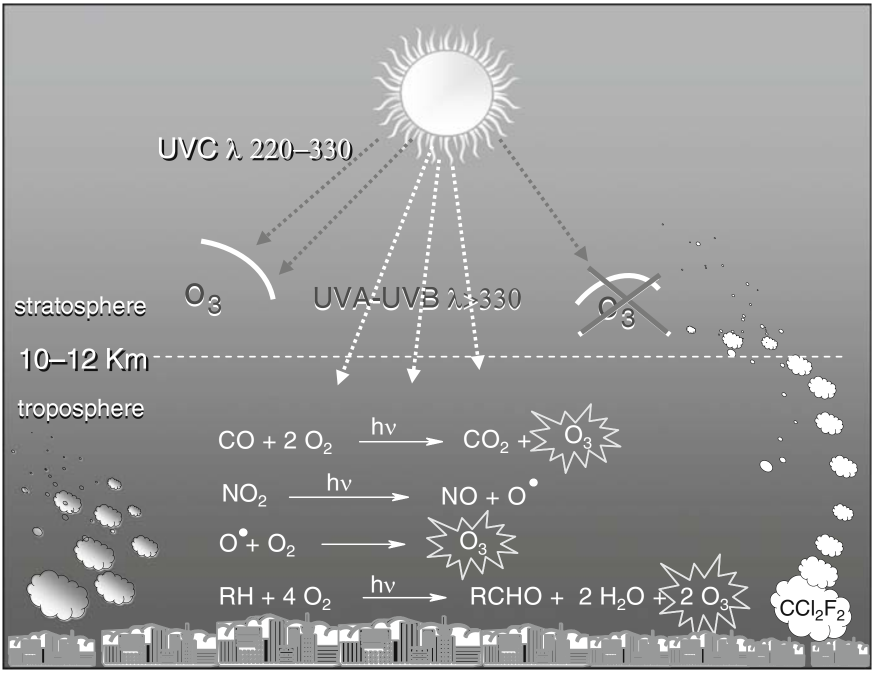


Fig. 11.2 Beneficial and detrimental pools of ozone in the stratosphere and troposphere, respectively. In the stratosphere, this pool is depleted by fluorochlorocarbons (CCl_2C_2) while in the troposphere, it is raised by photochemical smog

where the climatic conditions originate, and temperature decreases with elevation, ozone is regarded as a pollutant (Logan 1985).

Tropospheric ozone is an oxidant constituent of the photochemical smog. It is a secondary pollutant produced through reactions among primary pollutants, emitted directly into the air (mainly nitric oxides, sulphur oxides, carbon oxides and hydrocarbons), catalyzed by sunlight (Crutzen and Lelieveld 2001) (Fig. 11.2). Hence, ozone is produced on bright sunny days over areas with intense primary pollution, mainly due to vehicle exhausts, fossil fuel burning and industrial processes, in the so-called photochemical cycle (Fowler et al. 1999; Kley et al. 1999). Meteorological conditions may exacerbate the rate of ozone formation too, particularly atmospheric inversion, a restricted air circulation associated to a warmer air layer above a cooler zone (Baumbach and Vogt 2003).

11.3 Chemistry of Ozone Injury

From the Greek *ozein* (to smell), ozone, the triatomic allotropic form of oxygen, is a colourless gas with a slightly sweet, water melon-like odour (odour threshold between 0.0076 and 0.036 ppm). Because of its strong oxidizing potential

(+ 2.07 eV), O_3 is a powerful oxidizing agent capable of reacting with virtually any biomacromolecule, including lipids, proteins, nucleic acids and carbohydrates, although it is neither a radical species nor a ROS (Mustafa 1990; Kelly et al. 1995). Ozone is considered too reactive to penetrate far into tissues, so that only a minor amount of the pollutant is believed to pass not reacted through a membrane, and nothing through a cell (Pryor 1992). Furthermore, its toxicity can be greatly enhanced by the spontaneous hydroxyl radical ($OH^{\cdot}$) generation in the aqueous solution, strongly accelerated by traces of Fe^{2+} and favoured at alkaline pH, though occurring even at physiological pH (Pryor 1994).

In cell membrane, polyunsaturated fatty acids represent the primary target for ozone, stimulating lipid peroxidation and impairing membrane fluidity. The chemistry of O_3-induced lipid peroxidation, known as Criegee ozonation pathway, involves ozonolysis of alkenes in polyunsaturated chains, i.e. the electrophilic O_3 addition across the carbon-carbon double bonds, to give the Criegee ozonide (Criegee 1957). Afterwards, ozonide decomposes, under suitable conditions, to form organic radicals, aldehydes and peroxides. In further steps, H_2O_2 can react with transition metals (Cu or Fe), according to Fenton or Haber-Weiss reactions, to form other ROS (Pryor et al. 1991; Pryor 1993).

As a result of the ozone-induced oxidation, modification of proteins also occurs, both in their structure and activity. The pollutant directly or through highly reactive free radical mediated reactions, oxidizes the amino acidic residues, mainly of tyrosine, tryptophan, cysteine, methionine and histidine (Mudd et al. 1969). In particular, it reacts with the exposed sulphydryl groups to form disulphides bridges, and with tryptophan to give protein ozonides, in turn generating protein hydroperoxides and hydrogen peroxide (Freeman and Mudd 1981). Tyrosine residues can be cross-linked too, after the oxidation of their HO- groups, to give O,O'-dityrosine (Ignatenko et al. 1984). DNA damage can be produced as well, as shown by the increased activity of poly (ADP-ribose) synthetase, a chromatin-bound enzyme promoting damaged DNA repair (Hussain et al. 1985).

11.4 Plant Secondary Metabolism

In living organisms, secondary metabolites are not essential for growth and development unlike the products of primary metabolism. However, this does not mean they are not necessary. Infact, secondary metabolites relate plants with the components of their ecosystem, that is to say the physic environment (biotope) and the living community (biocenosis), thus resulting indispensable for the survival of the species. Additionally, stress metabolism could be regarded as a particular expression of secondary metabolism, when stressful conditions, of both biotic and abiotic nature, change the dynamic equilibrium of the ecosystem. As an instance, phytoalexins are compounds synthesized ex novo or whose synthesis increases after pathogen challenging, raising their concentration in the tissues (Frank 1993; Paiva 2000).

In plants, chemical diversity has determined their evolutionary success. Because of their sessile habit, plants are unable to avoid the worsening environmental conditions as well as they cannot escape the plethora of the laying before biotic stresses. Consequently, unlike animals, plants have evolved an enormous number of secondary metabolites to overcome any danger. The functional role of these phytochemicals ranges from the ecology to defence, improving protection against both biotic and abiotic stresses, besides being involved in ecological roles as attractants or repellents for pollinators and phytophagy, respectively, and colours and scents of reproductive organs (flowers and fruits).

Generally, precursors of secondary metabolic pathways are products of the primary metabolism. Therefore, a severe or long-lasting stress factor could induce an excessive shift between primary and secondary metabolism and consequently, a diversion of essential available resources from growth to defence. To a large extent, secondary metabolites derive from three biosynthetic routes, namely the phenylpropanoid, isoprenoid and alkaloid pathways. Phytochemicals arising from these pathways include not only compounds with a broad-spectrum antibiotic activity, but also powerful antioxidants, able to efficiently scavenge ozone-induced ROS (Facchini 2001; Holstein and Hohl 2004; Iriti and Faoro 2004).

11.4.1 Phenylpropanoid Pathway

Phenylpropanoids are a class of phenylalanine derivatives with the basic skeletal C_6-C_3 (phenyl-propane) as depicted in Fig. 11.3.

Precursors of this pathway are phosphoenolpyruvate derived from glycolysis and erythrose 4-phosphate from pentose phosphate pathway, leading to two important intermediates, shikimic acid and chorismic acid. In further steps, after a branch point, phenylalanine and tyrosine are synthesized from prephenic and arogenic acid, whereas tryptophan comes from anthranilic acid (Weaver and Hermann 1997).

The removal of an amino group (deamination) from phenylalanine via the step catalyzed by the enzyme phenylalanine ammonia-lyase (PAL), leads to formation of cinnamic acid and, in turn, the precursor of hydroxycinnamates is produced after a series of hydroxylation of the benzene ring (Fig. 11.4). These compounds, including coumaric, ferulic and sinapic acids are reduced to the corresponding alcohol via aldehyde intermediates, namely coumaryl, coniferyl and sinapyl alcohol, collectively termed monolignols. Dimerization or polymerization of monolignols leads to formation of lignans and lignin, respectively (Fig. 11.4). Lignification is a complex reaction in which peroxidases catalyze the polymerization of lignin units consuming H_2O_2. Apposition of lignin in plant cell wall is a process occurring during the development of particular tissues, as well as in plant defence responses, in order to strengthen the cell wall and to protect the plasmalemma. Benzoic and hydroxybenzoic acids (C_6-C_1), such as salicylic acid (SA), a molecule involved in systemic acquired resistance (SAR), are another group of cinnamic acid derivatives,

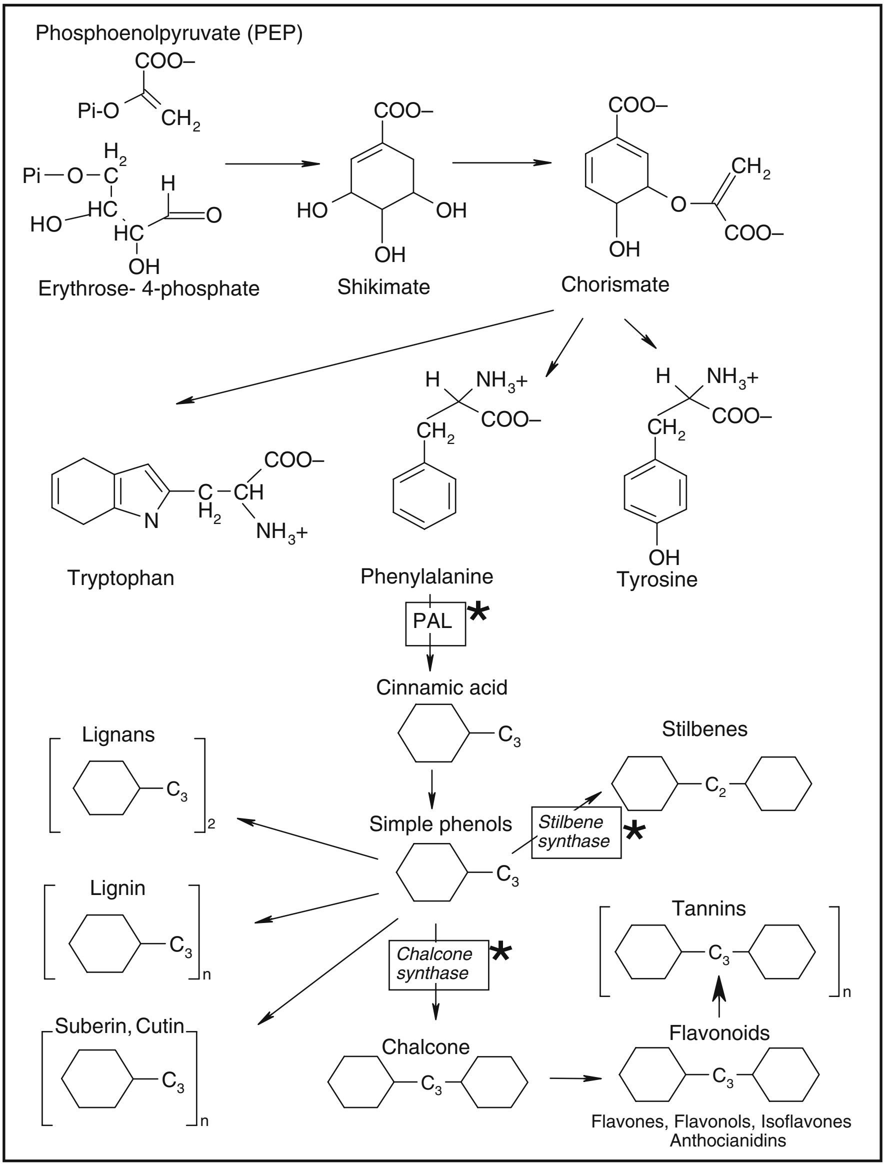


Fig. 11.3 Aromatic amino acid biosynthesis from shikimate and phenyl-alanine derivatives of the phenylpropanoid pathway. The enzymes with an asterisk are known to be influenced by ozone

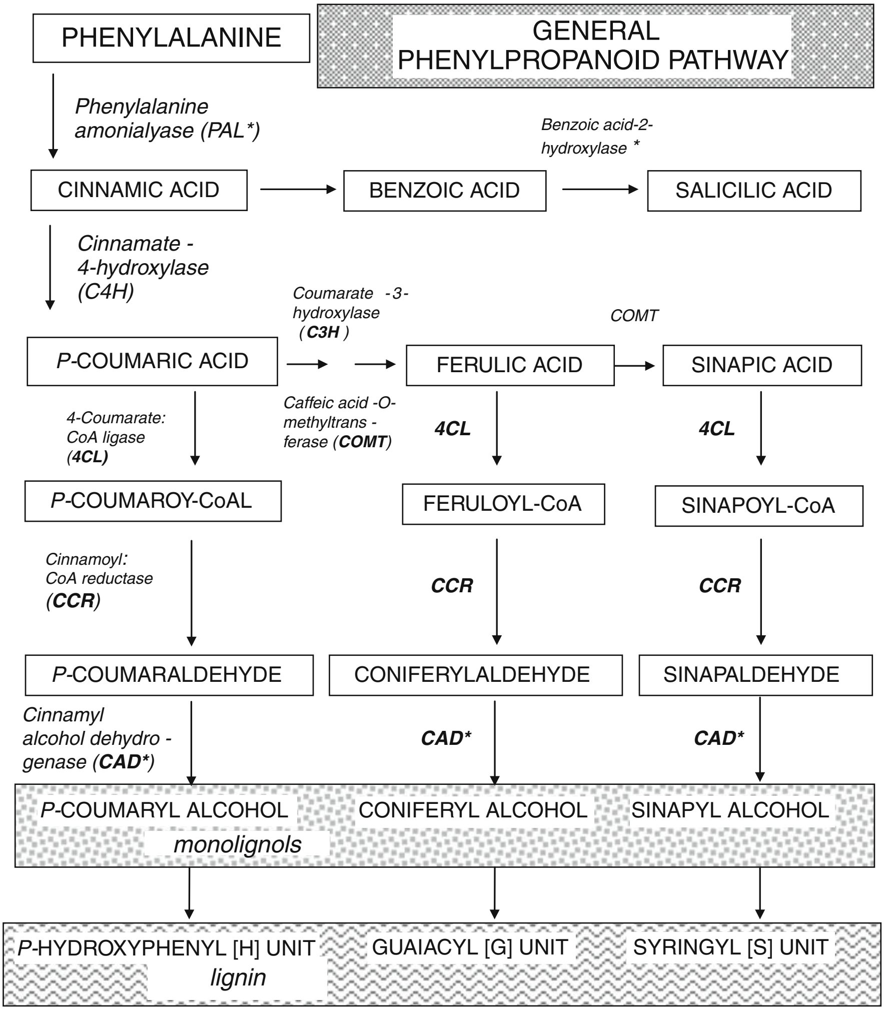


Fig. 11.4 Main steps of the phenylpropanoid pathway leading to formation of benzoic acids, hydroxycinnamates (coumaric, ferulic and sinapic acids) and lignin. The enzymes with an asterisk are known to be influenced by ozone

formed by cleavage of a C_2 fragment from the phenylpropane structure (Fig. 11.4) (Hahlbrock and Scheel 1989; Iriti and Faoro 2004).

The metabolites till now mentioned may be collectively named as simple phenols to differentiate them from polyphenolic compounds. The latter have an additional benzene ring, arising from the cyclization of three malonic acid (C_2) residues via stilbene sinthase (STS) or chalcone synthase (CHS), to give stilbenes (C_6-C_2-C_6) or flavonoids (C_6-C_3-C_6), respectively (Fig. 11.3). Finally, proanthocyanidins (PA), or condensed tannins include oligo- and polymeric flavonoid derivatives with polymerisation degree ranging from 2 to 17 and more. The numerous hydroxyl groups available in these molecules promote the formation of complexes with macromolecules, such as proteins, polysaccharides and metal ions (Hahlbrock and Scheel 1989; Iriti and Faoro 2004).

11.4.2 Isoprenoid Pathway

Isoprenoids, also named terpenoids, are the chemically and functionally most diversified class of low molecular mass lipids in plants, both primary and secondary metabolites (Iriti and Faoro 2006). They include electron carriers (quinones), membrane constituents (sterols), vitamins (A, D, E and K), plant hormones (side chain of cytokinins, abscisic acid, gibberellins and brassinosteroids), photosynthetic pigments (chlorophyll, phytol and carotenoids) and essential oils (Sacchettini and Poulter 1997).

Acetyl coenzyme A (CoA) represents the precursor for the isprenoid biosynthesis (Fig. 11.5). Firstly, two molecules of acetyl CoA react to give acetoacetyl CoA and then, with a further acetyl CoA to produce β-hydroxy-β-methylglutaryl-coenzime A (HMG-CoA). In plants, the same enzyme, HMG-CoA synthase, catalyses both reactions. The conversion of HMG-CoA into mevalonate, via HMG-CoA reductase, is the rate limiting enzyme of this pathway (Chappell 1995). Mevalonate kinase and mevalonate phosphate kinase phosphorylate, respectively, mevalonate and, then, mevalonate 5-phosphate, yielding mevalonate 5-diphosphate. Further, mevalonate diphosphate decarboxylation, via mevalonate diphosphate decarboxylase, produces isopentenyl diphosphate (IPP), the five-carbon building block for the formation of isoprenoid chains. The enzyme IPP:dimethylallyl-PP isomerase converts IPP into dimethylallyl diphosphate (DMAPP), the acceptor for successive transfers of isopentenyl residues (Bach 1987; Chappell 1995).

Hemiterpenes (C_5), such as isoprene, originate from dimethylallyl-PP, upon the release of diphosphate. Differently, dimethylallyl-PP can condense with IPP to form geranyl-PP via geranyl-PP synthase. In the same way, further chain elongation is attained by head to tail condensation of geranyl-PP to IPP, to produce farnesyl-PP via farnesyl-PP synthase. Analogously, geranylgeranyl-PP synthase catalyses the head to tail condensation of farnesyl-PP to IPP thus yielding geranylgeranyl-PP (Poulter and Rilling 1978).

Geranyl-PP is the precursor for the formation of monoterpenes (C_{10}) or essential oils, including highly volatile open chain and cyclic compounds, such as menthol, limonene, geraniol, linalool and pinene. They are active in plant-microbe, plant-pronubi, plant-phytophagous and plant-plant interactions due to their attractiveness and repulsiveness (Croteau 1987).

Farnesyl-PP is the precursor for the synthesis of open chain and cyclic sesquiterpenes (C_{15}), the largest group of isoprenoids, including essential oils and antibiotic compounds (phytoalexins) (Cane 1990). Diterpenes (C_{20}) derive from geranylgeranyl-PP consisting of phytoalexins, plant hormones, the phytol side chain of chlorophylls, tocopherols and phylloquinone (Dogbo and Camara 1987).

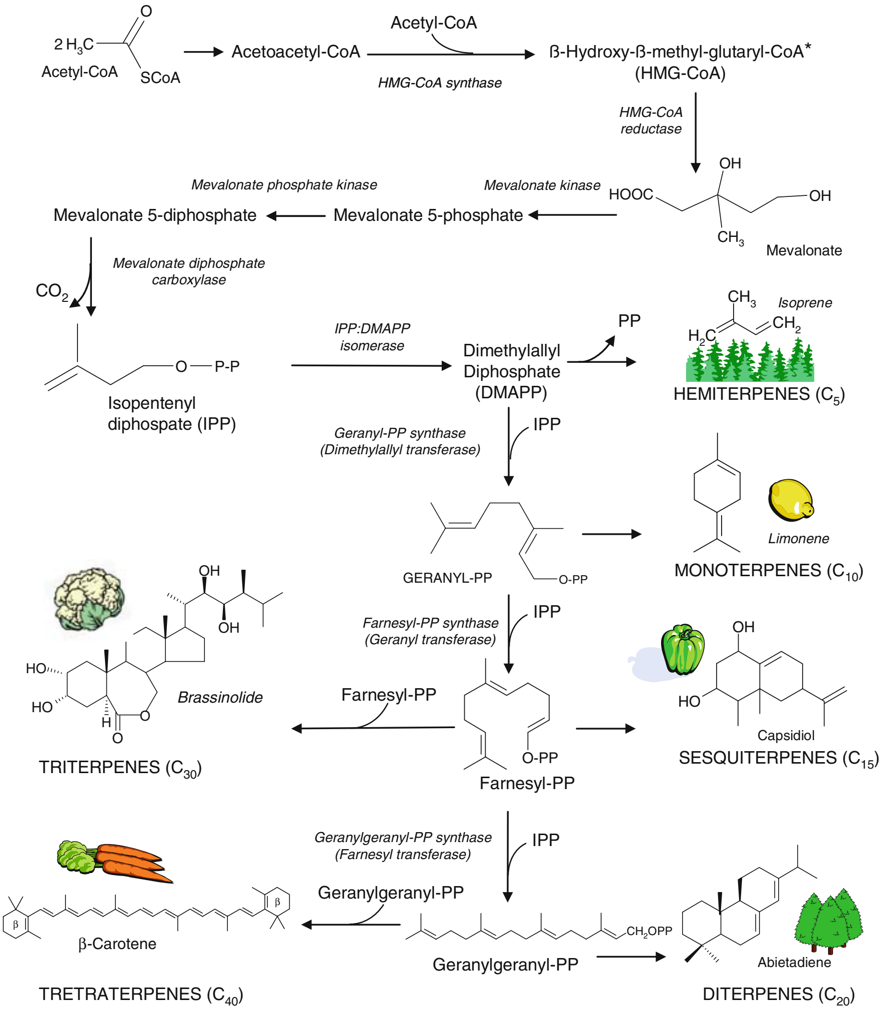
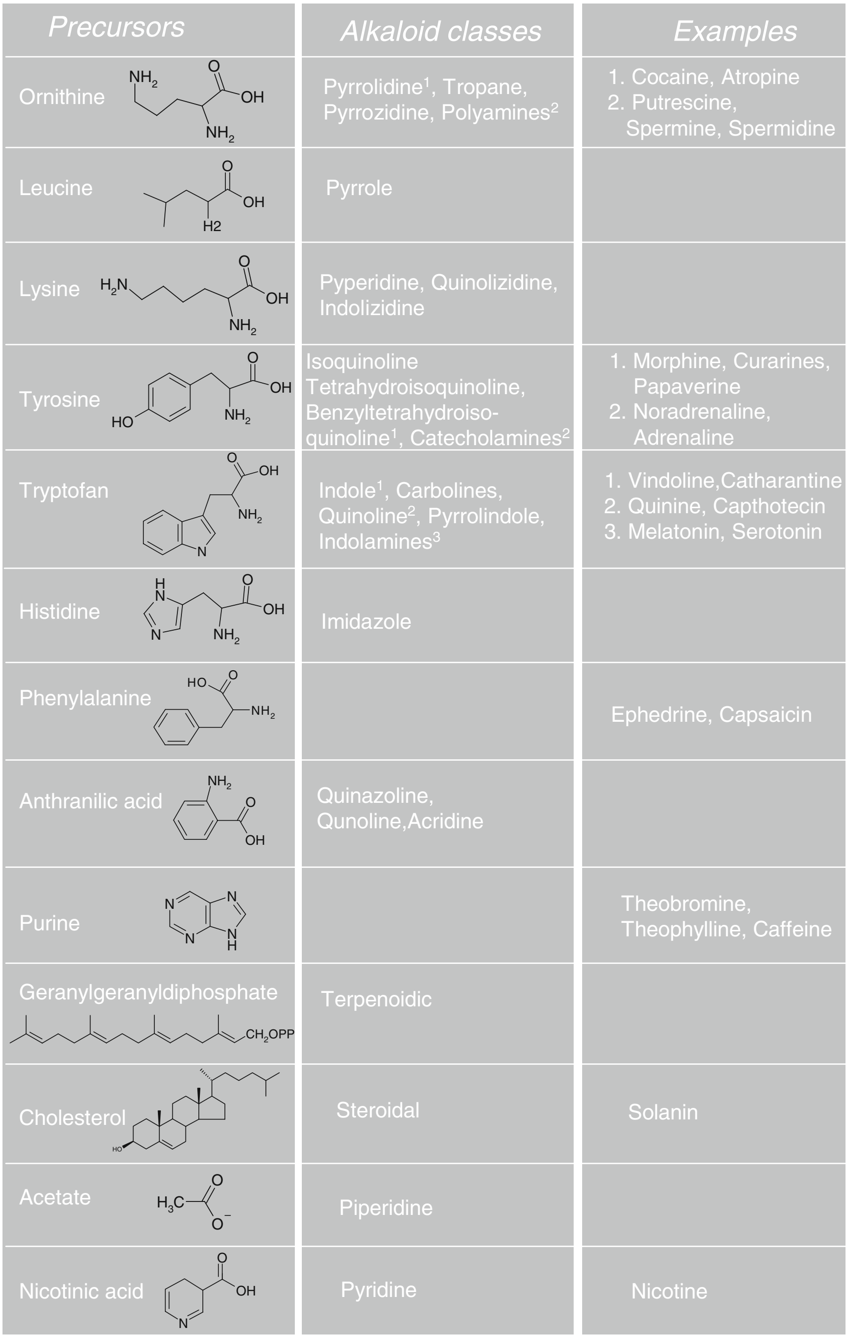


Fig. 11.5 Isoprenoid pathway from Acetyl-CoA. The enzyme with an asterisk is reported to be influenced by ozone

Furthermore, triterpenes (C_{30}) are synthesized from two molecules of farnesyl-PP (C_{15}), by a reductive head to head condensation. Squalene triterpene is the precursor for sterols, important membrane constituents, via squalene synthase (Abe et al. 1993). Analogously, head to head condensation of two molecules of geranylgeranyl-PP (C_{20}) leads to tetratepenes (C_{40}), such as carotenoids (carotene, lycopene) and xanthophylls (lutein, zeaxanthin, violaxanthin). Besides, isoprenoids are involved in protein prenylation, that is the synthesis of variously lengthened isoprenoid chains, anchoring proteins in membranes, such as G proteins, ubiquinone, plastoquinone and cytochrome-*a*. Finally, natural rubber is a polyterpen, composed



Precursors	Alkaloid classes	Examples
Ornithine	Pyrrolidine[1], Tropane, Pyrrozidine, Polyamines[2]	1. Cocaine, Atropine 2. Putrescine, Spermine, Spermidine
Leucine	Pyrrole	
Lysine	Pyperidine, Quinolizidine, Indolizidine	
Tyrosine	Isoquinoline Tetrahydroisoquinoline, Benzyltetrahydroisoquinoline[1], Catecholamines[2]	1. Morphine, Curarines, Papaverine 2. Noradrenaline, Adrenaline
Tryptofan	Indole[1], Carbolines, Quinoline[2], Pyrrolindole, Indolamines[3]	1. Vindoline,Catharantine 2. Quinine, Capthotecin 3. Melatonin, Serotonin
Histidine	Imidazole	
Phenylalanine		Ephedrine, Capsaicin
Anthranilic acid	Quinazoline, Qunoline,Acridine	
Purine		Theobromine, Theophylline, Caffeine
Geranylgeranyldiphosphate	Terpenoidic	
Cholesterol	Steroidal	Solanin
Acetate	Piperidine	
Nicotinic acid	Pyridine	Nicotine

Fig. 11.6 Main classes of alkaloid precursors and derivatives (the example number refers to the corresponding alkaloid class)

of over 1000 isoprene units and derived from polymerisation of geranylgeranyl-PP units (Wendt and Schulz 1998).

11.4.3 Alkaloid Pathways

A common pathway for alkaloid biosynthesis does not exist. The greater amounts of alkaloid compounds are amino acid derivatives, grouped on the base of their precursor and chemical structure. Therefore, the main groups include alkaloids arising from ornithine, leucine, lysine, tyrosine, tryptophan, histidine and phenylalanine, in addition to alkaloids arising from nicotinic (pyridine alkaloids) and anthranilic acid, acetate, isoprenoids and purine (Fig. 11.6). In turn, these classes include some minor divisions. Accordingly, pyrrole alkaloids arise from leucine; pyrrolidine, tropane and pyrrolizidin alkaloids from ornithine; piperidine, quinolizidine and indolizidine alkaloids derived from lysine; catecholamines, isoquinoline, tetrahydroisoquinoline and benzyltetrahydroisoquinoline alkaloids originated from tyrosine; indolamines, indole, carboline, quinoline, pyrrolindole, ergot alkaloids come from tryptophan and imidazole alkaloids from histidine. The same way, anthranilic acid is the precursor of quinazoline, quinoline and acridine alkaloids, whereas isoprenoid alkaloids include mono- (geraniol), di- (geranylgeranyl-PP) and triterpenes (cholesterol) derivatives (Fig. 11.6) (Cordell et al. 2001; Facchini 2001; Hughes and Shanks 2002).

Alkaloids consist of an enormous number of phytochemicals of toxicological, pharmacological, nutritional and cosmetic interest, and also of ecological importance for plants. For instance, tropane alkaloids include cocaine and atropine, nicotine is a pyridine alkaloid, noradrenaline (or norepinephrine), adrenaline (epinephrine), papaverine, curarines and morphine arise from tyrosine. Melatonin and serotonin are indolamines, vindoline, catharantine (and their derivatives vincristine and vinblastine) are indole alkaloids, quinine and capthotecin are quinoline alkaloids and lysergic acid diethylamide (LSD) is an ergot alkaloid, all these arising from tryptophan. Histamine is an imidazole alkaloid, ephedrine and capsaicin derive from phenylalanine, solanin is a steroid glycoalkaloid from cholesterol. Finally, purine alkaloids include theophylline, theobromine and caffeine which are found in tea, cacao and coffee, respectively (Cordell et al. 2001; Facchni 2001; Hughes and Shanks 2002).

11.5 Ozone and Phenylpropanoids

In plants, phenylpropanoid metabolism is induced as a general response to stress. Therefore, enhancement of key enzyme activities and accumulation of secondary metabolites are events that occur in order to improve the resistance against pathogen attack and/or tolerance to adverse environmental conditions and pollutants. PAL is an extremely sensitive indicator of stress conditions, and commonly considered as a biochemical marker indicating the activation of plant defences which include the synthesis of both structural and protective compounds (Fig. 11.7). In particular, ozone exposure elevates the level of flux through the phenylpropanoid pathway, thereby supplying carbon skeletons for secondary metabolites (Toumainen et al. 1996).

The enhancement of phenylpropanoid biosynthesis by ozone is well documented. A very early report, dated more than 30 years ago, demonstrated the accumulation of isoflavonoid in soybean following ozone exposure (Keen and Taylor 1975). Ever since, ozone-stimulated induction of transcripts for defence-related genes, the

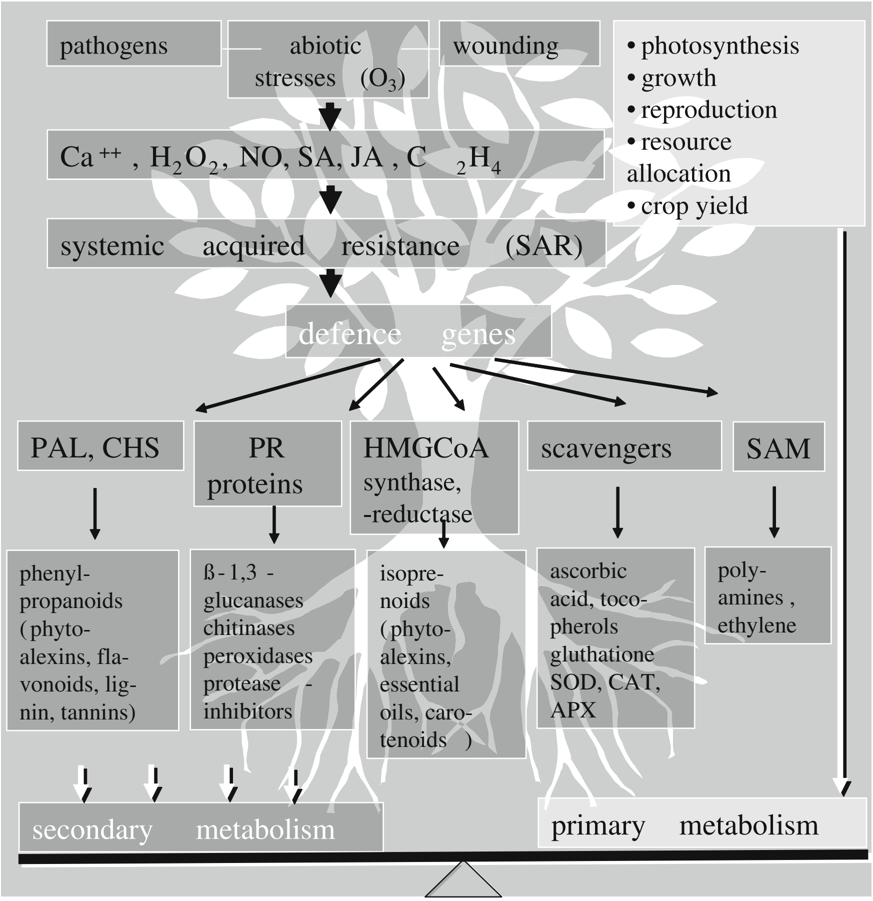


Fig. 11.7 Influence of different stresses on plant metabolism. The activation of systemic acquired resistance results in the accumulation of numerous defence compounds that may imbalance the equilibrium between primary and secondary metabolism, thus resulting in fitness costs for the plant. On the other end, the synthesis of secondary metabolites due to a stress may protect plant from other different stresses

same induced by pathogens, as well as the increased activity of key enzymes of the phenylpropanoid pathway have been reported in several plant systems (Figs. 11.3, 11.4, and 11.7). In Arabidopsis (*Arabidopsis thaliana*), PAL mRNA is rapidly and transiently induced within 3h of ozone treatment (300 nL L^{-1} daily for 6 h), reaching a 3-fold higher levels than control plants (Sharma and Davis 1994). A similar trend has been reported in parsley (*Petroselinum crispum*) plants, in which ozone treatment (200 nL L^{-1} for 10 h) induced an early 3-fold and 1.2-fold increase of PAL and CHS activity, respectively, followed by a 2-fold increase of total leaf furanocoumarins and flavone glycosides (Eckey-Kaltenbach et al. 1994). The content of psoralen, bergapten and other furanocoumarins in celery (*Apium graveolens*) dropped 24 h after ozone (0.2 nL L^{-1} for 2 h) fumigation, but levels of these chemicals, in treated leaves, increased rapidly at 120 h (Dercks et al. 1990). Further studies have shown an overlap between patterns of genes induced by ozone exposure and pathogen infection, probably due to the role of ROS as effector molecules involved in transduction pathways activated either by pathogens and ozone. The enhancement of SA content in plant tissues due to ozone treatment is well documented (Rao and Davis 2001). Intriguingly, in tobacco (*Nicotiana tabacum*) plants, a pulse ozone treatment (120–170 nL L^{-1} for 5 h) enhanced the emission of methyl salicylate, a volatile SA derivative, to a greater extent in sensitive cv. Bel W3 compared to tolerant cv. Bel B (Heiden et al. 1999). In Arabidopsis, SA accumulation, necessary for the expression of hypersensitive response to pathogens (HR) and SAR, is also required for the accumulation of some ozone-induced mRNAs, particularly PAL and pathogenesis related protein 1 (PR1) transcripts, although a SA-independent signal transduction pathway is activated by the pollutant (Sharma et al. 1996). In fact, in transformed *Arabidopsis* plants unable to accumulate these transcripts, ozone exposure induces resistance against the bacterial pathogen *Pseudomonas syringe*, a phenomenon known as cross induction (Sharma et al. 1996). HR is a type of programmed cell death (PCD) triggered at the attempted pathogen penetration site, frequently at the onset of systemic immunity (SAR) (Langebartels et al. 2002; Iriti and Faoro 2007), whereas PR proteins are enzymes induced in plants by pathogen infection as well as by abiotic and environmental stresses including ozone (Schraudner et al. 1992; Thalmair et al. 1996; Pääkkönen et al. 1998). In sensitive bean (*Phaseolus vulgaris*) cv. Pinto, ozone exposure (120 nL L^{-1} for 4 h) causes a stimulation of phenylpropanoid route and flavonoid branch, as shown by the increased mRNA accumulation of PAL, CHS and chalcone isomerase (CHI), the latter involved in isoflavonoid biosynthesis (Paolacci et al. 2001). In grapevine (*Vitis vinifera*), STS, the first enzyme of the stilbene branch involved in the synthesis of resveratrol and other stilbenic compounds, is considered the most sensitive ozone-induced biomarker (Schubert et al. 1997). In grapevine callus, either PAL or STS activity increase, after ozone fumigation (0.3 μmol mol^{-1} for 2 h) unlike CHS (Sgarbi et al. 2003). This could be due to the STS and CHS competition for the same substrate (Fig. 11.3), during the véraison (the berry ripening), when a metabolic switch occurs between the two branches of the same pathways. At this phenological stage, declined STS activity and resveratrol concentration in berries may cause grey mould (*Botrytis cinerea*) infection, whereas enhanced CHS activity and anthocyanin accumulation are required for berry colouring (Jeandet et al. 1995). Furthermore, a general drop in the amount of some assayed phenylpropanoids (coumaric acid, ferulic acid, gallic acid and catechin) has been reported unlike caffeic acid, whose level raised only in one cell line (Sgarbi et al. 2003). In 20 soybean (*Glyicine max*) cultivars, ozone tolerance was associated with the presence of kaempferol glycosides, a powerful antioxidant flavonol, as well as tolerance to manganese (Mn) toxicity in one soybean line (Foy et al. 1995). In European silver birch (*Betula pendula*) chronically exposed to ozone, it has been reported a 16.2% increase in total phenylpropanoids and a corresponding 9.9% increase of 10 compounds, among simple phenols and flavonoids, such as chlorogenic acid and catechin, respectively (Saleem et al. 2001). Interestingly, the combined action of CO_2 and ozone greatly enhances the synthesis of total and polymeric PA, in *Betula* sp. leaf tissues, suggesting an additive effect of these environmental pollutants on phenylpropanoid biosynthesis (Karonen et al. 2006).

Shikimate dehydrogenase (SKDH), a key enzyme of shikimate pathway, PAL and cinnamyl alcohol dehydrogenase (CAD), a key enzyme of lignin biosynthesis which forms monolignols (Fig. 11.4), have been investigated in poplar (*Populus tremula x alba*) leaves. Under ozone exposure (60–120 $\text{nL}\,\text{L}^{-1}$, during the 14 h light period, for 1 month), either CAD activity and transcript levels were rapidly and strongly stimulated, increasing up to 15-fold and 23-fold the control values, respectively. In contrast, SKDH and PAL activities raised only in old and middle-aged leaves, but not in the youngest ones. Interestingly, the increased activity of these enzymes was associated with a higher lignin content in ozone-exposed leaves and additionally, the newly synthesized lignin structurally differed from the control lignin. Particularly, stress lignin appeared more condensed, i.e. enriched in carbon-carbon interunit linkages, in *p*-hydroxyphenyl (H) units and in terminal units with free phenolic groups (Fig. 11.4) (Cabané et al. 2004). The same enzymes have been studied in two genotypes of ozone-treated (150 $\text{nL}\,\text{L}^{-1}$ for 3 h) tomato (*Lycopersicon esculentum*) plants (Guidi et al. 2005). However, while SKDH and PAL activity augmented significantly only in one line, CAD activity diminished in both the genotypes in contrast to the results reported by other authors (Galliano et al. 1993; Cabané et al. 2004). An explanation could reside in the different response of herbaceous and woody plants and in the different acute or chronic ozone dose employed in tomato and poplar, respectively.

To conclude, the importance of phenylpropanoids in plant tolerance against ozone injury is related to their different properties (Iriti and Faoro 2004). These compounds include an array of molecules with a plethora of biological activities, besides being precursors of structural biopolymer, such as lignin. Particularly, their protective role is mainly inherent to their antioxidant power, i.e. the ability to trap free radicals (ROS) functioning as electron donors. Nonetheless, increased synthesis of lignin, as well as structural modifications of lignin itself, represent another important defence mechanism in order to protect plasmalemma from the ROS injury, thus preventing membrane damages due to lipid peroxidation.

11.6 Ozone and Isoprenoids

Strictly speaking, not all the isoprenoids are secondary metabolites, as some primary metabolites, such as sterols, arise from isoprenoid pathway as well. The effects of the pollutant on either the sterol concentration and composition have been early reported in several plants (Tomlinson and Rich 1971; Trevathan et al. 1979; Grunwald and Endress 1985). Sterols are important component of cell membranes, involved in their stabilization. Generally, exposure to high ozone concentrations ($>12\,\mu\text{L}\,\text{L}^{-1}$) results in a decrease of free sterols (FS) and an increase in bound sterols (BS). Conversely, at low ozone concentration ($<12\,\mu\text{L}\,\text{L}^{-1}$), an accumulation of FS occurs, resulting in a decrease of the FS:BS ratio. In this case, the enhancement of FS synthesis is followed by a shift towards sterols with a more bulky C-17 side chain, i.e. sitosterol and stigmasterol vs. campesterol. As FS have a much greater capacity to stabilize membranes than BS, the FS:BS ratio has a more determinant effect on membrane permeability than the composition of FS fraction itself. Accordingly, a decrease in FS:BS ratio with high ozone concentration results in cell injury and visible damages, whereas modification of FS composition, occurring at lower ozone levels, results neither in permanent membrane injury nor in visible damages (Grunwald 1971; Evans and Ting 1973; Grunwald 1974). In tobacco leaves, ozone fumigation enhances total lipid concentrations, but it decreases levels of FS and triglycerides (Trevathan et al. 1979), whereas a higher amount of total phytosterols was reported in fumigated plants by other researchers (Menser et al. 1977).

In ozone treated ($300\,\text{nL}\,\text{L}^{-1}$ for 8 h) Scots pine (*Pinus sylvestris*), it was reported a transient increase of a transcript corresponding to the cytosolic/endoplasmic ß-Hydroxy-ß-methyl-glutaryl-CoA (HMG-CoA) synthase, a key enzyme of isoprenoid biosynthesis (Wegener et al. 1997). The C_5 precursor of isoprenoids, IPP, arises either from mevalonate pathway, in the cytosol, as previously described, or from plastidial precursors (Cheng et al. 2007). In ozone-treated pine seedlings ($250\,\text{nL}\,\text{L}^{-1}$, 12 h day^{-1} for 4 days), the biosynthesis of plastidial IPP was inhibited, differently from the mevalonate synthesis in the cytosol, due to the resource allocation between the two IPP synthesis pathways (Shamay et al. 2001).

Biogenic Volatile Organic Compounds (BVOC) comprise mainly isoprenoids (particularly hemi-, mono- and sesquiterpenes) emitted from plants during cell growth and in response to several kinds of stresses, they are having various ecophysiological functions and mediating plant-arthropod interactions (Fig. 11.8) (Kesselmeier and Staudt 1999). BVOC can act as attractants for pollinators or repellents for noxious insects, besides being involved in tritrophic signalling, i.e. the relationship between plant, herbivorous and carnivorous arthropods. In particular, phytophagus feeding can induce BVOC emission from plants, which can function as foraging cues for the recruitment of the natural enemies of herbivores (Dicke 2000). Besides wounding and trithrophic interactions, several environmental factors can affect the BVOC emission from plants, such as light intensity, temperature, water supply and pollutants (Fig. 11.8) (Peñuelas and Llusià 2001).

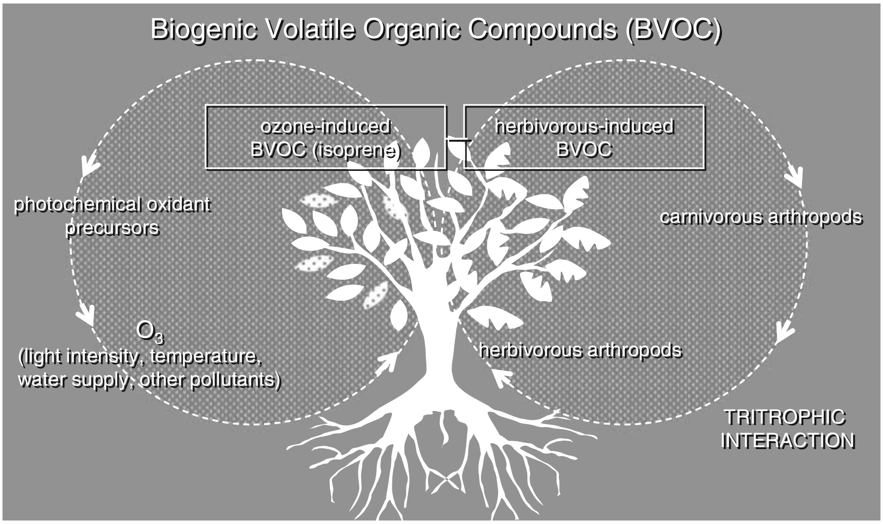


Fig. 11.8 Relationships between ozone and herbivorous-induced BVOCs and their influence on photochemical ozone precursors and on tritrophic interactions, respectively

The relationships between ozone and BVOC is somewhat complex, both exerting a mutual influence (Llusià et al. 2002). On one hand, the release of either BVOC or volatile organic compounds (VOC) from anthropogenic origin into the atmosphere can constitute a significant input of photochemical oxidant precursors, thus contributing to the regional-scale air pollution, on the other, BVOC emission can be triggered by the exposure to high ozone concentration (Roselle 1994; Heiden et al. 1999). Additionally, chronic ozone exposure can modify the composition of the plant BVOC emissions, thus not only affecting the tritrophic interactions, but also directly weakening plant defence responses against arthropods (Alstad et al. 1982). Isoprenoids can be synthesized and emitted in order to tolerate the ozone injury. Isoprene, a hemiterpene (Fig. 11.5), has been reported to reduce ozone damages in leaves, because of its antioxidant activity. This gas protects the photosynthetic apparatus, quenches ozone byproducts and radical species responsible for lipid peroxidation of cell membranes and cell death (Loreto and Velikova 2001; Loreto et al. 2001; Velikova et al. 2005). Besides, monoterpenes may exert an isoprene-like antioxidant activity, too (Loreto et al. 2004).

11.7 Ozone and Alkaloids

The effect of ozone on alkaloid biosynthetic pathways has not been extensively investigated, although a generalized influence of the pollutant on the nitrogen metabolism has been ascertained (Menser and Chaplin 1975; Aycock 1975; Jackson et al. 2000). Generally, ozone exposure reduces the amount of total alkaloids in tobacco plants (Menser and Chaplin 1969; Aycock 1975), and lower levels of nicotine, a pyridine alkaloid, in ozone exposed plants were related to increased survival, growth and development of hornworm (*Manduca sexta*) larvae (Jackson et al. 2000). Several studies reported increased preferences or enhanced survival and fitness of different insect species on ozonated plants, thus pointing out again the detrimental effect of the pollutant on the plant chemical defences (Jeffords and Endress 1984; Endress and Post 1985; Trumble et al. 1987; Chappelka et al. 1988; Heagle et al. 1994).

The role of polyamines, important alkaloid precursors (Fig. 11.9), has been correlated with ozone tolerance. Polyamines are polycationic nitrogenous compounds of low molecular weight ubiquitous in all living organisms. In plants, they function as growth regulators involved in an array of physiological processes, being involved in embryogenesis, cell division, morphogenesis, development, flowering and senescence (Martin-Tanguy 2001). In addition, they serve as an integral component of plant response to both biotic and abiotic stresses (Walters 2000;

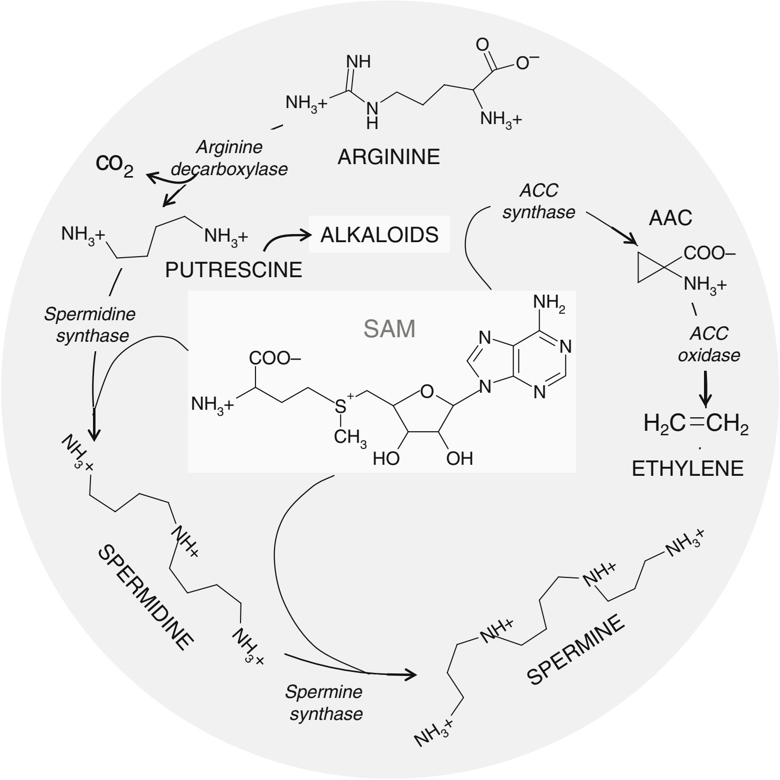


Fig. 11.9 Involvement of S-adenosylmethionine (SAM) in polyamine and ethylene biosynthesis. The metabolic shift to ethylene or polyamine biosynthesis can enhance ozone susceptibility or tolerance, respectively

Navakoudis et al. 2003; Liu et al. 2007). The most important polyamines are the diamine putrescine (Put), the triamine spermidine (Spd) and the tetraamine spermine (Spm), arising directly from the free non-proteinogenic amino acid ornithine, by ornithine decarboxylase (ODC), or indirectly from arginine by arginine decarboxylase (ADC). Further steps include Put conversion into Spd, via spermidine synthase (SPDS), and the similar synthesis of Spm from Spd via spermine synthase (SPMS). Either of two enzymes employed the aminopropyl moiety provided by S-adenosylmethionine (SAM) (Bagni and Tassoni 2001). In plants, ADC activity is enhanced by ozone exposure, whereas ODC remains unchanged (Rowland-Bamford et al. 1989; Langebartels et al. 1991). Free and conjugated polyamines improve ozone tolerance with two different mechanisms: (i) by inhibiting the ethylene biosynthesis and (ii) by direct ROS scavenging (Ormrod and Beckerson 1986; Bors et al. 1989). Ethylene and polyamines share the same biosynthetic precursor, SAM and thereby, they mutually inhibit their own biosynthesis. In particular, aminocyclopropane carboxylic acid (ACC), a precursor of ethylene, arises from SAM via ACC synthase, a rate-limiting step in ethylene production. Therefore, this metabolic shift to ethylene or polyamine biosynthesis can enhance ozone susceptibility or tolerance, respectively, due to the correlation between the stress ethylene production and visible ozone injury (Langebartels et al. 1991).

Additionally, apoplastic polyamines can form conjugates with hydroxycinnamates and phenolic acid derivatives, effective in ROS detoxification (Bouchereau et al. 1999). As reported above, polyamine are precursors of alkaloids and apart from it, they play a role in ozone tolerance. Higher amounts of these compounds, as a consequence of ozone exposure, could induce an increase of pyrrolidine, tropane and pyrrolizidin alkaloids, deriving from ornithine via Put.

11.8 Conclusion

In the global climate change scenario, the relation between the anthropogenic emissions of greenhouse gases, responsible for climate warming and rainfall alteration, and troposheric ozone is quite complex, specially in the agroecosystem context. Crop potential yield depends on defining factors (CO_2, radiation, temperature and crop traits), limiting factors (water, nitrogen and phosphorous) and reducing factors (pests, pathogens, weeds and pollutants) (Goudrian and Zadoks 1995). A deep knowledge of the role played by these factors under altered climatic conditions is necessary to evaluate the responses at crop and agroecosystem level (Fuhrer 2003).

It is well known that increasing troposheric ozone concentration at ambient CO_2, causes a decline in the yield of many crop species, and this negative effect is reduced in a CO_2-enriched atmosphere, probably due to the decrease of stomatal conductance and ozone flux, or to the increase in the activity of anti-oxidant enzymes (Heagle et al. 1998, 1999, 2000). Thus, the detrimental effect of enhanced CO_2 concentration on Heart temperature is in some way, compensate by its protective role in plants under ozone stress, as assessed by biomass and yield stimulation studies in these conditions (McKee et al. 1995, 1997).

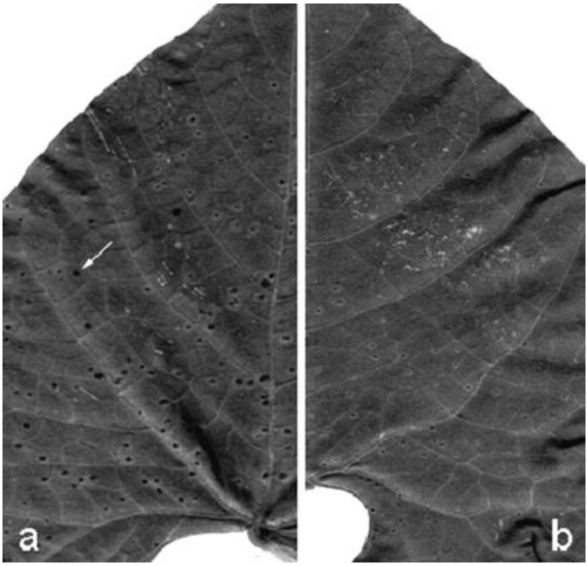


Fig. 11.10 Protective effect of ozone against *Uromyces appendiculatus*, causal agent of bean rust (Iriti and Faoro, unpublished). The leaf in (**b**) has been fumigated with ozone, 48 h before inoculation, and does not show any disease symptoms, while the control leaf in (**a**), only inoculated with the fungus, has developed typical rust pustules (*arrow*)

The influence of global climate change in plant-pathogen interactions is quite complex too (Violini 1995; Manning and von Tiedemann 1995). On one hand, ozone may induce the same sequel of events involved in plant immunity (Iriti and Faoro 2007) i.e. oxidative burst and hypersensitive response at the onset of systemic acquired resistance (SAR) (Fig. 11.10) (Langebartels et al. 2002), on the other, plants weakened by ozone stress may be particularly susceptible to infections (Manning and von Tiedemann 1995). Interestingly, wheat rust (*Puccinia recondita* f. sp. *tritici*) is strongly inhibited by ozone, but unaffected by elevated CO_2, both in presence or absence of ozone stress (von Tiedemann and Firsching 2000). Vice versa, a protective effect of rust (*Uromyces fabae*) infection in broad bean (*Vicia faba*) was reported against ozone, sulphur dioxide either alone or combined (Lorenzini et al. 1994).

Finally, regardless of the specific ecophysiological meaning, plants cope with the plethora of stressful abiotic and biotic conditions by modifying their secondary metabolic pathways. In this view, during the evolution, the chemical diversity improved the fitness of plant organisms, thus ensuring their evolutionary radiation. Phytochemicals, with their broad spectrum activities, are primarily involved in plant tolerance against environmental pollutants and worsening climatic conditions, as well as in resistance against pests and pathogens. The metabolic processes activated in these defence responses may be tightly separated or overlapping, according to the stress factor and the plant cultivar, as a result of negative (trade off) or positive (cross resistance) cross talk, that is to say the communication between molecular signals and transduction pathways involved in different plant defence responses. The most important consequence of this cross talk is the alert of defence mechanisms against an abiotic stress in consequence of priming with a biotic elicitor and vice versa.

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Chapter 12
Crop Responses to Enhanced UV-B Radiation

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12.1 Introduction

The Earth's climate has always been changing. Recent observations indicate that the changes are occurring faster and greater than those in the past. Consequently, the main scientific concern is with the ability of organisms to cope with these changes, especially terrestrial plants and crops that are sessile.

Stratospheric ozone depletion, resulting in increased intensity of UV-B radiation at the Earth's surface, has been one of the most evident changes since the 1980s. The effects of enhanced UV-B radiation on plants have been widely studied, showing that UV-B radiation can have deleterious effect on plants i.e. damage to DNA, alterations in transpiration, photosynthesis and respiration potential, growth, development and morphology (Rozema et al. 1997; Björn 1999; Gaberščik et al. 2002; Jansen 2002). The intensity of the effects depends on the species and on the balance between potential damage and the induction of protective mechanisms (Stephanou and Manetas 1998; Gaberščik et al. 2001). Consequently, natural systems can be affected by changes in plant phenology, biomass, seed production, plant consumption by herbivores, diseases and changes in species composition (Caldwell et al. 1998). Crops have been shown to be even more sensitive, because they have been subjected to long-term breeding (Breznik et al. 2005).

New agricultural practices are being developed to adapt food production to recent climate changes, since UV-B radiation could eliminate crops that are less tolerant. Thus, it is important to estimate the tolerance of certain species and to improve their tolerance by plant breeding and by application of protective substances on the field, such as soaking of seeds or foliar spraying by selenium (Se) compounds.

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12.2 Changes of UV-B Radiation at the Earth's Surface

Depletion of the stratospheric ozone layer is the consequence mainly of huge emissions of chlorofluorocarbons (CFCs), methyl bromide (CH_3Br) and nitrogen oxides (NO_x) arising from human activities. Continuous observation since the 1980s has shown this layer to have decreased by 3–6%, resulting in 6–14% increase of UV-B radiation at Earth's surface (WMO 2003). The recovery of the ozone layer has been predicted either to speed up (Dyominov and Zadorozhny 2005) or to slow down, the uncertainty arising from the fact that the possible interactions with global warming are poorly understood. The recent, generally accepted forecast is that the stratospheric ozone layer will recover by 2050, but other possible scenarios are also being studied. In any case, the enhanced UV-B radiation will still be an issue and could significantly affect biocenosis (McKenzie et al. 2003). Climate change can also affect UV radiation due to changes in cloud cover and reflectivity, since temperature changes over the 21st century are likely to be about 5 times greater than in the previous century. Ozone is expected to increase slowly over the coming decades, but it is not known whether it will return to levels that are higher or lower than those present prior to the onset of ozone depletion in the 1970s.

The intensity of UV-B radiation at the Earth's surface depends also on other atmospheric conditions, i.e. clouds, aerosols, tropospheric ozone and other gases, season and location (Nielsen 1996). The stratospheric ozone depletion in high latitudes of the Northern Hemisphere (Jones and Shanklin 1995) is less pronounced and more erratic than in the Southern Hemisphere (Dahlback 2002).

Knowledge of the level of biologically effective UV-B (UV-B_{BE}) radiation is essential for interpreting the effects of UV-B radiation on organisms. Different action spectra have been obtained for different processes in organisms (Björn 1999). The increasing shorter wavelength UV-B requires that action spectra are determined for interpretation of UV-B effects. The Thimijan curve of biological effectiveness is the basis for developing action spectra that express the relative influence of a particular wavelength on the biological response (Thimijan et al. 1978).

12.3 The Effects of Enhanced UV-B Radiation on Crops

12.3.1 The Cell and Its Components

The most sensitive cell component is DNA because of its absorption in the shortwave region of solar radiation (Ravanat et al. 2001) and its importance for the organisms. The formation of cyclobutane pyrimidine and pyrimidine-(6,4)-pyrimidone dimers, that affect DNA duplication and gene transcription, has been shown in tobacco and *Arabidopsis* (Jansen et al. 1998; Bray and West 2005).

The structure, and hence function, of cell membranes is disrupted by radiation (Yuan et al. 2000a,b), mainly as a result of lipid peroxidation (Murphy 1990).

12.3.2 Photosynthesis

The effect of UV-B radiation on photosynthesis depends on species, cultivars, UV-B intensity and PAR:UV-B ratio (Kakani et al. 2003). However, enhanced UV-B radiation undoubtedly affects photosynthesis in most crops. Reduced net photosynthesis can result from disturbed PSII, reduced content and activity of Calvin cycle enzymes (Rubisco, PEPCase), down-regulated transcription of photosynthetic genes, decreased thylakoid integrity and altered chloroplast structures, but the primary target of the photosynthetic apparatus is not clear. The photochemistry of PSII has been shown to be disrupted in two buckwheat species (Breznik 2007, unpublished), in line with previous studies on pea, rice and spinach (Iwanzik et al. 1983; Renger et al. 1989; He et al. 1993). Although altered photochemistry of PSII has been ascribed to photoinhibition, it could also be a consequence of disturbed thylakoid membrane integrity caused by peroxidation of lipids and proteins (Sharma et al. 1998). On the other hand, the enhanced UV-B radiation may inhibit photosynthesis without affecting PSII photochemistry. Sharma et al. (1998) reported 66% decrease in net photosynthesis and only 16% decrease in Fv/Fm for wheat exposed to enhanced UV-B radiation, indicating a metabolic limitation of photosynthesis due to degradation/inactivation of Calvin cycle enzymes rather than disruption of PSII photochemistry. Disruption of the Calvin cycle because of altered content and inactivation of Rubisco and PEPCase has been confirmed in maize (Correia et al. 2005).

The effect on photosynthetic pigments, that must impact on photosynthesis, has been related to different UV bands and different species in studies on *Cucumis sativus* and *Glycine max*. Under disproportionate UV/PAR light conditions, photosynthetic pigment content was decreased by UV-A as well as by UV-B; in cucumber seedlings, the ratio of total carotenoids to total chlorophyll was increased (Yao et al. 2006a). Kakani et al. (2003) stated that a decrease in photosynthetic pigment content, following exposure to UV-B radiation in most crop species, is mainly the result of breakdown of chloroplast structure.

Responses of superhigh-yield hybrid rice *Liangyoupeijiu* to enhanced ultraviolet-B radiation showed that the chlorophyll content of UV-B exposed leaves was slightly reduced at the tillering stage, but recovered in new growth tissues (Xu and Qiu 2007). The light-saturated photosynthetic activity of UV-B exposed plants was, however, 45.2 and 35.3% higher than their controls.

12.3.3 Transpiration and Stomatal Conductance

Early studies on the effect of enhanced UV-B radiation on stomatal conductance and transpiration rate did not show any significant changes (Teramura et al. 1984). In contrast, recent findings revealed increasing or decreasing stomatal conductance as a result of UV-B radiation (Pal et al. 1999; Jansen and van-den-Noort 2000) that were in line with changes of transpiration rate (Correia et al. 1999). Nogués et al. (1998) demonstrated 65% decrease in adaxial stomatal conductance for pea. Our study on common and tartary buckwheat indicated a progressive decrease and about 50% lower transpiration rate at the end of the growth period (Breznik 2007, unpublished). A decrease was also found by Gaberščik et al. (2002) in common buckwheat, where transpiration rate was about 15% lower at the end of the season, and by Gitz III et al. (2005), 50% lower transpiration rate in soybean cultivar. Altered transpiration rate is ascribed to changes of stomatal function (Jansen and van-den-Noort 2000) and frequency (Gitz III et al. 2005). The mechanism of action of UV-B radiation on stomatal function has not been elucidated, but Yang et al. (2000) suggested that changes in ABA content and K^+ outflow from guard cells could be the reason.

12.3.4 Respiration

A few studies have been carried out on UV-B effects on cell respiration. Bassman et al. (2003) reported that crops responded to enhanced UV-B radiation with either increasing cell respiration or with no change. Germ et al. (2004) claimed that an increase in cell respiration is related to the greater need for energy to protect against UV-B and for repair of UV-B damage. Sharma et al. (1998) reported that enhanced UV-B radiation might induce gene expression for the enzymes cytochrome oxidase, manganese dismutase and peptidase, all of which are involved in cell respiration.

On the other hand, some recent findings revealed depressed cell respiration resulting from enhanced UV-B radiation. Studies of respiratory potential for common buckwheat, tartary buckwheat and pumpkin under irradiation indicated lower terminal electron transport (ETS) activity and thus lower respiration (Germ et al. 2005; Breznik 2007, unpublished). The reason for the inhibitory effect of enhanced UV-B radiation on ETS activity is not clear. However, Lukaszek and Poskuta (1996) provided an evidence for damage of mitochondrial structures that might result in decreased respiration.

12.3.5 Development and Habitus

Only a few studies have been conducted on crop phenology under enhanced UV-B radiation which showed conflicting results (Santos et al. 1998). Delayed germination and flowering were found in bean (Saile-Mark and Tevini 1997; Li et al. 1998). In contrast, Al-Oudat et al. (1998) reported an approximately one week earlier flowering time for wheat and bean. Yao et al. (2006b) showed shortened flowering time for tartary buckwheat, resulting in earlier grain ripening. It has been suggested that enhanced UV-B radiation could change the morphology of reproductive organs, even though they are mostly well protected against dangerous UV-B radiation by UV-B absorbing compounds (Caldwell et al. 1998).

The habitus of crops is usually also affected by enhanced UV-B radiation. Numerous crops, such as tartary buckwheat, common buckwheat, soybean, rice and cotton, were shorter when treated by UV-B radiation (Feng et al. 2003; Gao et al. 2003; Breznik et al. 2004, 2005) as a consequence of an effect during the early developmental stage of crops (Gao et al. 2003). It is likely that a disturbed elongation process results in plant stunting (Ballaré et al. 1996), but the mechanism was not worked out. According to recent findings, photooxidative destruction of the phytohormone, indole acetic acid (IAA), and thus a loss of apical dominance, could contribute to interruptions in the elongation process (Mark and Tevini 1997; Meijkamp et al. 2001). In line with loss of apical dominance, many crops, including bean (Meijkamp et al. 2001) and *Lolium* sp. (Deckmyn and Impens 1999), exhibited increased tillering. On the other hand, our study revealed reduced tillering for common and tartary buckwheat, despite plant being stunted (Breznik et al. 2004; Breznik 2007, unpublished).

Leaf area index (LAI) decreased because of the reduced leaf area in many crops, e.g. cotton (Gao et al. 2003), common and tartary buckwheat (Breznik et al. 2004, 2005; Breznik 2007, unpublished).

Increased leaf thickness is a common response of plants to enhanced UV-B radiation. Thicker leaf tissue successfully diminishes penetration of UV-B radiation (Ballaré et al. 1996).

12.3.6 Production and Yield

The numerous studies of UV-B effects on crop production have revealed no general trend, because different varieties of the same crops often react differently to elevated UV-B radiation. This might be attributed to differences in other environmental factors occurred during field experiments (Caldwell et al. 1998). Despite this, it is well known that enhanced UV-B radiation frequently affects biomass distribution and reproduction of most crops (Gao et al. 2003). Recent studies on common buckwheat, tartary buckwheat, pea plants, wheat, bush bean, rice, soybean and cotton, showed decreases in total biomass production of 10–50% (Barnes et al. 1993; Saile-Mark and Tevini 1997; Nogués et al. 1998; Correia et al. 1999; Gao et al. 2003; Breznik et al. 2004, 2005; Breznik 2007, unpublished). Lower total biomass production may reflect lower shoot or root biomass accumulation, as was found for wheat, common and tartary buckwheat (Correia et al. 1999; Breznik et al. 2004, 2005; Breznik 2007, unpublished). However, no alteration in the root-to-shoot ratio was found, which indicated that biomass allocation was not affected (Nogués et al. 1998; Correia et al. 1999; Gaberščik et al. 2002; Zheng et al. 2003; Breznik 2007, unpublished). An alteration in biomass production evidenced complex changes in morphogenetic and physiological processes that included reduced enzyme activities, lower efficiency of PSII and stomatal conductance (Saile-Mark and Tevini 1997), disturbance in water economy and water deficiency (Larcher 1995), carbohydrate partitioning (Gaberščik et al. 2002) and decreased leaf area and lower tillering (Breznik et al. 2004; Breznik 2007, unpublished).

In contrast to the mostly reduced total biomass production, a few studies showed no effects on biomass production of barley and strawberry (Valkama et al. 2003) or

Table 12.1 Responses of common and tartary buckwheat to enhanced UV-B radiation corresponding to 17% reduction of stratospheric ozone layer (Breznik 2007, unpublished). Arrows indicate the trend: increase (↑), decrease (↓) or no changes (–) in parameters values

	Crop species	
Parameters	Common buckwheat	Tartary buckwheat
Chl $a + b$ (mg/g dry weight)	↓	↓
UV-A a.c. (relative units/dm^2)	↑	–
UV-B a.c. (relative units/dm^2)	↑	–
Fv/Fm	–	↓
Y	–	↓
TR (mmol H_2O/m^2 s)	↓	↓
Plant height (cm)	↓	↓
Leaf area (cm^2)	↓	–
Total plant biomass (g)	↓	↓
Seeds biomass (g)	↓	↓

UV-A a.c. – UV-A absorbing compounds; UV-B a.c. – UV-B absorbing compounds; TR – transpiration rate ; Fv/Fm – potential quantum yield of PSII; Y – effective quantum yield of PSII.

even an increase in broad bean and one cultivar of wheat (Al-Oudat et al. 1998). Al-Oudat et al. (1998) hypothesised that species and cultivars, that originate from locations with high total solar radiation, exhibit efficient protection, even respond positively to enhanced UV-B radiation. For example, higher tolerance to UV-B was detected in plants in Mediterranean areas (Stephanou and Manetas 1998).

Successful reproduction is very important for agriculture. Literature data on crop yield are contradictory, but the studies reporting negative effects are more frequent. Lower seed biomass has been detected in common and tartary buckwheat (Gaberščik et al. 2002; Breznik 2007, unpublished), rice, pea and mustard (Caldwell et al. 1998). Although the crop yield decreased under enhanced UV-B radiation, the harvest index (economic yield) was not affected in all crops, for example, it was not affected in tartary buckwheat (Yao et al. 2006b) and winter maize (Zheng et al. 2003). Responses of common and tartary buckwheat to enhanced UV-B radiation studied by Breznik (2007) have been reflected in Table 12.1.

12.4 UV-B Radiation in Combination with Other Environmental Parameters

12.4.1 Effects of Carbon Dioxide and Ultraviolet-B Radiation on Plants

The UV-B radiation effect can be increased or mitigated by other environmental factors. Koti et al. (2007) showed that elevated CO_2 levels compensated the damaging effects caused by negative stressors, such as high temperature and high UV-B radiation on growth and physiological parameters of *Glycine max*. Authors suggest that there is a need to develop breeding strategies to develop genotypes that will be able to cope with future climates at both vegetative and reproductive stages. Qaderi et al. (2007) studied the combined effects of UV-B radiation and CO_2 concentration on reproductive organs of canola. UV-B radiation exerts a detrimental effect on canola siliquas and seeds, and some of the negative effects of UV-B on these reproductive parts can be partially mitigated by CO_2. Additionally, elevated CO_2 induced increased assimilation and water use efficiency, but lowered transpiration under UV-B conditions.

12.4.2 Ultraviolet-B Radiation in Combination with Low Temperatures

UV-B radiation increased freezing tolerance in winter wheat seedlings (*Triticum aestivum* L.). This response probably involves the scavenging enzymes and compounds in the antioxidant defence systems, particularly the ascorbate–glutathione cycle (Yang et al. 2007).

12.4.3 Drought Mitigates UV-B Radiation Effects

Evidence of both synergistic and antagonistic interactions between enhanced UV-B radiation and drought in plants has emerged in recent years (Kakani et al. 2003), but the mechanisms involved have received little attention and remained quite unknown (Alexieva et al. 2001).

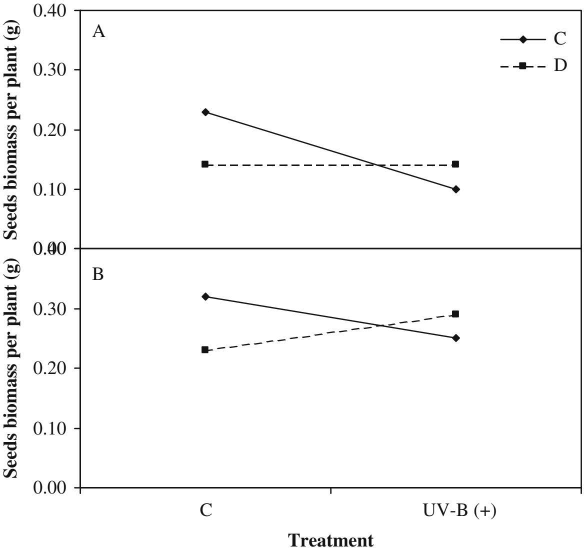


Fig. 12.1 Interaction plots presenting combined effect of enhanced [UV-B (+)] UV-B radiation (corresponding to 17% reduction of stratospheric ozone layer) and drought (D) on seeds biomass in common (**A**) and tartary (**B**) buckwheat

Studies on pea and wheat showed that negative effects of enhanced UV-B radiation were alleviated by moderate drought, and resulted in increased dry mass production (Alexieva et al. 2001). It is quite well known that during drought, some plants promote the production of phenolic substances and wax that improves UV-B radiation screening and reflection, and thus reduces UV-B damaging effects (Kulandaivelu et al. 1997). Ameliorating effects of drought were ascribed to increased activity of antioxidative enzymes that protect plants against oxidative damages caused by UV-B radiation (Alexieva et al. 2001). On the other hand, Caldwell et al. (1998) suggested that drought masks UV-B effects on plants, because drought constitutes a stronger stress for plants than enhanced UV-B radiation.

In contrast, the earlier study of Runeckles and Krupa (1994) revealed that enhanced UV-B radiation increased the sensitivity of crops to drought. These UV-B effects were ascribed to altered stomatal function by UV-B radiation that additionally affects plant water status. Effect of combined effect of enhanced UV-B radiation coupled with drought on seed biomass has been shown in Fig. 12.1.

12.4.4 UV-B Radiation and Selenium

Plants treated with selenium exhibit increased tolerance to enhanced UV-B radiation. This suggested that an anti-oxidative role of selenium that prevented plants from oxidative damage (Ekelund and Danilov 2001).

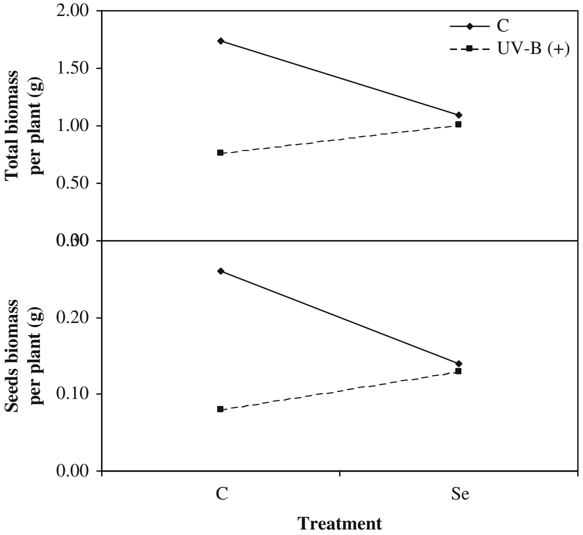
Germ et al. (2005) studied the impact of ambient and filtered solar UV-B radiation and selenium (Se) treatment on photochemical efficiency, respiratory potential, measured by electron transport system (ETS) activity, and yield in pumpkins (*Cucurbita pepo* L.). Exclusion of UV-B from the solar radiation more than doubled the yield of pumpkin fruits in Se untreated plants and increased the yield by 1.6 fold in Se treated plants. These results indicate the sensitivity of yield of pumpkins to current levels of UV-B in the ambient solar radiation in Slovenia.

UV-B radiation and selenium treatment did not affect potential or effective photochemical efficiency of PSII. The values of the potential photochemical efficiency were on the same level as the theoretical maximum of 0.83, indicating an undamaged antenna complex (Schreiber et al. 1995; Bischof et al. 1998).

Respiratory potential, measured by ETS activity, was lower in pumpkin plants grown under ambient radiation conditions, while the addition of Se had no effect on their ETS activity and hence their metabolic potential. The high yield established in pumpkins grown under UV-B exclusion conditions was related to the high ETS activity. Results suggest that UV-B radiation causes the reduction of the flow of electrons in the terminal electron transport system, which consequently results in lower yield of fruits.

The stimulatory effect of selenium on the yield, similar to the effect in pumpkin fruits, was reported in *Lolium perenne* (Hartikainen et al. 2000), lettuce (Xue et al. 2001) and potato (Turakainen et al. 2004). The increase in yield was much higher in plants that were exposed to UV-B excluded solar radiation, together with

Fig. 12.2 Interaction plot presenting combined effect of enhanced UV-B [UV-B (+)] radiation (corresponding to 17% reduction of stratospheric ozone layer) and selenium (Se) (1 mg Se/l Na_2SeO_4) on total and seeds biomass in common buckwheat



selenium spray, than in those that received either treatment alone. The positive role of Se has been demonstrated for *Euglena gracilis* (Ekelund and Danilov 2001) and for lettuce (Pennanen et al. 2002); the accelerating effect of Se on the growth of UV-B treated plants could be the result of the protection of chloroplast enzymes. Interactive effect of enhanced UV-B radiation in combination with Se treatment studied by Breznik (2007) has been depicted in Fig. 12.2.

12.5 Conclusion

Stratospheric ozone depletion, resulting in increased intensity of UV-B radiation at the Earth's surface, has been one of the most evident changes since the 1980s. The effects of enhanced UV-B radiation on plants have been widely studied, showing that UV-B radiation can have a deleterious effect. Crops have been shown to be even more sensitive, because they have been subjected to long-term breeding. The effects of enhanced UV-B radiation showed that UV-B radiation damaged DNA, as shown in tobacco and *Arabidopsis*. Transpiration rate was decreased by 50% in common and tartary buckwheat as well as in soybean. The changes in photosynthesis in many species, e.g. rice and maize, are mainly due to degradation or inactivation of Calvin cycle enzymes rather than disruption of PSII photochemistry. Studies of respiratory potential for common buckwheat, tartary buckwheat and pumpkin under irradiation indicated lower terminal electron transport activity and thus lower respiration. Studies of crop phenology under enhanced UV-B radiation showed variable results. Delayed germination and flowering were found in bean. The habitus of crops is also affected by enhanced UV-B radiation. Numerous crops, such as tartary buckwheat, common buckwheat, soybean, rice and cotton, when treated by UV-B radiation, were shorter as a consequence of disturbances during the early development stage. As a consequence of a loss of apical dominance, many crops, including bean and *Lolium sp.*, exhibited increased tillering. However, the study on common and tartary buckwheat revealed reduced tillering. Leaf area index decreased in many crops, like cotton and common and tartary buckwheat. Enhanced UV-B radiation frequently affects the biomass distribution and reproduction of most crops. Recent studies on common buckwheat, tartary buckwheat, pea plants, wheat, bush bean, rice, soybean and cotton showed decreases in total biomass production of 10–50%. In contrast, a few studies showed no effects on biomass production of barley and strawberry or even an increase, as reported in broad bean and a cultivar of wheat. The effect of UV-B radiation can be increased or mitigated by environmental factors. Elevated CO_2 levels compensated the damaging effects caused by UV-B radiation on growth and physiological parameters of soybean and canola. Further, UV-B irradiation increased freezing tolerance in winter wheat seedlings. Studies on pea and wheat showed that negative effects of enhanced UV-B radiation were alleviated by moderate drought. Ameliorating effects of drought could be ascribed to increased activity of anti-oxidative enzymes that protect plants against oxidative damage caused by UV-B radiation. On the other hand, it was suggested that drought masks UV-B effects on plants, because drought constitutes a stronger stress for plants than enhanced UV-B radiation. It was also shown that plants treated with selenium exhibit increased tolerance to enhanced UV-B radiation due to an anti-oxidative role of selenium that prevents plants from oxidative damage as has been detected in pumpkin, lettuce, ryegrass, potato and buckwheat.

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Chapter 13
Physiological Responses of Higher Plants to UV-B Radiation

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13.1 Introduction

Seven percent of the electromagnetic radiation emitted from the sun is in the range of 200–400 nm. As it passes through the atmosphere, the total flux transmitted is greatly reduced, and the composition of the UV radiation is modified. Short-wave UV-C radiation (200–280 nm) is completely absorbed by atmospheric gases. UV-B radiation is often defined as 280–320 nm. However, the legal definition provided by the International Commission on Illumination sets the UV-B radiation range as 280–315 nm. UV-B radiation is maximally absorbed by stratospheric ozone and thus, only a very small proportion is transmitted to the Earth's surface, whereas UV-A radiation (315–400 nm) is hardly absorbed by ozone. In the past 50 years, the concentration of ozone has decreased by about 5%, mainly due to anthropogenic pollutants, such as chlorofluorocarbons, releasing Cl atoms that catalytically remove ozone molecules from the atmosphere. The surface concentration of ozone has risen from less than 10 ppb prior to the industrial revolution to a day-time mean concentration of approximately 40 ppb over much of the northern temperate zone. If current global emission trends continue, surface ozone might rise over 50% by this century. Ozone depletion is particularly severe over the Antarctic continent, where a dynamically isolated air mass cools down to extremely low temperatures during the austral winter, facilitating ozone photo-destruction and formation of the so called springtime "ozone hole". Depletion of stratospheric ozone has increased solar ultraviolet-B radiation at high- and mid-latitudes in both Southern and Northern hemispheres (Frederick et al. 1994). However, ozone destruction is more intense over the Southern hemisphere with measured solar UV-B fluxes up to 50% more than those at comparable latitudes in the Northern hemisphere (Seckmeyer et al. 1995). Consequently, enhanced solar UV-B may have a greater impact on plants in agricultural production and in natural ecosystems in the Southern than in the Northern hemisphere (Madronich et al. 1995). The global trend of increasing solar UV-B

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radiation has been confirmed in mid-latitude in Japan (Sasaki et al. 2002) and in New Zealand in the 1990s (McKenzie et al. 1999). Furthermore, according to the report of UNEP (2002) (United Nations Environmental Programme: Environmental effects of ozone depletion and its interactions with climate change), it will take several years before a beginning of an ozone recovery can be confirmed at individual locations, nevertheless the concentrations of most anthropogenic precursors of ozone depletion are now decreasing (McKenzie et al. 2003). Although UV-B is only a minor component of solar radiation, its potential for causing biological damage is exceptionally very high due to its high energy and even small increase could lead to significant biological damage. The UV-B region of the spectrum is absorbed by a wide range of biologically active molecules, such as nucleic acids, aromtic amino acids, lipids and phenolic compounds. However, no specific photoreceptor molecule has been identified that can perceive the UV-B signal.

The crop growth, total biomass, final seed yield, thousand-grain weight and the amount of photosynthetic pigments decreased under near-ambient and enhanced UV-B radiation, while crop development was promoted by enhanced UV-B radiation (Yao et al. 2006). UV-B radiation-exclusion studies have indicated that many plants exhibit greater productivity in the absence of solar UV-B radiation (Krizet et al. 1997, 1998; Mazza et al. 1999; Kumagai et al. 2001).

The differences in the sensitivity to UV-B radiation were not related to the geographical locations. Physiological effects of UV-B on plants depend on plant species, the nature of UV-B treatment and interaction with other environmental factors. There were differences in the UV-B sensitivity among the cultivars of the same crop (Hidema et al. 2001). Deleterious UV-B effects on plants include reduced photosynthesis, biomass reduction, decreased protein synthesis, impaired chloroplast function, damage to DNA. UV-B radiation also produces oxidative stress increasing active oxygen species (AOS). These AOS have been proposed as the common second messenger for signal cascade leading to activation of a number of transcription factors (Schreck and Baeuerle 1991). The four compounds – AOS, salicylic acid (SA), jasmonic acid (JA) and ethylene, have been shown to be key regulators of gene expression in response to UV-B. The rise of AOS levels appears to regulate the synthesis of SA, JA and ethylene.

UV-B stress and resultant damage in plants can be classified as (i) DNA damage: formation of single-strand breaks (cyclobutane dimers) and indirect physical damage to the DNA by free radical formation, (ii) membrane damage: peroxidation of unsaturated fatty acids and alteration in the membrane lipid composition, (iii) photosynthetic apparatus damage: changes in the routes of electron paths in PSII, thylakoid ultrastructural damage, chlorophyll reduction and damage to Rubisco, (iv) hormonal damage: photodestruction of indole acetic acid and resultant inhibition of cell expansion. Each of these types of damage is directly or indirectly related to the formation of AOS under UV-B stress. Physically avoiding UV-B from penetrating into the plant system offers the first line of strategy to counter UV-B stress; this can be done by manipulating epidermal and cuticular layers of the leaf (wax, hairs). The second line involves increasing UV-B absorbing pigments, such as flavonoids and anthocyanins. The third line is to ameliorate or alleviate UV-B stress. Recent research has identified several beneficial effects of Se in plants which include antioxidative properties that can stimulate plant growth (Hartikainen et al. 2000).

13.2 UV-B Sensitivity and Productivity

A reduction in biomass accumulation is often a reliable indication of plant's sensitivity to UV-B radiation, since it represents the cumulative effects of damaged or inhibited physiological functions (Smith et al. 2000). However, measurements of other physiological parameters, such as chlorophyll content and levels of UV-B absorbing compounds, have also proved to be useful indicators of UV-B tolerance or sensitivity (Greenberg et al. 1997). Morphological characteristics can also affect species' sensitivity to UV-B radiation by influencing the amount of UV-B radiation intercepted (Bornman and Teramura 1993). For example, canopy leaves receive more sunlight, and hence more UV-B than under storey plants. Monocots, which have a vertical pattern of leaf growth, tend to intercept less direct sunlight than herbaceous dicots, whose leaves tend to grow horizontally (He et al. 1993). In addition, leaf hairs can also attenuate ultraviolet light (Karabourniotis et al. 1992, 1994) and leaf reflectance.

Since photosynthesis is dependent on the light-harvesting properties of the chlorophylls *a*, UV-B induced reduction in chlorophyll may result in lower levels of biomass accumulation, and hence it might be an useful indicator of UV-B sensitivity. Many plants display reduced photosynthetic function and chlorosis as symptoms of UV-B stress (Renger et al. 1989; Barnes et al. 1990; Strid et al. 1994). Conversely, plants adapted to high UV-B conditions retain higher photosynthetic integrity during UV-B exposure than those originating from low UV-B environments (Caldwell et al. 1994; Barnes et al. 1987). Therefore, the plants, which are able to maintain chlorophyll levels during UV-B exposure, are often less sensitive (Bornman and Vogelmann 1990; Greenberg et al. 1997). Increased levels of UV-B absorbing compounds lessen the symptoms of UV-B damage, such as growth inhibition and photosynthetic damage (Mirecki and Teramura 1984; Murali and Teramura 1986). Flavonoids and related compounds absorb strongly the light in the UV-region, but not in the photosynthetically active regions of the spectrum (Bornman and Vogelmann 1990; Cen and Bornman 1993), thus allowing photosynthesis to continue while UV wavelengths are attenuated at the epidermis. It has been reported that cultivars with higher levels of flavonoids prior to the onset of UV-B treatment (Gonzales et al. 1996) as well as those that can rapidly accumulate these compounds (Murali and Teramura 1986) are better protected during UV-B exposure. Since their protective properties are well established, it was expected that plants with high levels of UV-absorbing compounds would be more tolerant than those with lower levels. However, such a trend was not observed by Smith et al. (2000). Mean levels of UV-absorbing compounds did not differ significantly between the tolerant and sensitive groups, nor did an ability to increase the level of UV-screening pigments in response to UV-B necessarily reduce sensitivity. As tolerance to UV-B radiation probably involves many mechanisms, it is not unexpected that a species' sensitivity to UV-B cannot be determined by a single factor, such as UV-absorbing compounds. Leaf surface properties can affect the amount of UV-B that reaches underlying tissue through altered reflectance (Cen and Bornman 1993). Leaf hairs (trichomes) modify the micro-environment of the leaf, primarily through their extension of the boundary layer and reduction in the water loss (Ehleringer 1984). They can also reduce the amount of UV-B radiation penetrating to the epidermis (Karabourniotis et al. 1992), by their UV-absorbing pigment content (Karabourniotis et al. 1992, 1994). Waxes, which are sometimes UV-B targets themselves (Gonzales et al. 1996), are among the factors that can contribute to reflectance, at least in the visible light region. Hence, plants with a thick waxy cuticle were UV-B tolerant. The plants, which wax cover is lesser, were more susceptible to UV-B radiation, considering primary photosynthesis reactions, recorded by chlorophyll a fluorescence (Skórska and Szwarc 2007). Waxiness may contribute to tolerance, but when a few plants with waxy cuticles were tested, the connection was found tenuous. Recently, Gonzales et al. (1996) showed that the difference in sensitivity among six *Pisum sativum* cultivars was not associated with surface wax amounts or properties. Similarly, the removal of waxes from the epidermis of plants does not increase UV-B damage (Day et al. 1992). Thus, it seems unlikely that the presence of epicuticular wax is itself a protective feature. High reflectance of visible radiation by desert or alpine plants is attributed to epicuticular wax on the leaf surface (Reicosky and Hanover 1978), but this property does not cover the UV-B range of the spectrum (Day et al. 1992; Gonzales et al. 1996). The heavy shading, that occurs in plants with large leaf area and small internodes, could make these plants less susceptible to UV-B damage. Differences in UV-B sensitivity between species represent the relative contributions of morphological, physiological and biochemical differences, but variations in sensitivity within species are usually subtle. Sato and Kumagai (1993) examined 198 rice cultivars, belonging to five Asian rice ecotypes and Japanese lowland and upland rice groups. They found that sensitivity to UV-B radiation varies widely among different cultivars belonging to the same ecotype and the same group. In addition, rice cultivars originating from regions with higher ambient UV-B radiation do not necessarily exhibit higher tolerance. The differences in sensitivity to UV-B radiation were not related to the geographical locations at which these cultivars are grown.

Transgenic rice plants bearing the CPD photolyase gene of the UV-resistant rice cultivar had 5.1 and 45.7 fold higher CPD photolyase activities than the wild-type and were significantly more resistant to UVB-induced growth damage, and maintained significantly lower CPD levels in their leaves during growth under elevated UV-B radiation. Conversely, plants with little photolyase activity, was severely damaged by elevated UV-B radiation, and maintained higher CPD levels in its leaves (Hidema et al. 2007).

It is suggested that generalizations on plant sensitivity to UV-B, based on growth form and functional type, could be misleading (Musil et al. 2002). Seventeen herb, shrub and tree species of commercial and ecological importance in southern Africa were exposed to ultraviolet-B (UV-B, 280–315 nm) radiation. Leaves of trees had altered chlorophyll *a* and *b*, carotenoid and flavonoid concentrations, but those of shrubs or herbs did not. Correlation analyses did not support the view that growth is less negatively affected in species with thick leaves or in those where leaf thickness increases, or in species with naturally high leaf flavonoid contents or that are able to synthesize additional flavonoids in response to UV-B enhancement.

Crop response to UV-B radiation is associated with UV-B intensity, environmental factors and growing season (Yao et al. 2006). Plant photosynthetic productivity is determined by leaf area and the photosynthetic rate per unit area. Light attenuation can be modified by leaf ultrastructural changes under enhanced UV-B, and this could contribute to resultant changes in photosynthesis. A common response is addition of spongy mesophyll cells which increases leaf thickness. Several layers of shorter and wider palisade cells are formed under high UV irradiance. The logical interpretation of these changes is that a padding of cells is developed in the surface to prevent UV from reaching the site of electron transport, both in the chloroplast and mitochondria. The reduction of mesophyll cells under UV radiation would affect the CO_2-concentrating mechanism in C_4 plants and subsequent transport of CO_2 to bundle sheath cells. The increase in mitochondria and decrease in peroxisomes would imply that there would have been an imbalance in the transport of malate and glycine between mitochondria and peroxisome and a concomitant decrease in photorespiration (Shanker 2006). Bornman and Teramura (1993) suggested that the seedlings in the early stages of development may be particularly sensitive to UV-B, while Teramura and Caldwell (1981) reported greater inhibition of photosynthesis in developing leaves of soybean than those irradiated at full expansion. Leaves growing under elevated UV-B will receive greater cumulative exposure throughout the lifetime of leaves than mature leaves irradiated after developing without UV-B. Nogués et al. (1998) have suggested that the primary cause of reduced area of leaves was UV-B-induced inhibition of cell division. According to Logemann et al. (1995), this is an adaptive rather than injurious response.

Pumpkin (*Cucurbita pepo* L.) plants, grown in the field, have been shown to be sensitive to ambient UV-B radiation, resulting in significantly reduced yields of fruit (Germ et al. 2005). Total grain nitrogen content and grain storage protein content increased under elevated UV-B radiation, while tiller number, dry mass, panicle number, grain yield and grain size significantly decreased (Hidema et al. 2005). Such reductions were enhanced by lower temperature and less sunshine (Kumagai et al. 2001). UV-B radiation significantly increased the concentrations of both rosmarinic and carnosic acids, as well as other rosemary compounds, such as naringin and carnosol (Luis et al. 2007). Leaf rutin concentration and the amount of UV-B absorbing compounds were increased by UV-B exposure (Yao et al. 2006).

13.3 UV-B-Induced Photomorphogenesis

Many morphological and anatomical changes have been reported from plants grown under long-term UV-B regimes. Photomorphogenesis in seedlings is largely controlled by red/far-red-absorbing phytochromes and by blue/UV-A- absorbing cryptochromes (Batschauer 1999; Quail 2002). Rather, the perception of UV-B radiation has been either connected to the action of phytochromes and cryptochromes, as they partially absorb UV-B, or attributed to DNA, aromatic amino acids, and phospholipids (Beggs et al. 1996). The nature of UV-B receptors, however, has been not elucidated so far. There is a large agreement that a UV-B receptor consists of a protein with a bound pterin or flavin as chromophores (Galland and Senger 1988). Low doses of UV-B stimulate photomorphogenesis in etiolated seedlings, because the inhibition of hypocotyls elongation and opening of the apical hook are mediated independently of phytochromes and cryptochromes and exhibit a UV-B influence response relationship (Ballaré et al. 1991; Suesslin and Frohnmeyer 2003). Enhanced UV-B significantly inhibited pollen germination and tube growth in most of investigated species (Feng et al. 2000). Plants grown at high UV-B produced smaller flowers with shorter standard petal and staminal column lengths. Flowers so produced had less pollen with poor pollen germination and shorter tube lengths (Koti et al. 2005).

Decreases in whole plant growth were associated with reductions in attributes linked to cell division and cell expansion (Hofmann et al. 2001). The reduced proportion of palisade parenchyma and increased proportion of spongy parenchyma and intercellular space may have been specific reactions to enhanced UV-B radiation (Bornman and Teramura 1993; Heijari et al. 2006). Nogués et al. (1998) observed that in response to UV-B, there was a large reduction in the number of palisade mesophyll cells rather than formation of smaller cells. The reduced number of leaves was a further UV-B-induced change in morphology. In the cells exposed to enhanced UV-B radiation, the number of mitochondria increases, while the number of peroxisomes significantly decreases (Heijari et al. 2006). It was suggested that this might be an adaptive mechanism to compensate for the photodamage in peroxisomes. Even below-ambient levels of UV-B were capable to induce an increase in thylakoid surface area relative to the chloroplast volume typical of a low-PAR shade response in sunflowers and alter starch metabolism (Fagerberg 2007).

Supplementary UV-B radiation suppresses the growth not only of shoots, but also of plant roots (Naito et al. 1997). Kalbina and Strid (2006) established UV-B-dependent decrease in biomass, rosette size and leaf area and significant ecotype-specific genetic variability in general UV-B responses in *Arabidopsis*.

13.4 UV-B and Other Factors

In the nature, plants usually experience several stresses simultaneously. The effect of enhanced UV-B radiation on plants can be modified by other co-occurring or simply by changing environmental factors, like atmospheric CO_2, water availability, mineral nutrient availability, heavy metals, temperature, air pollutants. Plant response can be also modified by plant growth rate, developmental stage, growth form (herbs cf. trees) and functional type. Visible radiation is an important ameliorating factor. There are some suggestions that plant responsiveness to UV-B may be influenced by the ratio of UV-B light to visible sunlight as much as by the absolute level of UV-B radiation (Deckmyn and Impens 1997). White light ameliorates UV-B induced responses, including gene expression. These effects were the consequences of photosynthetic electron transport and photophosphorylation (A-H-Mackerness et al. 1996; Jordan 2002). The sensitivity of plants to UV-B irradiation is much greater at low PPFD and UV-A levels (Cen and Bornman 1990). UV-A was particularly effective at ameliorating UV-B damage when PPFD was low, but has no effect at higher PPFDs (Caldwell et al. 1994). High PPFDs may confer protection from UV-B damage by increasing photosynthesis and the available biochemical energy for defence and/or repair processes. This hypothesis was supported by the observations that UV-B-induced reduction in plant growth were less severe when elevated CO_2 concentrations were used to increase photosynthesis in the absence of any change in flavonoid content (Adamse and Britz 1992). UV-B-induced damage was alleviated either by elevated CO_2 or exposure to high irradiance-visible radiation (Kumagai and Sato 1992). It was shown that either elevated CO_2 or somewhat higher temperature had similar effects in reducing the growth-inhibiting effects of elevated UV-B radiation on sunflower and maize seedlings (Mark and Tevini 1997). Elevated CO_2 levels compensated the damaging effects caused by UV-B radiation on the physiological parameters, such as plant height, leaf the area, total biomass, net photosynthesis, total chlorophyll content, phenolic content, relative injury and wax content (Koti et al. 2007). UV-A, which is closer to PAR, has generally been assumed to be less detrimental to plant growth, but recent findings have demonstrated that UV-A has the ability to decompose the UV-B absorbing pigments, thereby reducing the capacity of the plants to fight off UV-B stress (Fukuchi et al. 2004). The effects observed under UV-A stress can be attributed to the presence of a cryptochrome, operating in the blue-UV-A spectral range. It must be noted that the ratio of UV-A to biologically effective UV-B radiation in the field is up to 7.5 times greater than in the greenhouse (Dai et al. 1995). Heijari et al. (2006) suggest that biological effects of the UV-A component of sunlight may be of much greater importance than was believed formerly. UV-A helps to develop a protective pathway against UV-B-induced damage of photosynthetic apparatus (Joshi et al. 2007).

Sullivan and Teramura (1990) have established that as a result of water stress, the concentration of leaf flavonoids in plants increased, which provided greater UV-B protection. On the other hand, elevated UV-B radiation in the field tended to alleviate drought symptoms (Manetas et al. 1997). In the moss species, UV-B radiation inhibited growth when the moss was under water stress, but stimulated growth when the moss was well hydrated (Gehrke 1998). Enhanced UV-B radiation and drought stress have an additive negative effect on growth of willows (Turtola et al. 2006). Higher tolerance to drought stress for *Arabidopsis* plants grown under UV-B radiation may be attributed to both increased proline content and decreased stomatal conductance (Poulson et al. 2006).

In pine seedlings grown in a growth chamber, enhanced ozone concentration led to an increased sensitivity of the seedlings to UV-B radiation, since ozone reduced the contents of UV-B-absorbing pigments in the plant tissues. In tobacco, UV-B radiation increased the level of ozone-induced foliage lesions (Thalmair et al. 1996). Soybean plants grown in the field were found sensitive to ozone in the air, but not sensitive to supplementary UV-B (Miller et al. 1994).

The uptake of certain nutrients may be modified by UV-B radiation. In oilseed rape (*Brassica napus*) plants grown under enhanced UV-B and simultaneously exposed to different cadmium concentrations, the manganese content in the shoots decreased while the contents of magnesium, calcium, cooper and potassium significantly increased. The UV-B had no additional influence on the nutrient content of the roots (Larsson et al. 1998). Prasad and Zeeshan (2005) reported that the effect of combination UV-B and Cd was more detrimental to growth, photosynthesis and antioxidant enzymes. According to Yao and Liu (2007), supplemental nitrogen made the plants more sensitive to enhanced UV-B, although some antioxidant indexes increased. The ability of Se to ameliorate UV stress has been shown by Hartikainen and Xue (1999) and Germ et al. (2005).

UV-B exposure caused increases in jasmonic acid (JA) levels and ethylene production in *Arabidopsis thaliana* (A-H-Mackerness et al. 1999a, b). Both jasmonic acid and ethylene are components of signal pathways leading to regulation of gene expression in response to UV-B radiation and subsequent resistance/tolerance to UV-B (A-H-Mackerness et al. 2001). The production of ethylene in plant tissue under normal conditions is low, but the level of this hormone increases in response to external stresses (Wang et al. 1990). JA and its methyl ester (JA-Me) are endogenous growth substances identified in many plant species (Sembdner and Parthier 1993). Jasmonates are one of the simplest non-traditional plant hormones with diverse roles and functions, including a potential role in plant defence (Creelman and Mullet 1997a, b). Jasmonic acid is derived from linolenic acid via octadecanoid pathway (Reinbothe et al. 1994). UV-B irradiation results in the perturbation of plant membranes and/or the activation of lipases which induces the signal-transduction pathway which activates stress inducible genes; DNA damage might be initial signal that activated the octadecanoid pathway. Thus, the response to UV-B radiation may be regulated by jasmonic acid derived from linolenic acid through the octadecanoid pathway.

Salt (NaCl, KCl, $NaNO_3$) pre-treatment resulted in considerable loss of UV-induced and UV-B absorbing compounds. In the meantime, chlorophyll fluorescence parameters and oxygen evolution in salt pre-treated seedlings were less affected by UV-B (Fedina et al. 2006). Application of mineral nutrients (N, P and K) showed a significant positive response in wheat and mung bean by ameliorating the negative impact of UV-B (Agarwal and Rathore 2007). An increase of polyamines (Lutz et al. 2005), and especially of putrescine level in thylakoid membranes upon elevated UV-B exposure, comprises one of the primary protective mechanisms in the photosynthetic apparatus of the tobacco variety against UV-B radiation.

13.5 UV-B and Photosynthesis

Photosynthetic damage is associated with stomatal behaviour, photosynthetic enzymes and pigments (Teramura and Sullivan 1994; Tevini 1994), electron transport chain (Tevini et al. 1991), as well as disruption of the chloroplast membrane

(Bornman 1989). Examination of a single process or component does not allow identification of the primary UV-B-induced limitation. UV-B can cause decreases in the light-saturated CO_2 assimilation rate in the absence of any major inhibition of the quantum efficiency of PSII photochemistry, thus demonstrating that UV-B inhibition of PSII photochemistry is not a ubiquitous primary limitation to photosynthesis (Middleton and Teramura 1993). UV-B radiation can induce stomatal closure directly by inhibiting K+ accumulation (Wright and Murphy 1982) as well as opening of stomata. However, the mechanism of this complex effect of UV-B is not clear (He et al. 2005). Allen et al. (1997) demonstrated that UV-B-induced decrease of CO_2 assimilation rate was associated with reduction in the maximum carboxylation activity of Rubisco and this was confirmed by in vitro biochemical assays of Rubisco activity. UV-B induced reduction in both Rubisco activity and content (He et al. 1993; Fedina et al. 2007). The rate of RuBP regeneration can decline as a result of UV-B irradiation in the absence of any significant effects on quantum efficiencies of PSII photochemistry (Ziska and Teramura 1992). The inhibition of photosynthesis or electron transport under excess light or UV irradiation (Niyogi 1999) may elevate the photosensitization process as well as the formation of AOS in this way. The formation of singlet oxygen via photosensitization was suggested to play an important role in damaging the D1 protein (Hideg et al. 1994). The probable electron transfer from electron transport chain, especially in photosystem I (PSI), to molecular oxygen, the way to quench extensive energy is an alternative source of AOS. Photoreduction of molecular oxygen by primary electron acceptor in PSI complex is thought to be the main source of superoxide in illuminated chloroplasts. The rate of H_2O_2 production in the Mehler reaction is sufficiently high to cause an inhibition of CO_2 fixation of up to 50% (Asada 1994). Although more attention has been focused on the production of AOS during photosynthesis, UV-B-induced damage in the respiration pathway may increase the electron transfer from the respiratory transport chain to molecular oxygen and thus results in the oxidative stress indirectly (Norbury and Hickson 2001). A research on etiolated tissue is indicative of a strong link between the photosynthetic apparatus and UV-B-induced gene expression (Jordan et al. 1994). This may, in part, account for the lack of UV-B effect on gene expression in etiolated tissue when photosystems are not functional. In etiolated seedlings, the degree of increase in UV-induced cyclobutyl pyrimidine dimers (CPD) levels was the highest, while the contents of UV-absorbing compounds were the lowest and no photorepair of CPD could be detected (Kang et al. 1998). In addition, the levels of AOS and antioxidants have been related to UV-B response at different developmental stages of photosynthetic apparatus (A-H-Mackerness et al. 1998). Photosystem II and to some extent, PSI may also be impaired (Okada et al. 1976). In PSII, ultraviolet-B radiation has been shown to inactivate primarily the water-oxidizing complex with additional damage to quinone electron acceptors and tyrosine donors (Renger et al. 1989; Vass et al. 1996). UV-B-induced impairment of photosynthesis resembled visible light-induced photoinhibition, being associated with enhanced degradation of the D1, and to a lesser extent, the D2 protein (Greenberg et al. 1997). It appeared that no significant replacement of the D2 occurred under UV-B stress (Babu et al. 1999).

Although most studies suggest that UV primarily affects D2 and D1 proteins of PSII, Baker et al. (1997) suggested that the primary damaging effect of UV-B on photosynthesis is not on PSII reaction centers, but on a range of important soluble enzymes in the chloroplast. On the other hand, it has been reported that solar UV-B filtering does not cause any change in the photochemical efficiency of PSII (Huiskes et al. 2001).

UV-B radiation caused reductions in the amount of chlorophyll, oxygen evolution, activity of PSII and CO_2 fixation in barley seedlings (Fedina et al. 2003), chloroplast proteins, especially ribulose-5-bisphosphate carboxylase/oxygenase (Rubisco), light-harvesting chlorophyll a/b binding protein of PSII (Takeuchi et al. 2002). Allen et al. (1997) reported that the loss of Rubisco was a primary factor in UV-B inhibition of CO_2 assimilation. According to Hidema et al. (1996), a reduction in Rubisco was greater in UV-sensitive than in UV-resistant strains. Takeuchi et al. (2002) showed that synthesis of Rubisco, but not LHCII, was significantly suppressed by UV-B.

13.6 DNA Damage and Repair

DNA is one of the major targets of UV damage, and UV-B radiation is capable of directly altering its structure. UV radiation induces lethal photodamage in DNA, resulting in the production of CPDs and pyrimidine (6-4) pyrimidone phoproducts [(6-4) products]. The majority of DNA damage consists of CPDs (approximately 75%) that are formed between adjacent pyrimidines on the same strand. Such DNA damage may be lethal or mutagenic to organisms and may impede replication and transcription (Brash et al. 1987; Sancar et al. 2004). DNA damage can be repaired by three kinds of reaction: photoreactivation of CPDs and 6-4 photoproducts by specific photolyase, nucleotide excision repair (dark repair) and recombination repair (homologous recombination - the removal of CPDs (Ries et al. 2000). Photoreactivation is a light-dependent enzymatic process using UV-A and blue light to monomerize pyrimidine dimers: the enzyme photolyase binds to the photoproducts and breaks the chemical bonds of the cyclobutane ring and restores integrity of the bases. Photolyase is activated by light ranging from 300 to 600 nm (Sancar 1994). Photolyases of 6-4 photoproducts are expressed constitutively in etiolated seedlings (Chen et al. 1994; Hada et al. 1999), whereas the expression of CPD photolyases is frequently regulated by the light. Photorepair of CPDs has been reported in several plant species, including gingko (Trosko and Mansour 1969), Arabidopsis (Britt et al. 1993), alfalfa (Quaite et al. 1994), soybean (Sutherland et al. 1996), cucumber (Takeuchi et al. 1996), rice (Hidema et al. 1997), maize (Stapleton et al. 1997), and wheat (Taylor et al. 1996). CPD photoreactivation is the major pathway in plants for repairing UV-radiation induced DNA damage (Britt 1996). Studies in various plant species have shown that this photoregulation may involve phytochromes (Langer and Wellmann 1990), blue/UV-A (Hada et al. 1999), visible light (Ahmad et al. 1997; Kang et al. 1998), and UV-B-receptors (Ries et al. 2000). There is no information about the photocontrol of photolyase activity under natural conditions, when all these photoreceptors are simultaneously excited. Hidema et al. (2000) found that CPD photorepair ability is one of the main factors determining UV-B sensitivity in rice. Examination of 17 rice cultivars showed that the more resistant cultivars to UV-B exhibited higher photolyase activities in comparison to less resistant cultivars (Teranishi et al. 2004). Visible radiation included in the natural sunlight was sufficient to photorepair CPDs formed by solar UV-B radiation alone or solar plus supplemental UV-B. An increase in visible radiation during culture could enhance the capacity for photorepair in rice. The degree of photorepair of CPDs in the seedlings grown at high irradiance ($350\ \mu\text{mol m}^{-2}\ \text{s}^{-1}$) was higher than that in the seedlings grown at low irradiance ($50\ \mu\text{mol m}^{-2}\ \text{s}^{-1}$). No photorepair of CPDs could be detected in etiolated dark-grown seedlings (Kang et al. 1998). In this connection, Langer and Wellmann (1990) observed differences in photoreactivating enzyme activity of CPDs between hypocotyls from etiolated and light-grown bean seedlings and concluded that the induction of CPD photolyase was under phytochrome control. Ahmad et al. (1997) have reported that the expression of PHR1 gene for photolyase in etiolated seedlings of Arabidopsis is induced by high white light. Takeuchi et al. (1996) have shown that a high activity of CPD photolyase was present in etiolated cucumber cotyledons.

In excision repair, dimmers are replaced by de novo synthesis in which the undamaged complementary strand is employed as a template. This multistep process involving multiple enzymes has been found to operate with only a low capacity in plants.

13.7 UV-B Defence-Flavonoids and Antioxidants

UV-B radiation causes an increased production of active oxygen species (AOS) (Smirnoff 1993; Rao et al. 1996; A-H-Mackerness et al. 1998; Costa et al. 2002) which include superoxide radicals ($^1O_2^{-*}$), singlet oxygen ($^1O_2$), hydrogen peroxide (H_2O_2) and hydroxyl radical (OH^*). The mechanism of singlet oxygen generation by UV radiation is not clear, but the process does not appear to be linked to the UV-induced damage of photosynthesis (Barta et al. 2004). Under UV-B irradiation, the generation of AOS is dependent on, but not proportional to, the dose of UV-B. The presence of photosynthetically active radiation (PAR, 400–700 nm) in the irradiation regime is important for photosynthetic organisms. It was found that AOS formation was higher when PAR was included. This suggests that photosynthetic electron transport chain and photosynthetic processes both contribute to the formation of AOS (Foyer et al. 1994a; Asada 1994). The overproduction of AOS is potentially toxic to cells and induces oxidative damage, while on the other hand, increased AOS may act as an alert signal to induce protective responses. It has been shown that H_2O_2 acts as a signal to induce defence gene expression (Desikan et al. 2000). As might be expected, some of these genes encode antioxidant enzymes, defence and stress-related proteins (Desikan et al. 2001). The antioxidant defence system includes enzymatic antioxidants, such as superoxide dismutase (SOD), catalase (CAT), peroxidase (POD), ascorbat peroxidase (APX), dehydroascorbate reductase (DHAR) and glutathione reductase (GR) and non-enzymatic low molecular weight antioxidants, such as ascorbat, glutathione, α-tocopherol and carotenoids (Ahmad 1995). The antioxidants, like ascorbat and glutathione participate in both enzymic and non-enzymic H_2O_2 degradation (Foyer et al. 1994b).

SOD (EC 1. 15. 1. 1) converts ${}^1O_2^{-*}$ radicals into H_2O_2 and O_2. In plants, CAT (EC 1.11.1.6) is one of the main H_2O_2-scavenging enzymes that dismutates H_2O_2 into water and O_2, and it is a constitutive component of peroxisomes (Corpas et al. 1999). CAT activity slightly increased at UV-B irradiation (Yannarelli et al. 2006), but the pattern of isoforms remained unchanged. APX (EC 1.11.1.11) is a specific peroxidase that catalyses the elimination of the toxic product H_2O_2 at the expense of oxidizing ascorbate to monodehydroascorbate. APX isoenzymes are distributed in at least four cells compartments: the stroma, the thylakoid membrane, the microbody and the cytosol (Asada 1992). Peroxidases (EC 1.11.1.7) are enzymes that catalyse the H_2O_2-dependent oxidation of a wide variety of substrates, mainly phenolics (Dunford 1986), and they are often found in multiple molecular forms. The function of these isozymes and their regulation remain largely unknown. It was suggested that sunflower acclimatizes themselves to UV-B radiation by induction of different isoforms of POD (Yannarelli et al. 2006). After supplemental UV-B radiation, CAT and SOD activities enhanced; meanwhile APX did not change (Dai et al. 1997). In UV-B irradiated sunflower cotyledons, total APX activity was not modified, but their isozymes were clearly affected demonstrating a selective response of each isoform, presumable due to its location (Yannarelli et al. 2006). The induction of antioxidant enzymes with peroxidase activity (CAT and GPX) indicated that hydrogen peroxide participates actively on UV-B plant response (Rao et al. 1996).

13.7.1 Flavonoids

It has been reported that flavonoids and related phenolic compounds are specifically increased, when plants are exposed to enhanced UV-B radiation (Beggs and Wellmann 1994; Wagner et al. 2003) and were linearly dependent on UV-B influence (Cen and Bornman 1993). In plants, the protective role of flavonoid polyphenolics in the expression of tolerance to UV-B radiation has been shown repeatedly (Li et al. 1993; Lavola et al. 2003). Flavonoid synthesis can be increased by elevating PPFD and UV-A levels (Cen and Bornman 1990). UV-A and blue light stimulates expression of the chalcone synthase gene of the flavonoid biosynthesis pathway (Jenkins et al. 1997; Wade et al. 2001). As very low irradiance is not inductive for chalcone synthase gene expression, the size of the irradiance may also be important (Bilger et al. 2007). In *N. officinale*, total flavonoid quantities were 10-fold lower in UV exposed young leaves compared to *S. alba*, in which flavonoid accumulation was induced by UV specifically in old leaves. In *S. alba*, relative contents of quercetin flavonols increased at the expense of kaempferols in UV exposed leaves. Hydroxycinnamic acid concentrations were not affected in both species (Reifenrath and Muller 2007).

Accumulation of phenolic compounds in epidermal cells is an useful tool for effective screening of UV-B radiation, thus protecting the mesophyll tissue (Bornman et al. 1997; Kolb et al. 2001). UV-absorbing compounds are mainly phenylpropanoids, such as cinnamoyl esters, flavones, flavonols, and anthocyanins esterrified with cinnamic acids. These compounds are likely to impart protection to the plants by absorbing radiation in the UV-B region of the spectrum (not absorbing photosynthetically active radiation) and thereby diminishing the penetration of UV radiation. In the pea (*Pisum sativum*) mutant *Argenteum*, whose epidermal cell layers can be removed easily and analyzed separately from the mesophyll, flavonol glycosides (putatively protective UV-B-absorbing compounds) were found to be located mainly (Weissenböck et al. 1986) or entirely (Hrazdina et al. 1982) in the epidermis. UV-B absorbing pigments are present all through the leaf, but accumulate significantly in adaxial epidermal cells (Cen and Bornman 1993). Olsson et al. (1999) showed that in *Brassica napus*, kaempferol glycosides were the most abundant flavonoid compounds of the adaxial epidermis, whereas the abaxial cell layer chiefly contained hydroxycinnamic acid derivatives. Quercetin glycosides were the compounds that were increased most in both epidermal cell layers in UV-B stressed plants relative to control plants (Olsson et al. 1999). The protecting action of flavonoid compounds is due to their strong absorbance in the range of 220–380 nm and their photostability. However, a little is known about the relative contribution of different phenolic compounds to the UV-B screening capacity of leaves. In this respect, the investigation of UV-B tolerance and resistance of flavonoid deficient mutants can throw more light on involvement of different classes of UV-B absorbing compounds. Two groups of higher plants can be distinguished in relation to flavonoid accumulation in different tissues. In the most of the dicotyledons (legumes, bean, soybean, pea etc.), flavonoids are usually situated in the epidermis. However, in monocotyledons (barley, oat, rye), the epidermis as well as mesophyll can accumulate flavonoids (McClure et al. 1986; Weissenböck et al. 1986). Protective responses are stimulated by UV-B radiation, including increased production of UV-B-absorbing compounds. Sensitivity to UV-B radiation is strongly parallel to the levels of constitutive and UV-induced flavonoids, thus demonstrating the protective role of flavonoids towards UV radiation. On the other hand, the response of plant to UV-B radiation depends also on development and physiological state of the plant and on other stress factors acting simultaneously with UV-B radiation. Temperature is a significant factor controlling biosynthesis of phenolic compounds and modulating UV-B screening and possibly also UV-B resistance (Bilger et al. 2007). As a results of exposure to UV-B, the content of total flavonoids increases in a time dependent manner. For barley plants, flavonoid accumulation starts 4 h after UV-B exposure and increases up to 120 h after radiation (Fedina et al. 2005). In the case of callus cultures of *Lysimachia*, continuous irradaiation with UV-B results in more vigorous increase at the first 24 h and reached its maximum at 72 h and slightly decreased thereafter

(Hollósy 2002). UV-B radiation stimulates the expression of the genes encoding phenylalanine ammonia-lyase, as the first stage of the phenylpropanoid pathway, and chalcone synthase, the key stage which commits the pathway to flavonoid synthesis (Bornman and Teramura 1993; Beggs and Wellmann 1994). In addition, they may offer an additional protection by having antioxidant activity (Brown et al. 1998). Kolb et al. (2001) observed that PSII was protected against UV-B damage by epidermal screening, related to increased leaf phenolics (A_{314} and A_{360}), however, UV-B inhibition of CO_2 assimilation rate was not diminished. According to Middleton and Teramura (1993), although both UV-B-absorbing compounds and carotenoids increased in response to UV-B irradiation, only carotenoids and not the UV-B absorbing compound (A_{300}) could be related to protection of photosynthesis. Although some laboratory experiments have demonstrated that the concentration of phenolic compounds increases with increasing irradiance (Cen and Bornman 1990; Day 1993), the experimental data regarding the protective role of these compounds against UV-B radiation are still a few and speculative. An accumulation of certain phenolpropanoid compounds (such as flavonoids and anthocyanins) in the vacuoles of the epidermal and sub-epidermal cell layer, which is thought to act as UV-B filters, plays an important role in coping with UV-B-induced damage. Hada et al. (2003) have found that the higher accumulation of anthocyanins and UV-B absorbing compounds in rice plants did not effectively function as a shield to protect plants from supplementary UV-B radiation. Dai et al. (1995) also reported that the differences in the sensitivity to UV-B radiation among rice cultivars could not explain quantitative differences in flavonoid-shielding compounds. Markham et al. (1998) reported that flavonoids might play a more subtle role in plant UV-B protection than simple UV-B screening in a UV-B tolerant rice cultivars. Flavonoids act also as antioxidants (Gould et al. 2002). Thus, their importance in protecting mechanisms against UV-B induced biological damage in plants has been questioned.

13.8 Conclusion

UV-B radiation presents a potential risk for the plant growth, physiology, and quality of plant productivity. The threat to productivity in global agriculture due to stratospheric ozone depletion cannot be overstated, nor should it be overlooked. Attempts at quantitative and qualitative predictions of expected effects and the search for a suitable ameliorant or a stress alleviant are being met with mixed outcomes. One of the reasons for this is the limitation in controlled-environment studies. Results from greenhouse or growth-chamber studies and field studies on UV-B effects are often conflicting or difficult to interpret, because of unrealistically high UV irradiation levels, inadequate levels of UV-A, low PAR or other technical difficulties. Evidences do exist for quantitative and qualitative changes in crop yield. Plant breeding and genetic engineering for UV-B tolerance is an important aspect to be considered in order to avoid significant crop production loss. UV-B sensitive cultivars of many important plant species have been found in field and laboratory tests. In most cases, the origin of the sensitivity is unknown, making the design of bioengineering or breading programs for improving UV light resistance difficult. Identification of the receptors involved in perception of UV-B would be of great value and provides a means of manipulating UV-B responses. It is very important to perform studies under field conditions, at the different climates and latitudes, in which different cultivars are grown and where the UV-B quality is quite different. The effects of UV-B radiation on plants are strongly influenced by seasonal microclimate conditions. Unusual climatic conditions, such as lower temperature and less sunshine, enhanced harmful effects of UV-B. To such studies, should be added elevated levels of CO_2. While such information is needed for direct effects on crop species, the studies must also include information about the possible long-term effects on growth, joint effects with other pollutants, incidence of pathogens and insect pests, intra-species competition, and crop-weed relationships. It appears that the effects of UV-B radiation on photosynthesis, growth, and development of plants are caused by altered gene action as consequences of UV-B damage of nucleic acids during the longer-term impact. This is currently a topic of intensive research. The ecological effects of UV-B at the community level are difficult to predict, as large variations occur between species. Perhaps the most pressing need at the moment is to obtain field information about the effects of UV-B.

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Chapter 14
Possibility of Water Management for Mitigating Total Emission of Greenhouse Gases from Irrigated Paddy Fields

Kazunori Minamikawa and Kazuyuki Yagi

14.1 Paddy Fields as a Source of Greenhouse Gas Emission

14.1.1 Rice Production and Its Water Use

Rice (*Oryza sativa* L.) is one of the most important cereals that supplies 20% of the total calorie consumption of the world in 2000 (IRRI 2007). Rice is cultivated in many countries throughout the world, but 88.4% of rice area is concentrated in Asia and its production reaches 90.4% of the total of the world (Table 14.1). The current rice production is more than twice of that in the 1960s, and will further increase to feed the growing human population, especially in Asian countries.

According to the water availability, rice area is distributed to the following four environments: irrigated, rainfed lowland, upland, and deepwater. In Asia, the rice area distributed to irrigated, rainfed lowland, upland, and deepwater is 57, 33, 7 and 3%, respectively (Fig. 14.1). Irrigated rice is grown in lowland fields with ensured irrigation water for one or more crops a year. Now-a-days, some kinds of irrigation systems are adapted to all the producing countries, but are not entirely spread over because of the socio-economic and geographical reasons. Rice yield in each country almost corresponds to the spreading rate of irrigation system, though other inputs, such as fertilizer and machinery, also contribute to the yield (Table 14.1 and Fig. 14.1). Therefore, it is expected that irrigation systems will spread to paddy fields in currently non-irrigated lowland area with the socio-economic advancement in the near future.

14.1.2 Mechanism of CH_4 Emission from a Paddy Field

Rice can be semi-permanently cultivated in the flooded paddy fields. This is not the case in upland fields where the injury by continuous cropping occurs. Weed rankness

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Table 14.1 Rough rice production, area, and yield for major producing countries in Asia, 2005

Country	Production ($\times 10^3$ t)	Area ($\times 10^3$ ha)	Yield ($\mathrm{t\,ha^{-1}}$)
China	183354	29270	6.26
India	130513	43400	3.01
Indonesia	53985	11801	4.57
Bangladesh	40054	11000	3.64
Vietnam	36341	7340	4.95
Thailand	27000	10200	2.65
Myanmar	24500	6270	3.91
Philippines	14615	4000	3.65
Japan	11342	1706	6.65
Asia	559349	136142	4.11
World	618441	153953	4.02

Source: FAO (2006)

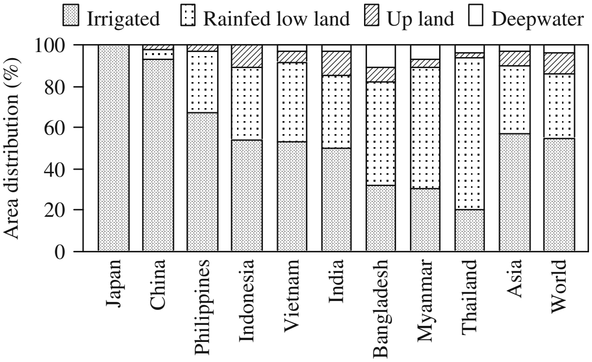


Fig. 14.1 Distribution of rice area by water use for major producing countries in Asia, 2000 (IRRI 2007)

is also controlled by flooding. The followings are known as the reasons to inhibit the injury by continuous cropping in flooded paddy fields: the supplement of micronutrients, the wash away of harmful substances, and the decrease in insect pest.

However, the development of reductive conditions in the flooded soil may cause the root rot of rice plants by sulfide toxicity. Furthermore, CH_4, a greenhouse gas (GHG) is produced by methanogenic bacteria (methanogens) in the reduced soil, and emitted to the atmosphere (Fig. 14.2a). In the flooded paddy soil, after the trapped molecular O_2 is quickly consumed, sequential reduction of the following soil oxidants progresses according to the thermodynamic theory: NO_3^-, Mn^{4+}, Fe^{3+}, $SO_4{}^{2-}$, and CO_2. Methane is the end product of this process.

Generally, the oxidative-reductive conditions in the soil are indicated by soil redox potential (soil Eh). There are also other indicators, such as the terminal electron-accepting processes and the oxidative capacity (Gao et al. 2002). Soil Eh is a comprehensive measure of soil chemical and biological redox processes. With the development of the reductive conditions, soil Eh changes from $+600$ to <-200 mV. Generally, CH_4 production begins at approximately -150 mV, and gets stimulated at lower range.

The major carbon substrates for CH_4 production are $H_2 + CO_2$, formate, acetate, methyl alcohol, and methylamine. The following two reactions are dominant in the

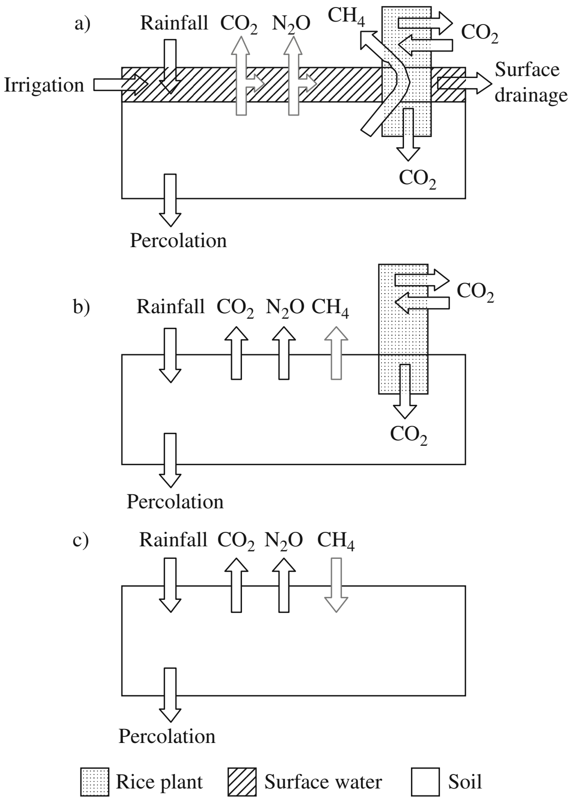


Fig. 14.2 Major GHG exchanges in a paddy field during: (**a**) flooded period, (**b**) drainage, and (**c**) fallow period. Dissolved GHGs in water include CO_2, CH_4 and N_2O. *Light color arrow* indicates the limited flow (slight amount, temporal, or conditional). Weeds and algae are excluded

methanogenesis: the reduction of CO_2 using H_2 and the transmethylation of acetate (Takai 1970; Schütz et al. 1989). These substrates are supplied from soil organic matter, exudates and sloughed tissues of rice plants, and applied organic matter (Watanabe et al. 1999; Kimura et al. 2004). A part of produced CH_4 is consumed by methanotrophic bacteria (methanotrophs) in the oxidative zones as the rhizosphere of rice plants and thin soil layer interfacing with the surface water. During the flooded period, produced CH_4 is emitted to the atmosphere mainly through the aerenchyma of rice plants, and other pathways are ebullition and diffusion through the surface water (Schütz et al. 1991; Butterbach-Bahl et al. 1997). On the other hand, during drainage, CH_4 trapped in the soil is emitted by the direct diffusion (Fig. 14.2b). During the fallow period, a little amount of CH_4 is consumed in the oxidative soil by methanotrophs (Fig. 14.2c).

From the recent estimates (Cao et al. 1998; Mosier et al. 1998; Neue and Sass 1998; Yan et al. 2003), the total CH_4 emission from the paddy fields in the world accounts for 4.2–9.0% of the global CH_4 emission (600 Tg CH_4 y^{-1}). Worldwide, rice area is slightly increasing year by year (IRRI 2007), and this trend will not turn around due to growing human population. Therefore, paddy fields will continue to be a major CH_4 source of the world from now onwards.

14.1.3 Other GHG Exchanges

The CO_2 and N_2O are also emitted and/or consumed through various pathways (Fig. 14.2). Flooding and drainage by water management drastically change the pattern of GHG exchanges. Other kinds of field management, such as ploughing, straw incorporation, and nitrogen fertilization, also affect the GHG exchanges.

Between the soil and the atmosphere, GHG exchanges occur by the ecologies of microorganisms and plants. Carbon dioxide is exchanged by the photosynthesis and respiration of rice and weed plants, and the respiration of heterotrophs in the soil (i.e. the decomposition of soil organic matter). Carbon fixed by rice plants is partly removed from the field by harvesting, and the rest is incorporated to the soil by ploughing. Nitrous oxide is produced as a by-product of nitrification under the oxidative conditions and an intermediate product of denitrification under the semi-reductive conditions. Under the strict reductive conditions where CH_4 production occurs, N_2O produced by denitrification is further denitrified to N_2, the end product of this process. The nitrogen substrates for N_2O production are mainly supplied from fertilizer, soil organic matter, and applied organic matter.

The other pathway for GHG exchanges is the dissolved emission to soil percolating water or surface drainage (Fig. 14.2). As reported by Mosier et al. (1998b), the indirect N_2O emission through the groundwater and surface flow has a quantitative significance. The N_2O detected as the indirect emission is produced mainly by denitrification, using NO_3^- as the sole substrate, in the soil and the subsequent groundwater. Although it is not clear about the contribution of N_2O produced in the sub-surface soil layer of the field to the total indirect emission, non-negligible amount of dissolved N_2O was detected in the percolating water at a lysimeter experiment (Minamikawa et al. 2007a). The CO_2 and CH_4 produced in the soil are also, more or less, dissolved in the water, and then indirectly emitted to the atmosphere.

Soil organic carbon has various existing forms of itself. Although this is not a flow, but a stock, decomposed organic matter is used as a major substrate for CO_2 and CH_4 production. The amount of soil carbon in the topsoil is more than several thousands $\mathrm{g\,C\,m^{-2}}$ depending on the soil type. Moreover, a huge amount of carbon is stored in the soil including the sub-soil and the subsequent layers. Therefore, its decomposition, even if a little amount, has a strong impact on CO_2 emission. The conservation of soil carbon is known as ‘soil carbon sequestration’.

14.2 Mitigation of GHG Emission from Irrigated Paddy Fields

14.2.1 Quantitative Significance of Mitigating GHG Emission from Paddy Fields in Asia

How much can agriculture contribute to the mitigation of GHG emission? Figure 14.3 shows the inventories of GHG emitted from major anthropogenic sources in the selected Asian countries. Generally, the distribution of agriculture

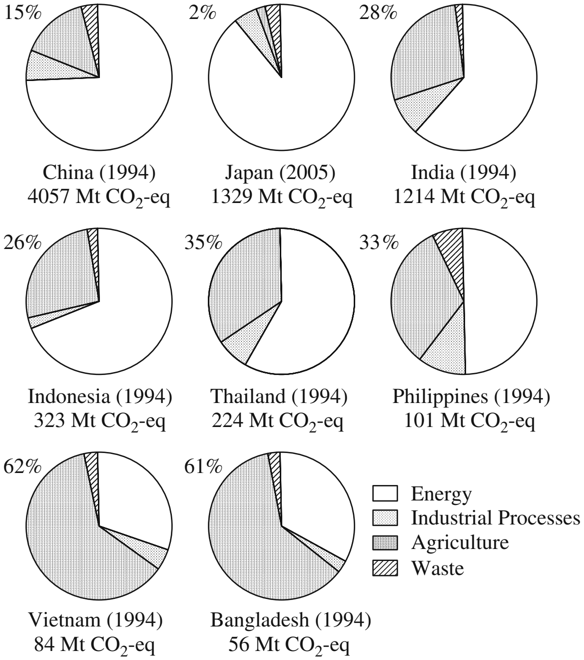


Fig. 14.3 The GHG inventories for rice producing countries in Asia (UNFCCC 2005; Nojiri et al. 2007). The value indicates the distribution of agriculture sector. Land-use change and forestry are excluded

sector is diverse and reaches one third to two third of the national total in the developing countries. Although the percentage of agriculture will fall with the development of socio-economic conditions in the developing countries, the decrease in GHG emission from agriculture is effective in mitigating the total GHG emission (i.e. the total CO_2-equivalent emission considering the Global Warming Potential, GWP) under the present conditions.

As for the distribution of each GHG emission within a country, those of CH_4 and N_2O, that mainly emitted from agriculture, are relatively high in the developing countries (Table 14.2). In these countries, rice production is also high (Table 14.1), and thus CH_4 emitted from paddy fields has a large contribution to the national

Table 14.2 Distribution of GHG emission for rice producing countries in Asia (%)

Country	CO_2	CH_4	N_2O
Japan	96.3	1.8	1.9
China	75.8	17.7	6.5
India	64.2	31.3	4.6
Thailand	63.2	29.2	7.7
Philippines	57.4	28.7	13.9
Indonesia	55.1	39.3	5.6
Vietnam	30.1	57.9	12.0
Bangladesh	35.8	54.5	9.7

Source: UNFCCC (2005) and Nojiri et al. (2007)

Calculated by CO_2 equivalent considering the GWP. Land-use change and forestry are excluded.

GHG emission. Therefore, the mitigation of CH_4 emission from paddy fields will have significant impact on the total mitigation of GHG emission in Asia.

Well then, does the mitigation have the cost economy? Originally, paddy fields are the place for rice production, and thus its productivity is given the highest priority. An appropriate combination of field management is practiced to obtain suitable rice yield, but these practices have various effects on the GHG emission. Of course, some management practices can be win-win options that sustain rice yield and decrease CH_4 emission. However, farmers generally aim at economic rice production only. Although the consideration of environmental issues on various scales (e.g. GHG emission and water pollution) has some advantages in the developed countries, however, environmental concerns are not comparable to economic one in the present socio-economic situation. In the near future, mitigation of GHG emission will be profitable through some environmental taxes and the Clean Development Mechanism of the Kyoto Protocol all over the world. Therefore, researchers should look ahead into the future, and have to clarify the efficiency of mitigation options quantitatively. Furthermore, even in disregard of profitability, researchers should prepare the mitigation options as a approach to contain global warming.

14.2.2 How Can We Mitigate the Total GHG Emission from An Irrigated Paddy Field Most Effectively?

14.2.2.1 Concrete Method of Mitigation

The factors controlling GHG emission range from microbial to a global scale. The global warming occurs on the largest scale, while microbial GHG production and consumption occur on the smallest scale. As reviewed by Minamikawa et al. (2006a), present mitigation options are mostly on the field scale, and conducted as one part of field management. At the present research level of the GHG emission from agro-ecosystems, the field-scale mitigation is the most feasible method that we can do it now.

All of the mitigation options essentially affect the ecology of microorganisms. However, the insights into the microbial ecology are often disregarded in the field-scale mitigation. Conrad (1996) has reviewed the significance of soil microorganisms as controllers of GHG emission. As suggested by Kumaraswamy et al. (2000) and Schimel (2000), further microbial studies will enable us to predict the ecosystem behavior and the ecological significance of diversity and community structure. In other words, further studies on the microbial scale will give us additional clues to the mitigation of GHG emission from paddy fields.

14.2.2.2 Which Gas and Pathway is the Main Target for the Mitigation?

It is very difficult to answer this question completely at the present research level. Then, what have the previous GHG studies clarified up to now?

Tsuruta et al. (1998) evaluated the direct GHG emission from Japanese paddy fields, considering the GWP. The CH_4, CO_2, and N_2O emission accounted for 78.2, 16.0, and 5.8% of the GWP, respectively. As for CO_2 emission (i.e. soil carbon budget), it is difficult to quantify because both the emission and consumption occur simultaneously, while CH_4 and N_2O have little consumption. It will take a little longer to evaluate CO_2 emission as accurate as CH_4 and N_2O emission. On the other hand, simultaneous measurement of CH_4 and N_2O emission has demonstrated the priority of CH_4 emission as the main mitigation target (Nishimura et al. 2004; Zou et al. 2005). Therefore, the decrease in CH_4 emission will be the best way to mitigate the direct GHG emission from paddy fields.

Next, how many are the GHGs emitted as dissolved into groundwater? Sawamoto et al. (2003) reported the result of simultaneous measurement of the direct and dissolved GHG emission at an upland field. As compared to the direct emission of CO_2, CH_4, and N_2O, 2.5, 58, and 4.6% were emitted to a subsurface drain, respectively. On the other hand, at a lysimeter paddy field, Minamikawa et al. (2007a) simultaneously measured the direct and dissolved GHG emission. The annual amount of dissolved N_2O emission to the percolating water was 30.6% of the direct one, and dissolved CO_2 was $34.2\,g\,C\,m^{-2}\,y^{-1}$. Between upland and irrigated paddy, soil redox conditions and amount of percolating water are largely different. Anyway, dissolved GHG emission would be lower than the direct one in an irrigated paddy field, but further studies are needed to understand the mechanism and amount of dissolved GHG emission.

The most difficult problem is the evaluation of soil carbon sequestration. The latest IPCC report suggests that soil carbon sequestration is one of the most effective options in mitigating CO_2 emission from agriculture (Smith et al. 2007). Of the total mitigation potential, the contribution of soil carbon sequestration, CH_4 emission, and N_2O emission was estimated to be 89, 9, and 2%, respectively. Its quantitative significance would defeat the mitigation of minor GHGs. However, there have been only a few reports on soil carbon sequestration in a paddy field (Witt et al. 2000; Ramesh and Chandrasekaran 2004). As reported by Sain and Broadbent (1977) and Devêvre and Horwáth (2000), flooding would delay the decomposition of soil organic carbon. Paddy fields may have a potential to store soil organic carbon more than upland fields.

There have been a lot of studies either related to GHG emission or soil carbon sequestration only. Now-a-days, we know that it is necessary to measure GHG emission simultaneously with soil carbon storage (soil carbon budget). Simultaneous measurement, considering the time scale, will give us the best answer to the optimal management practice for mitigating the total GHG emission from irrigated paddy fields.

14.3 Mitigation by Conventional Water Management

14.3.1 Different Aim Depending on the Water Availability

Water management is the most effective option in decreasing CH_4 emission from an irrigated paddy field, because it prevents the development of soil reductive conditions. From the statistical analysis of available data, Yan et al. (2005) reported that multiple drainage decreases CH_4 emission by 48% as compared to continuous flooding. This value has been adopted to the default emission factor by the IPCC report (2006).

The original aim of water management is different depending on the water availability. Where irrigation water is enough supplied as in the temperate region, water management is for the sound rice growth. Where irrigation water is not ensured as in the tropic region, water management has other aim, saving irrigation water. In the former region, mid-season drainage and intermittent irrigation are usually practiced by farmers. On the other hand, the alternate wetting and drying (AWD) is now being suggested to farmers in the later region.

14.3.2 Mid-Season Drainage and Intermittent Irrigation

14.3.2.1 Role for Sound Rice Growth

In the temperate region, water management is practiced to control surplus tillering and supply rice roots with molecular O_2 for preventing sulfide toxicity. Figure 14.4 mgpF4 shows a typical method of water management for an irrigated paddy field in Japan. Mid-season drainage is conducted at the panicle formation stage of rice plants for 1–2 weeks, aiming at the crack formation on the soil surface. Intermittent irrigation follows the mid-season drainage, and continues until several weeks before rice harvest. Intermittent irrigation is conducted at the several-day intervals of flooding and drainage. Generally, mid-season drainage and intermittent irrigation are conducted together.

Rice plant is especially sensitive to drought at the heading stage, and the rooting of rice seedling is stabilized by flooding. Therefore, the field during these periods should be flooded for sound rice growth. Additionally, control of surface water depth is conducted as occasion demands before mid-season drainage, in order to avoid the cool weather damage with the deep flooding of higher water temperature, and in order to stimulate the tillering of rice plants with the shallow flooding of lower water temperature. Although there remain some ambiguities in the timing and duration of drainage, appropriate water management certainly hastens rice growth and hence increases rice yield.

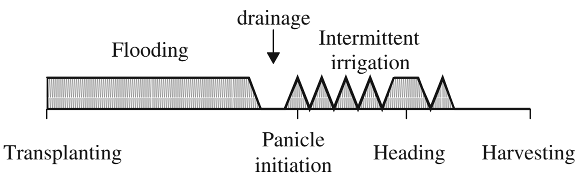


Fig. 14.4 Typical water management in an irrigated paddy field in Japan

14.3.2.2 Effects on GHG Emission

Many researchers have reported that mid-season drainage and/or intermittent irrigation decrease CH_4 emission without reducing rice yield (Sass et al. 1992; Yagi et al. 1996; Bronson et al. 1997; Wang et al. 1999). Well then, what about the effects on N_2O and CO_2 emission?

Generally, there is a trade-off relationship between CH_4 and N_2O emission (Fig. 14.5). Nitrous oxide has the high solubility of itself into water. Accordingly, almost all of N_2O dissolves into water if produced in the flooded soil, and then emitted to the percolating water. Furthermore, under the strict reductive conditions, dissolved N_2O is further denitrified to N_2. Therefore, the direct N_2O emission is usually observed at the non-flooded soil where CH_4 production and emission have been already interrupted (Figs. 14.2b, 14.2c, and 14.5). Hou et al. (2000) reported that significant N_2O emission only occurred when soil Eh was above +200 mV, which corresponds to the oxidative or semi-reductive conditions in the non-flooded soil. From the statistical analysis of available data, Akiyama et al. (2005) estimated the emission factor of fertilizer-induced N_2O emission for continuous flooding and mid-season drainage at 0.22 and 0.37%, respectively. This result supports that mid-season drainage increases N_2O emission as compared to continuous flooding.

N_2O emission was also observed just after fertilization even during the flooded period when the field received nitrogen fertilizer more than 300 kg N ha^{-1} to Chinese paddy fields (Cai et al. 1997; Zou et al. 2005). It derives from the diffusion through the surface water where dissolved N_2O concentration would be extremely high (Fig. 14.2a). On the other hand, with <100 kg N ha^{-1} application to Japanese fields, N_2O emission is hardly observed during the flooded period (Fig. 14.5).

The direct CO_2 emission also occurs during the drainage (Fig. 14.2b). Similar to N_2O emission, CO_2 emission increases with the duration of drainage. However, CO_2 emission (decomposition of soil organic carbon) as the direct and/or dissolved one, is inevitable whether under the flooded conditions or not. There were several cases that the GWP with mid-season drainage was higher than that with continuous flooding, such as when no straw was applied (Bronson et al. 1997) and when a large amount of chemical fertilizer or manure was applied (Cai et al. 1999). Therefore, a series of mid-season drainage and intermittent irrigation are effective in decreasing

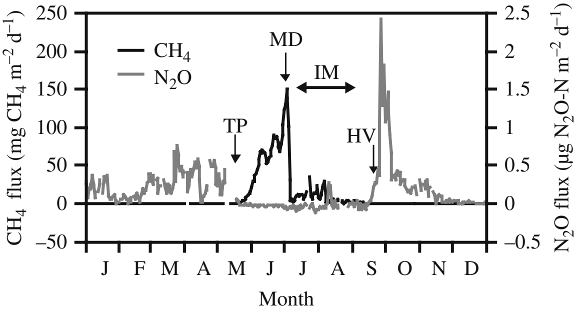


Fig. 14.5 Seasonal changes in CH_4 and N_2O flux at an irrigated paddy field (modified from Nishimura et al. 2004). TP, transplanting, MD; midseason drainage, IM; intermittent irrigation, HV; harvesting

CH_4 emission, but not necessarily brings the minimum GWP depending on other management practices.

14.3.3 Alternate Wetting and Drying

14.3.3.1 Role for Saving Irrigation Water

Agriculture accounts for 70% of the global water use (FAO 2006). Worldwide, water for agriculture is becoming increasingly scarce. Because of the competition with industry and domestic usage, it will be more difficult to secure a stable and adequate supply of irrigation water for paddy fields. To sustain a growing and rice-eating population in Asian countries, it is necessary to establish field management that will ensure current or higher levels of rice productivity with higher efficiency of water use without detrimental environmental impacts.

Several methods of water-saving irrigation have been developed according to the water availability, such as AWD (Bouman and Tuong 2001; Belder et al. 2004), continuous soil saturation (Borrell et al. 1997), and aerobic rice (Bouman et al. 2005; Xiaoguang et al. 2005) (Fig. 14.6). These methods reduce water inputs and increase the water productivity (rice yield/water use). Among them, AWD is suggested by the International Rice Research Institute (IRRI) to farmers, with a saving of about 15–30% of irrigation water input without reducing rice yield (Bouman et al. 2007).

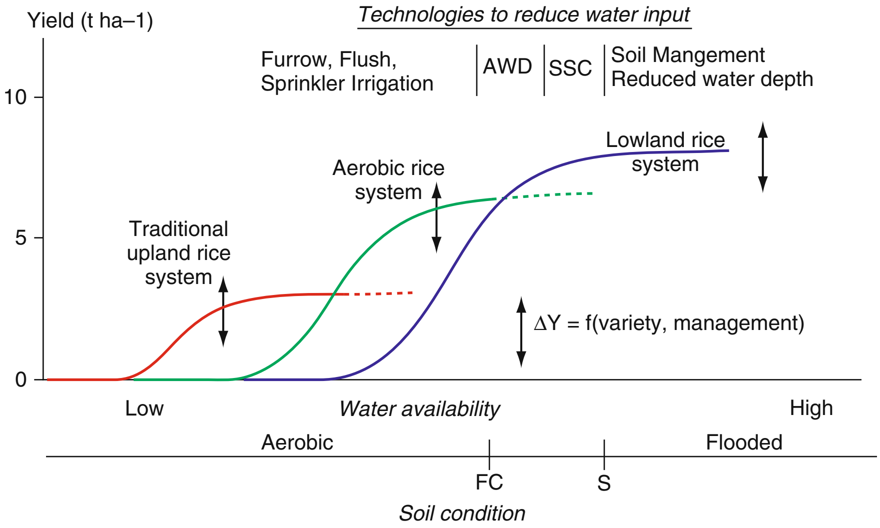


Fig. 14.6 Schematic presentation of yield responses to water availability and soil conditions in different rice production systems and their respective technologies to reduce water inputs (Tuong et al. 2005). SSC – saturated soil culture, FC – field capacity, S – saturation point, ΔY – change in yield

14.3.3.2 Method of AWD

The early stages of AWD were conducted as the several-day intervals of flooding and drainage (Tabbal et al. 2002; Belder et al. 2004). It is similar to the intermittent irrigation. The current method of AWD commonly practiced by farmers is based on the groundwater depth and/or dry conditions of surface soil. On the other hand, at the experimental level, AWD is conducted on the basis of soil moisture tension, an objective indicator for water content in the soil.

Soil moisture tension is measured by a tensiometer usually inserted to a depth of 15 cm. Figure 14.7 shows the method of water management for AWD. When the moisture tension decreases to the pre-determined criterion by drainage, the field is flooded to about 5 cm of surface water depth. The repetition of this routine is the method of AWD. The criterion is variable, and its determination depends on the water availability, soil type, target rice yield, etc. Similar to the conventional water management in the temperate region, the field is flooded to keep sound growth just after the transplanting and during the heading stage.

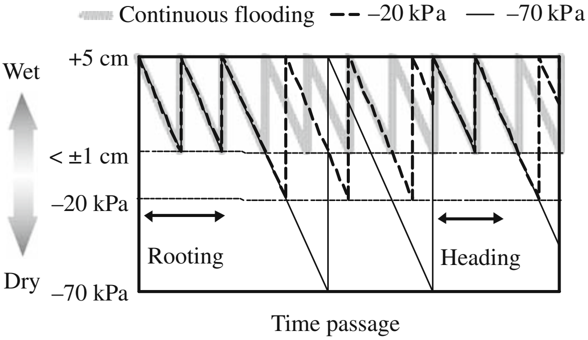


Fig. 14.7 Schematic diagram of the AWD. The −20 and −70 kPa are examples of the criterion for soil moisture tension. *Horizontal arrow* indicates the forced flooded period

14.3.3.3 Effects on CH_4 Emission

Drastic changes in water use by AWD will affect various aspects of paddy-field environment including GHG emission. Hosen (2007) reported the effects of AWD on CH_4 emission, using two criteria (−20 and −70 kPa). At the IRRI field in the Philippines, −20 kPa is used as a limit not to inhibit rice growth, which was derived from the previous studies (Hira et al. 2002). The −70 kPa is expediently set as a fairly severe limit for rice growth that would reduce rice yield. As compared to continuous flooding, the AWD, using the criterion of −20 kPa, decreased more than 70% of CH_4 emission during the dry-season growing period under the conditions of receiving fresh straw (4 t DW ha^{-1}). On the other hand, the AWD using −70 kPa decreased CH_4 emission by nearly 90% under the same conditions.

The AWD has a high potential to decrease CH_4 emission from irrigated paddy fields. However, in the AWD, the priority order of mitigating GHG emission seems to have been lower than water saving and increase in the water productivity. Further studies are necessary to evaluate the effects of AWD on the GHG emission.

14.4 Eh Control, Water Management Based on Soil Eh

14.4.1 Background for Development and Its Implications

In the temperate region, the timing and duration of mid-season drainage and intermittent irrigation depend on farmer's empirical knowledge and customary practices. However, these subjective decisions cannot cope with unusual weather conditions, such as cool temperature and intermittent rainfall, causing insufficient drainage and thus careless CH_4 emission. Moreover, CH_4 emission would occur before the pre-determined first drainage. These phenomena have a possibility to increase the uncertainty of the decreasing effects of water management on CH_4 emission.

Therefore, there is a room to improve the conventional water management to positively decrease CH_4 emission. One of the most useful indicators for predicting CH_4 emission from a paddy field is soil Eh. In order to positively decrease CH_4 emission without reducing rice yield, Minamikawa and Sakai (2005) have proposed the water management based on soil Eh, which named 'Eh control'.

Originally, water management is practiced to obtain suitable rice yield in various climate regions, but other aspects that must be considered are arising with the climate change. How is the new water management conducted? The problem is how to drain and flood, and thus it is necessary to establish a certain criterion to conduct the water management. One probable way to solve this problem is the use of an objective indicator for the factor controlling the issue. The AWD and Eh control are just the water management practices, using an objective indicator.

The use of an objective indicator is also to solve the secondary problem. The conventional water management finally depends on subjective decisions by farmers in any countries. Of course, the current method is useful to achieve a certain aim, but does not necessarily bring the best result. There is a room to reconsider and improve the current method more efficiently.

Therefore, an objective indicator is very useful to achieve the generalization and simplification of the conventional water management. Although it takes much to spread the new water management, it will be the mainstream in the future.

14.4.2 Method of Eh Control

The Eh control is now the water management at the experimental level. Figure 14.8 shows the schematic diagram of Eh control. The Eh control keeps the soil Eh between the pre-determined lower and upper limit by drainage and flooding. After the irrigation for rice transplanting, the flooded soil is drained when the soil Eh decreases to a lower limit. Thereafter, the drained soil is re-flooded when the soil

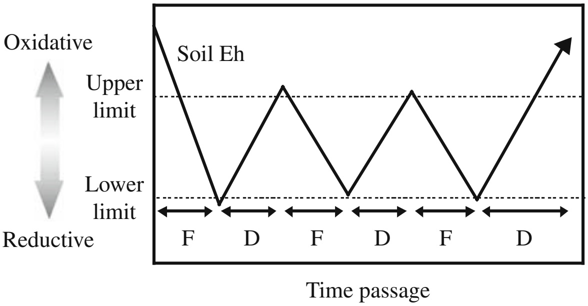


Fig. 14.8 Method of water management for the Eh control. Soil Eh is controlled between the upper and lower limit by drainage (D) and flooding (F)

Eh increases to an upper limit. The repetition of this routine is the method of Eh control. The lower limit is established to inhibit CH_4 production and emission. The upper limit is established to inhibit the excessive soil drying and also not to inhibit rice growth and yield. The change in soil Eh by Eh control is similar to that in soil moisture tension by AWD (Fig. 14.7).

The values for the lower and upped limit are variable. Generally, CH_4 production begins at approximately -150 mV and stimulated at lower values. Therefore, the lower limit has been empirically determined to be -150 mV. On the other hand, the upper limit has no correct solution. From the results of the preliminary pot experiment as described below, it is recognized that soil Eh should not be kept at over $+500$ mV for several days, which causes the excessive drying of soil and thus the reduction in rice yield. Moreover, soil Eh sharply increases after the drainage because of the crack formation on the surface soil, while the relatively slow decrease under the flooded conditions. Therefore, the upper limit has been expedientially determined to be +100 mV, in order not to reach the soil Eh $+500$ mV or so. It is desirable that the values of lower and upper limit are determined to prevent soil Eh from reaching the critical ones.

Soil Eh is measured by a common method using a portable Eh meter and platinum-tipped electrodes. Usually, the electrodes are inserted to a depth of 5 cm in the soil with 3–5 replications at a site. Although there is a variation within replications, soil Eh finally converges to almost similar value under the flooded conditions. On the other hand, during drainage, the crack formation causes a wide variation in soil Eh among replications. However, these are natural phenomena, and thus it is necessary to establish replications for monitoring the adequate values of soil Eh by the present method.

Surface soil level of the flooded field is usually uneven if puddled uniformly, and the surface is in a concave shape. Therefore, flooded water can not be drained entirely, indicating that soil Eh is also variable depending on the site within a field. The first drainage randomly forms the crack on the soil surface, and the following drainages promote the crack formation, where soil Eh is drastically fluctuated. Therefore, it is also necessary to select an adequate site for obtaining the representative value of soil Eh in the field.

14.4.3 Effects on CH_4 Emission and Rice Yield

14.4.3.1 Practicability of Eh Control

Can Eh control decrease CH_4 emission as compared to the conventional water management? Minamikawa and Sakai (2005) examined the effects of different kinds of water management on CH_4 emission under the pot conditions with two paddy soils. The early stages of Eh control had only the lower limit (−150 mV), and re-flooding was conducted 1–2 days after drainage. Figure 14.9 shows the seasonal changes in soil Eh and CH_4 flux with four kinds of water management at the Fluvisol pot. The Eh control had the distinctive change in soil Eh (Fig. 14.9a). Between the two soils, there was a wide difference in the frequency of the routine (the Fluvisol with high reductive potential > the Andosol with low reductive potential). Methane emission, measured by a closed chamber method, with Eh control was significantly lower than that with continuous flooding or mid-season drainage, and as low as that with intermittent irrigation (Table 14.3). Therefore, Eh control is effective to decreasing CH_4 emission. However, Eh control caused the excessive soil drying, and thus significantly reduced rice yield (Table 14.3), indicating that the necessity of considering rice growth while deciding the upper limit of soil Eh.

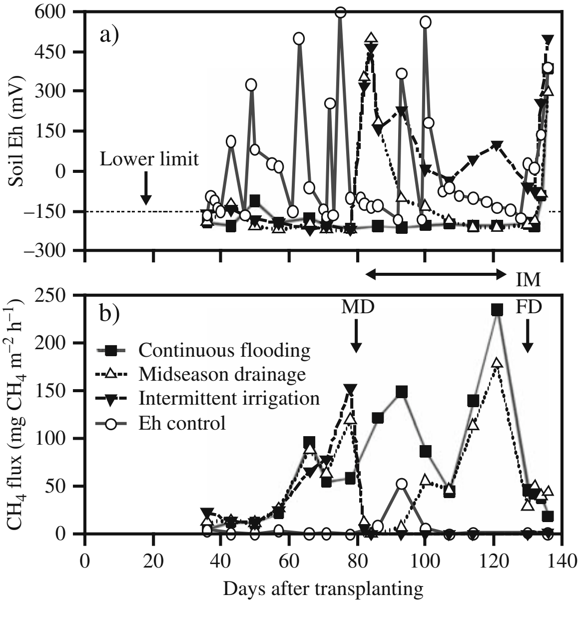


Fig. 14.9 Seasonal changes in (**a**) soil Eh and (**b**) CH_4 flux with four kinds of water management (modified from Minamikawa and Sakai 2005). MD – midseason drainage; FD – final drainage; IM – intermittent irrigation

14.4.3.2 Factors Affecting Eh Control

Is Eh control affected by other natural and artificial factors? Under the field conditions (Andosol) using the lower and upper limits (−150 and +100 mV), Eh control

Table 14.3 Total CH_4 emission and rice yield with four kinds of water management for two paddy soils

Treatment		Hulled rice yield	CH_4 emission
Soil	Water management	(g DW pot^{-1})	(g CH_4 m^{-2})
Andosol	Continuous flooding	12.6	135.3
	Mid-season drainage	11.9	75.2
	Intermittent irrigation	18.1	26.9
	Eh control	9.9	39.8
Fluvisol	Continuous flooding	13.3	192.6
	Mid-season drainage	20.3	133.8
	Intermittent irrigation	22.2	57.2
	Eh control	11.2	15.2

Source: Minamikawa and Sakai (2005).

worked well, and soil Eh was almost kept between the Eh limits (Fig. 14.10a). Accordingly, CH_4 flux with Eh control changed lower than that with continuous flooding (Fig. 14.10b), and the total CH_4 emission resulted in significant reduction (Table 14.4). Eh control with the upper and lower limit did not decrease rice yield as compared to continuous flooding (Table 14.4). However, Eh control did not increase rice yield with the mid-season drainage and intermittent irrigation.

Generally, rice straw application stimulates the development of soil reductive conditions and CH_4 emission. Although straw application increased the frequency of the routine, soil Eh was kept between the Eh limits as without straw application. Therefore, the total CH_4 emission with straw application was as low as that without one (Table 14.4).

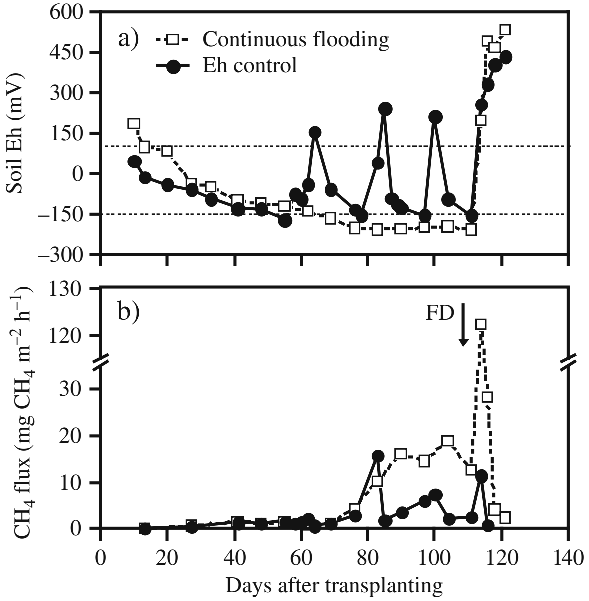


Fig. 14.10 Seasonal changes in (**a**) soil Eh and (**b**) CH_4 flux with different kinds of water management under the straw-applied field conditions (modified from Minamikawa and Sakai 2006b). The upper and lower limit of soil Eh is +100 and −150 mV, respectively. FD – final drainage

Table 14.4 Effects of water management on the total CH_4 emission and rice yield under the field conditions for two years

	Emission (g CH_4 m^{-2})		Yield (g DW m^{-2})	
Treatment	2003	2004	2003	2004
Eh control +S	7.31	6.99	508	663
Eh control −S	6.51	6.45	461	605
Continuous flooding +S	14.21	22.72	478	634
Continuous flooding −S	13.77	25.20	498	621

Source: Minamikawa and Sakai (2006b).
With (+S) and without (−S) rice straw incorporation (5 t DW ha^{-1}).

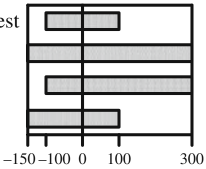
Annual variations in weather conditions usually affect the CH_4 emission. The field experiment was conducted in 2003 with the relatively cool summer and in 2004 with normal temperature summer. The effects of weather conditions were observed in the total CH_4 emission with continuous flooding and also rice yield level (Table 14.4). However, with Eh control, the total CH_4 emissions in both years were similarly low. Consequently, Eh control can always minimize CH_4 emission with regard to weather conditions and other field management.

14.4.3.3 Modification of Eh Limits

Is it possible to shift the Eh limit to more oxidative one both for further decreasing CH_4 emission and increasing rice yield? The experiment was carried out to examine the modifying effects of the upper limit (+100 or +300 mV) and the lower limit (−150 or −100 mV) in the Andosol field.

Table 14.5 shows the total CH_4 emission and rice yield with four combinations of the Eh limits. Rice yields with all Eh ranges were similar to or more than the conventional level, but the effects of the Eh limit were not significant because of the spatial variation within the field. Modification of lower limit (from −150 to −100 mV) had positive effects on decreasing CH_4 flux during flooded period, but the total CH_4 emission was predominated by a large amount of CH_4 flux during drainage, especially in the later period. While the lowest CH_4 emission was, as expected, observed in the Highest range (−100, +300 mV), the emissions from other ranges were fluctuated by the following reasons.

Table 14.5 Effects of shifting the Eh limit on the total CH_4 emission and rice yield

Range	Eh limit (mV)	CH_4 emission (g CH_4 m^{-2})	Rice yield (g DW m^{-2})
Narrowest		12.12	661
Widest		16.23	752
Highest		6.33	737
Lowest		8.46	636

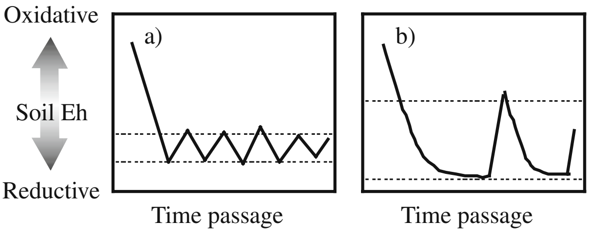


Fig. 14.11 Typical patterns of the changes in soil Eh causing higher CH_4 emission. (**a**) Narrow range keeps the semi-reductive conditions, (**b**) Wide range has the long-term reductive conditions. *Dotted line* is the Eh limit

Figure 14.11 shows typical changing patterns of soil Eh that cause a large amount of CH_4 emission. The Narrowest range (−100, +100 mV) had frequent routines, and did not dry the soil enough (Fig. 14.11a). The soil with the continuous semi-reductive conditions is ready to produce CH_4. Once the flooded period is kept longer, then CH_4 emission occurs. On the other hand, the widest range (−150, +300 mV) took a long time to lower soil Eh to −150 mV, and thus kept the reductive conditions, causing CH_4 emission (Fig. 14.11b). Therefore, under the conditions of this experiment, the combination of −100 and +300 mV (the Highest range) would be the best range that dries the soil enough and also has several times of short-term flooding. It will be possible to determine the ideal Eh range according to the soil type and other management practices.

14.4.4 Evaluation of Eh Control

In the last sub-section, the effects of Eh control on CH_4 emission and rice yield were reviewed from the results of three experiments. Although there remain some uncertainties of the result for other soil types than Andosol, soil Eh can be kept between the pre-determined range by Eh control disregard to other factors. Therefore, at the experimental level, Eh control is practical and can always decrease CH_4 emission with keeping rice yield more than the conventional level.

The N_2O emission with Eh control was not monitored. Similar to the conventional water management, N_2O emission will occur during the drainage for Eh control. Hou et al. (2000) indicated that an intermediate range of soil Eh between approximately −100 to 200 mV would be sufficient to minimize the direct emission of CH_4 and N_2O. Is this range the best one for Eh control? This range would be effective in mitigating the direct CH_4 and N_2O emission. However, with this range, N_2O production would occur in the flooded soil, and dissolved into the percolating water. Therefore, further studies are needed to determine the best Eh range that minimizes the direct and dissolved GHG emission from an irrigated paddy field.

Lastly, we will evaluate the effects of Eh control on the soil carbon budget. Table 14.6 shows the soil carbon budget under the different conditions of water management and rice straw application. Quantitatively, CO_2 exchanges predominated in the soil carbon budget with a little amount of CH_4 emission. The effect of water management on the total budget was fairly low. Of course, water management partially changed the CO_2 exchanges, but these were offset by each other. Considering

Table 14.6 Soil carbon budget of the four treatments in 2004 ($g\,C\,m^{-2}\,y^{-1}$)

	Component and budget		CF+S	CF−S	EH+S	EH−S
CO_2	Fallow period					
	Straw incorp.	A	−199	0	−199	0
	Ratoon	B	−24.7	−24.7	−24.7	−24.7
	Weed	C	−2.9	−2.9	−2.9	−2.9
	Soil resp.	D	254	209	254	209
	Fallow sub-budget	E	27	181	27	181
	Growing period					
	Grain	F	−267	−259	−279	−255
	Straw	G	−254	−267	−270	−261
	Root	H	−78.3	−77.5	−82.7	−80.1
	Stubble	I	−24.5	−25.9	−26.1	−25.3
	Dead straw†	J	−32.8	−33.1	−34.6	−32.7
	Exudates†	K	−20.3	−20.5	−21.4	−20.2
	Algae†	L	−25.7	−25.7	−20.6	−21.1
	Weed	M	−0.04	−0.04	−0.04	−0.04
	Flush	N	37	21	89	47
	Diffusion†	O	187	132	150	108
	Water†	P	17.5	17.5	17.5	17.5
	Growing sub-budget	Q	60	−12	71	−7
	CO_2 budget of soil	R	87	169	98	174
			(319)	(620)	(359)	(638)
CH_4	Growing period	S	17.04	18.90	5.24	4.84
			(523)	(580)	(161)	(148)
Soil carbon budget		T	104	188	103	179
			(842)	(1200)	(520)	(786)

Source: Minamikawa and Sakai (2007b).
CF – continuous flooding, EH - Eh control.
$T = R+S = (E + Q) + S = [(A + B + C + D) + (H + I + J + K + L + M + N + O + P)] + S.$
In the parentheses, CO_2 equivalent considering the GWP ($g\,CO_2\,m^{-2}\,y^{-1}$).
†Estimated value from the previous studies.

GWP of CO_2 and CH_4, quantitative significance of Eh control naturally increased and largely contributed to reduced GHG emission.

Consequently, the combination of Eh control and straw application is the ideal one among four treatments to decrease the GWP and sustain soil carbon storage. However, there is a need to take help of other practices, such as continuous organic matter application for distinguished effects on the soil carbon sequestration.

14.5 Conclusions

Is it possible to mitigate the total GHG emission from irrigated paddy fields by water management? The answer for the subject of this chapter may be ‘partly yes’ at the present research level. As for the direct GHG emission, the water management using an objective indicators, such as AWD and Eh control, have significant effects on the decrease in CH_4 emission. The concept of Eh control can be also applied to the mitigation of N_2O emission. As for the dissolved GHG emission, further studies are needed, especially on the problems that where GHGs are produced and how many GHGs are emitted. With the answer to these problems, the total mitigation of CH_4 and N_2O emission can be achieved by Eh control.

It is evident that decrease in the direct CH_4 emission has partly contribution to the total GHG mitigation in paddy fields. However, no one knows how much contribution it has. One of the goals for the GHG study in agro-ecosystems will be the integrated evaluation of the total GHG emission considering the GWP and soil carbon sequestration based on the concept of the Life Cycle Assessment (LCA).

At the present, there is no best water management available considering all the concerns. It will be worked out by the integrated evaluation, such as LCA and cost analysis, in the future. On the different backgrounds, several methods of water management based on an objective indicator have been developed. The ideal water management depends on the various situations. Therefore, it is necessary to prepare adequate solutions for probable scenarios of the future world.

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Chapter 15
Mitigating Greenhouse Gas Emission from Agriculture

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15.1 Introduction

Radiative forcing of Earth's atmosphere is increasing at unprecedented rates, largely because of increases in the concentrations of atmospheric trace gases, mainly carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) – collectively known as greenhouse gases (GHG). Concentrations of CO_2, CH_4 and N_2O have increased markedly as a result of human activities since 1750 and now far exceeded pre-industrial values as determined from ice cores spanning thousands of years (Table 15.1). The atmospheric concentrations of CO_2 and CH_4 in 2005 have exceeded the natural range over the last 650,000 years (IPCC 2007). The global atmospheric concentration of CO_2 has increased at an annual growth rate of 0.5%, while that of CH_4 at 0.6% and nitrous oxide at 0.25%. Agriculture plays a major role in the global fluxes of each of these gases and is considered as one of the major anthropogenic sources (Fig. 15.1). Agriculture comprises several activities, contributing to GHG emissions and globally, the most significant activities identified include (i) deforestation and other land-use changes as a source of CO_2, (ii) rice-based production systems (including rice-wheat rotation) as sources of CH_4 and N_2O (and also source of CO_2 due to burning of agricultural residues) and (iii) animal husbandry as a source of CH_4.

15.2 Global Emission Trends

With an estimated global emission of non-CO_2 GHGs over 5969 Mt CO_2 yr^{-1} in 2005, agriculture is estimated to account for about 14% of total global anthropogenic emissions of GHGs and 47 and 84% of total anthropogenic CH_4 and N_2O emissions, respectively (US-EPA 2006). CH_4 emission from enteric fermentation and N_2O emission from soils constitute the largest sources, accounting for 44 and

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Table 15.1 Major anthropogenic greenhouses from agriculture and their properties

Greenhouse gas	Pre-1750 concentration (ppmV)	Current tropospheric concentration (ppmV)	Atmospheric life time (years)	GWP (100 years time horizon)	Increased radiative forcing (W.m^2)
Carbon dioxide (CO_2)	280	379	Variable	1	1.46
Methane (CH_4)	0.715	1.774	12	23	0.48
Nitrous oxide (N_2O)	0.270	0.319	114	296	0.15

Source: IPCC (2007)

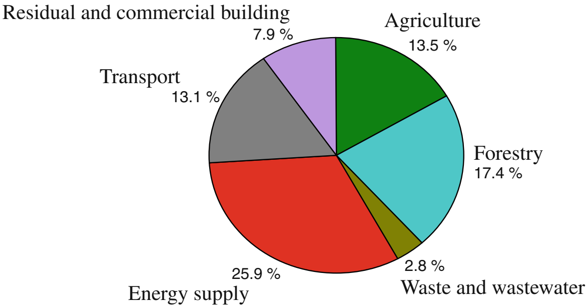

Fig. 15.1 Shape of different sectors in total anthropogenic GHG emission in 2004 in terms of CO_2 equivalents (Source: IPCC 2007)

31% of total non-CO_2 emissions in 2005. Besides, rice production, manure management and biomass burning contribute 11, 7 and 7%, respectively, to CH_4 annual emissions. Emissions of CO_2, mainly from land use change, especially deforestation for agricultural purposes, are estimated to account for 15% of anthropogenic CO_2 emissions (FAO 2003). However, a reliable assessment is difficult due to large spatial and temporal variability as well as concomitant emission and consumption of these gases over a large geographical area of the world.

Globally, agricultural emissions have increased by 14% from 1990 to 2005 with an average annual emission of 49 Mt CO_2 eq. yr^{-1} (US-EPA 2006). N_2O from soils and manure management and CH_4 from enteric fermentation were the agricultural sources, showing the highest increase in emissions at 21, 18 and 12%, respectively. N_2O emissions increased by 31 Mt CO_2 yr^{-1}, which is almost twice the rate of increase for CH_4 emissions. US-EPA forecasts acceleration in the global GHG emissions from agriculture for the period 2005–2020. In the developing countries, the growth is expected to continue at the same rate as in 1990–2005, whereas in the more developed regions, the decreasing trend would be reversed and emission would grow by 8% upto 2020 (US-EPA 2006). Two most significant sources, N_2O from soils and CH_4 from enteric fermentation, would also increase quite rapidly. N_2O emission, which is expected to an average of 49 Mt CO_2 yr^{-1}, would continue to grow faster than CH_4 emissions, projected to an average of 35 Mt CO_2 yr^{-1}.

15.3 Mitigation Options

Specific management options can be used to reduce agriculture's environmental impacts. Conservation practices, that help prevent soil erosion, may also sequester soil C and enhance CH_4 consumption. Managing N to match crop demands can reduce N_2O emission, while manipulating animal diet and manure management can reduce both CH_4 and N_2O emission from animal husbandry. Thus, all segments of agriculture have the management options which can reduce agriculture's GHG footprints. Opportunities for mitigating GHGs in agriculture can be grouped into three broad categories based on the following principles:

a. *Reducing emissions*: The fluxes of GHGs can be reduced by managing more efficiently the flows of carbon and nitrogen in agricultural systems. The exact approaches, that best reduce emissions, depend on local conditions and therefore, vary from region to region.
b. *Enhancing removals*: Agricultural ecosystems hold large reserves of C, mostly in soil organic matter. Any practice, that increases the photosynthetic input of C or slows the return of stored C via respiration, will increase stored C, thereby 'sequestering' C or building C 'sinks'.
c. *Avoiding emissions*: Using bioenergy feed-stocks would release CO_2-C of recent origin and would, thus, avoid release of ancient C through combustion of fossil fuels. Emissions of GHGs, can also be avoided by agricultural management practices that forestall the cultivation of new lands.

15.4 Mitigation Practices for Different Greenhouse Gases

Several practices have been adopted to mitigate GHG emissions based on the principles cited above. Often a practice would affect more than one gas, by more than one mechanism, sometimes in contradicting ways so that the net benefit depends on the combined effects on all gases (Robertson and Grace 2004). Further, the temporal pattern of influence may vary among practices or among gases for a specified practice. The impacts of various mitigation practices considered are summarized in Table 15.2.

15.4.1 Carbon Dioxide

Historically, land-use, land-use change and forestry (LULUCF), residue burning and emission and removal of C from soils are known to positively contribute to the global budget of CO_2. Agriculture also consumes fossil fuels during the manufacture of equipments, fertilizers and other chemical inputs as well as during machinery and grain handling operations (e.g. grain drying). Strides have been made to change

Table 15.2 Proposed measures for mitigating GHG emissions from agricultural ecosystems and their apparent effect on reducing emissions of individual gases

		Mitigative effects		
Mitigation measures	Processes	CO_2	CH_4	N_2O
Cropland management	Agronomy	+		±
	Nutrient management	+		+
	Tillage/residue management	+		±
	Water management (irrigation/drainage)	±	+	+
	Rice cultivation		+	±
	Agroforestry	+		±
	Land-use change	+	+	+
Pasture land management	Grazing intensity	±		±
	Fertilization	+		±
	Nutrient management	+		±
	Range fire	+		±
	Species introduction	+		±
Organic soil management	Avoid drainage of wetlands	+	-	±
	Erosion control, organic amendments, nutrient amendment	+		±
Livestock management	Improved feeding		+	
	Dietary additives		+	
	Animal breeding		+	
Manure management	Improved storage and handling		+	±
	Anaerobic digestion		+	±
	Efficient use as nutrient source	+		±
Bioenergy	Energy crops, residues	+		±

'+' denotes reduced emission or enhanced removal (positive mitigating effect), '−' denotes negative mitigating effect, '±' denotes uncertain or variable effect.

agriculture from a net contributor to GHG emission to a net sink via C sequestration (Reicosky et al. 2000).

a. *Tillage and residue management*: Advanced crop growing techniques now allow many crops to be grown with minimal tillage (reduced tillage) or without tillage (no-tillage). Since soil disturbance tends to stimulate soil C losses through enhanced decomposition and erosion, reduced or no-tillage agriculture often results in soil C gain (Ogle et al. 2005). At least, in the short-term, tillage induces CO_2 emission proportional to the volume of soil distributed (Reicosky and Archer 2007). During the course of 14 years, all tillage and cropping treatments lost SOC compared to the initial SOC levels, while conservation tillage and no-tillage lost the least (Huggins et al. 2007). Collectively, these results suggest that C sequestration due to no-tillage or conservation/reduced tillage depends on depth of soil sampling, crop management and duration of continuous low-intensity tillage system.
b. *Water management*: About 18% of the world's croplands now receive supplementary water through irrigation (Millenium Ecosystem Assessment 2005). Expanding this area or using more effective irrigation measures can enhance C storage in soils through enhanced yields and residue returns (Lal 2004). In a

long-term experiment, the long period of soil sub-mergence under rice cultivation conferred recalcitrant character to the SOC, leading to its stabilization in non-labile pools which results into an enrichment of the SOC stock and restriction to the gaseous C loading into the atmosphere (Mandal et al. 2008). However, some of these gains might be offset by CO_2 from energy used to deliver the water (Mosier et al. 2005).

c. *Land-use change*: One of the most effective methods of reducing emissions is to allow or encourage the reversion of cropland to another land cover, typically one similar to the native vegetation. Such land cover change often increases storage of C, e.g. converting arable cropland to grassland typically results in the accrual of soil C owing to lower soil disturbance and reduced C removal in harvested products (Paustian et al. 2004). Similarly, converting drained croplands back to wetland can result in rapid accumulation of soil C (removal of atmospheric CO_2). Planting trees can also reduce emissions and may also increase the soil C sequestration (Mutuo et al. 2005). However, since land conversion comes at the expense of loss of agricultural productivity, it is usually an option which banks on surplus agricultural land or crop lands of marginal productivity.
d. *Bioenergy*: Agricultural crops and residues are being increasingly seen as sources of feedstocks for energy to displace fossil fuels. These products can be burned directly, but are often processed further to generate liquid fuel (Richter 2004). Biomass fuels are C-neutral, because they release recently fixed CO_2 (via photosynthesis) which does not shift the C-cycle. The net benefit of atmospheric CO_2, however, depends on energy used in growing and processing bioenergy feedstock (Spatari et al. 2005).

15.4.2 Methane

Methane contributes about 20% of the estimated anthropogenic radiative forcing, second to CO_2 which contributes 60% (Lassey 2007). Enteric fermentation by ruminants, rice cultivation, anaerobic waste processing and manure managements are the principal sources of CH_4 from agriculture. US-EPA (2007) estimated that about 28% of CH_4 emission was the result of livestock products. However, a certain amount of CH_4 is also consumed by microorganisms in aerobic soils by oxidation, the only known biological sink.

a. *Rice cultivation*: Methane emission from rice fields to the atmosphere is controlled by CH_4 production (methanogenesis), CH_4 oxidation (methanotrophy) and CH_4 transport process (Aulakh et al. 2000a; Kruger et al. 2001). Thus, strategies to reduce CH_4 emission from rice cultivation may be targeted at (i) reducing CH_4 production, (ii) increasing CH_4 oxidation and (iii) reducing CH_4 transport through plants. CH_4 production from flooded paddy is closely associated with rice growing and different cultural practices, commonly used in rice cultivation, can influence CH_4 production and its emission. Rice cultivation, considered as one of the most important anthropogenic sources of CH_4 emission, also appears to be the most suitable candidate for reducing CH_4 in the atmosphere because of the possibility of controlling emissions by selected agronomic practices (Neue and Roger 1993; Wassmann et al. 1993, 2004). Worldwide researches indicated that water management (Mishra et al. 1997), organic amendment (Rath et al. 1999; Bharati et al. 2000), fertilizer management (Adhya et al. 1998; Rath et al. 1999; Babu et al. 2006) and rice cultivars (Adhya et al. 1994; Shalini et al. 1997; Satpathy et al. 1998; Mitra et al. 1999) affect the flux of CH_4 from this economically important agro-ecosystem and can be suitably manipulated for mitigation of CH_4 emission from flooded rice fields (Table 15.3). With the current technology, improvements in management of water and nutrients and other cultural practices could substantially reduce CH_4 emission from rice cultivation. The total potential for reducing CH_4 emissions from agriculture amounts to 24–92 Tg yr^{-1} (15–65% of current levels) with potential reduction from rice cultivation amounting to 8–35 Tg yr^{-1}, depending on effectiveness of proposed options and degree of implementations (Cole et al. 1997). Potential technologies include water management, fertilizer management, cropping pattern, varietal/cultivar selection and the use of selective inhibitors (Table 15.4).

b. *Manure management*: CH_4 is produced from anaerobic decomposition of animal manure in slurry pits, solid manure piles and from moist soil following incorporation of manure (Lassey 2007). Methane emission from swine lagoons from North Carolina was 8–62 Kg CH_4 ha^{-1} (Sharpe et al. 2002). In a series of four lagoons designed to successively purify water from a swine production facility, the gas flux from Lagoon I, directly receiving animal waste was dominated by CH_4 (79% of total gas flux), while the gas flux in subsequent lagoons were dom-

Table 15.3 Impact of cultural practices on CH_4 emission from rice fields (Data from field experiments conducted at Central Rice Research Institute, Cuttack, India)

Cultivation practice	Rice variety	Treatment	Seasonal flux of CH_4 (kg.ha^{-1})	Grain yield (Mg.ha^{-1})	% Change
Stand establishment	Gayatri	Transplanted	30.22	5.4	–
		Direct seeded	24.36	4.2	−19.40
Crop spacing	Ratna	Close spacing	28.34	2.7	–
		Wide spacing	26.22	2.8	−7.50
Ratooning	IR-36	Main crop	11.22	3.8	–
		Ratoon	15.71	1.4	40.01
Crop sequence	–	Rice-Rice	39.06	–	–
		Rice-upland crop	12.52	–	−68.67

Table 15.4 Estimated effect of management practices on mitigation of CH_4 from flooded fields plants to rice

Mitigation practice	Estimated decrease	Field potential (Tg CH_4.m^{-2}.h^{-1})
Water management	5.0 (3.3 – 9.9)	∼30%
Nutrient management	10.0 (2.5 – 15.0)	∼20%
Cultural practices	5.0 (2.5 – 10.0)	∼20%

inated by N_2 with smaller quantities of CH_4 (Sharpe et al. 2002). A portion of CH_4 emitted to the atmosphere can be sequestered by aerobic soils. Thus, land application of manure could significantly decrease the net quantity of CH_4 emitted to the atmosphere compared to stockpiling or long-term storage of manure. Manure applied to pasture land did not appear to impact CH_4 emission (Chadwick et al. 2000). On the other hand, net uptake of CH_4 under corn from moldboard plowed soils amended with manure was reduced relative to soils without manure.

c. *Animal management*: Livestock sources of CH_4 are predominantly enteric (i.e. from the breath of ruminants and flatus of monogastric animals) as a result of feed digestion. Globally, enteric production of CH_4 was estimated at $\sim 80\,\text{Tg.yr}^{-1}$, which was 20–25% of the observed increase in atmospheric CH_4 concentration (Lassey 2007). Methane emission can be reduced by feeding more concentrates, normally replacing fodder (Beauchemin and McGinn 2005). Although concentrates might increase daily CH_4 emission, emissions per kg feed intake and per kg product are reduced due to better efficiency. Other practices that can reduce CH_4 emissions include: (a) adding oils to the diet, (b) improving pasture quality and (c) optimizing protein intake. While dietary supplements in the form of ionophores, halogenated compounds, propionate precursors have been tested, their efficacy is doubtful and their overall environmental impact is also under cloud.

d. *Consumption of CH_4 by soil*: Soils have the potential to consume CH_4 by the activities of methanotrophic bacteria which constitute the only known net biological sink for atmospheric CH_4 and terrestrial emissions. Methanotrophic bacteria are able to oxidize CH_4 for energy purposes or for building up of microbial biomass (Hanson and Hanson 1996). Methane uptake is controlled by the interplay of biotic and abiotic factors providing proximate limitation on CH_4 oxidation. Methane oxidation in the rice fields is assumed to consume about one-third of the CH_4 production (Bosse and Frenzel 1997), although values as high as 90% have also been reported (Schutz et al. 1989). There is an evidence that agricultural practices have adverse effects on the CH_4-oxidizing ability of soils (Arif et al. 1996; Kessavalou et al. 1998; Hutsch 2001). Cultivation appears to decrease net CH_4 consumption, as CH_4 oxidation potentials of cultivated soils are less than in grasslands (Mosier et al. 1996; Kessavalou et al. 1998). Nitrogen fertilization has also been identified among other factors as an important contributor to this effect. In many cases, NH_4 is the most detrimental form of N for CH_4 oxidation (Bronson and Mosier 1994). In a long-term fertilization experiment, CH_4 consumption was significantly lowered after application of mineral N. On the contrary, stimulation of CH_4 oxidation by NH_4-based fertilizers in soil and around rice roots has also been reported – both in microcosm (Bodelier et al. 2000) and under field conditions (Kruger and Frenzel 2003). It was suggested that elevated $CH_4/NH_4{}^+$ ratio in the rooted soil greatly reduces the inhibitory effect of $NH_4{}^+$ (Cai and Mosier 2000). Methane oxidation in an alluvial soil planted to rice under a long-term fertilization experiment was stimulated following the application of mineral fertilizers or compost, indicating nutrient limitation as one of the factors affecting the process (Nayak et al. 2007). Combined application of compost and mineral fertilizer, however, inhibited CH_4 oxidation probably due to N immobilization by the added compost.

15.4.3 Nitrous Oxide

Nitrous oxide is emitted from agricultural soils by soil microbial processes of nitrification (aerobic transformation of ammonium to nitrate) and denitrification (anaerobic transformation of nitrate to N_2 gas (Sahrawat and Keeney 1986; Monteny et al. 2006). The main cause of increases in agricultural N_2O emission to the atmosphere is the application of N fertilizers and animal manure management. The major factors controlling soil nitrification-denitrification are soil pH, texture, organic C supply, crop residue management, temperature, soil N content, soil aeration and water status and certain agri-chemicals (Sahrawat and Keeney 1986).

a. *Animal manure management*: Animal manures can release significant amount of N_2O during storage, but their magnitude varies. Preliminary evidences suggest that covering manure heaps can reduce N_2O emissions (Chadwick 2005). For most animals worldwide, especially the grazing ones, there is limited scope for manure management, as excretion happens in the field. However, emissions from manure may be curtailed to a limited extent by altering the feeding practices (Kulling et al. 2003), but these mechanisms and the system-wise influence have not been widely explored.

   Rotz (2004) outlined several management options to reduce N loss from animal manure management. Management should focus on improving N-use efficiency of animals to reduce N excretion, retaining N contained in manure until it is applied to land and applying the appropriate amount of manure in a timely manner to enhance crop uptake. Nitrous oxide emission from livestock faeces deposited on pasture is dependent on rainfall, quantity and frequency of N-inputs from stocking rate and soil organic C level (Saggar et al. 2007).
b. *N_2O emission management from agricultural fields*: In general, N_2O emission increases with increased N-inputs (Gregorich et al. 2005). The proportion of applied N emitted as N_2O has been estimated at 1.25% (IPCC 1997). Both fertilized and unfertilized soils emit N_2O. While fertilizer-N is a source of N_2O in case of fertilized soils, mineralization of soil organic-N contributes to the production of N_2O from unfertilized soils (Aulakh et al. 2000b). The scope of different management practices mitigating N_2O emission from croplands and their field potential is listed in Table 15.5. McSwiney and Robertson (2005) reported that N_2O fluxes were low to moderate until the N-input exceeded crop needs, after which the flux nearly doubles, suggesting that prudent management of N-inputs can be an effective strategy to minimize N_2O emitted from croplands. Emission of N_2O after application of anhydrous ammonia was 2–4 times higher than surface applying urea, ammonium nitrate or broadcasting urea (Venterea et al. 2005).

Table 15.5 List of practices to improve fertilizer and manure N-use efficiency in agriculture and expected reduction of N_2O emissions assuming global application of mitigation practices (Mosier et al. 1998)

Practice followed	Estimated decrease in N_2O emissions (Tg.yr^{-1})	Field potential
1. Match N supply with crop demand • Use soil/plant testing to determine fertilizer N needs • Minimize fallow period to limit mineral N accumulation • Optimize split application schemes • Match N application to reduce production goals in region of crop over-production	0.24	~50%
2. Tighten N flow cycle • Integrate animal and crop production systems in terms of manure reuse in plant production • Maintain plant residue N on the production site	0.14	~80%
3. Use advanced fertilization techniques • Controlled release fertilizers • Place fertilizers below the soil surface • Foliar application of fertilizers • Use nitrification inhibitors • Match fertilizer type to seasonal precipitation	0.15	~50%
4. Optimize tillage, irrigation and drainage	0.15	~40%

Fertilizing with ammonium fertilizers, like urea, increased the potential for ammonia emission (Harrison and Webb 2001), but under anaerobic flooded soil, it could minimize gaseous N emissions *via* denitrification (Aulakh 1989).

N_2O emission from field crops is strongly related to the moisture status of the soil. Drying conditions affect nitrification favour low N_2O production, but when aerobic periods are followed by irrigation/flooding, large N_2O fluxes are observed. The N_2O emissions often increase with increasing aeration (decreasing water-filled pore space) during drainage of anaerobic rice soils. Cai et al. (1997) found a very small N_2O flux when the rice paddy plots were flooded, but it peaked at the beginning of the disappearance of floodwater, suggesting a trade-off between CH_4 and N_2O emissions. Substantial N_2O emission can occur during freeze-thaw events (Gregorich et al. 2005). Even though the soil temperatures may be near 0°C, the emission of N_2O is due to microbial activity (Chang and Hao 2001) and the production of N_2O exceeds its reduction to gaseous N_2 at low temperature, thus contributing to N_2O emission during spring-thaw events (Holtan-Hartwig et al. 2002).

15.5 Trade-Offs and GWP

Assessing the impact of agriculture on global climate change requires converting emission data to GWP. Cole et al. (1997) estimated that agriculture has the potential to reduce radiative forcing from 1.2 to 3.3 Pg CO_2-C eq. yr^{-1}. It was estimated that about 32% could be from reduced CO_2 emissions, 42% from C offsets through biofuel production on 15% of the existing croplands, 16% from reduced CH_4 emissions and 10% from reduced emission of N_2O.

A full-cost accounting of the effects of agriculture on greenhouse gas emissions is necessary to quantify the relative importance of all mitigation options. Such an analysis shows nitrogen fertilizer, agricultural liming, fuel use, N_2O emissions and CH_4 fluxes to have additional significant potential for mitigation (Robertson and Grace 2004). Net GWP calculations should take into consideration the sum of net GHG emission after deducting the biological consumption and chemical decay, biomass production and ideally net changes in soil organic carbon (SOC). Lal (2007) estimated that SOC sequestration potentially could offset $\sim$15% of the global CO_2 emission. There are potential trade-offs between SOC sequestration and GHG emission, as conservation tillage enhanced N_2O emission. Mosier et al. (2005) reported that SOC storage relative to total emission determined whether a site would provide a net increase or decease in GWP. Their comparisons included GWP from farm operations (planting, harvesting and applying pesticides), fertilizer, liming, irrigation, N_2O, CH_4 and change in surface SOC (0–5 or 0–7.5 cm).

Calculating net GWP appears simple, sum the GHG emission from all sources (soil, energy use etc.) and deduct the sum of total GHG consumption (C sequestration, methane consumption etc.). However, such measurements are fraught with high spatial and temporal variability. Mosier et al. (2006) compared two methods of estimating net GWP, one based on SOC change (0–7.5 cm) and the other based on soil respiration. The two calculation methods resulted in highly different estimates of GWP, both qualitative and quantitative. It is thus important that while reporting GWP, assumptions and calculations are carefully and clearly delineated.

15.5.1 Policy Issues on Agricultural GHG Mitigation

Recent study has shown that there is a significant economic potential for GHG mitigation in agriculture, with total potentials of 1900–2100, 2400–2600 and 3100–3300 Mt CO_2 yr^{-1} at carbon prices of 0–20, 0–50 and 0–100 US $ per ton CO_2-eq., respectively (Smith et al. 2007). About 70% of this potential arises from developing countries with a further 10% from countries with economies in transition (Trines et al. 2006). Despite such significant economic potential, there are several barriers that could prevent the implementation of these measures. Many of these barriers are particularly prevalent in developing countries and include economic, risk-related, political, logistical and education as well as societal barriers.

a. Economic barriers include the cost of land, competition for land, continued economic penury, lack of existing capacity, low price of carbon, population growth, transaction costs and monitoring costs.
b. Risk related barriers include the delay on returns from investment, issues of stability (particularly of C sinks) and issues related to leakage and natural variation in C sink strength.
c. Political barriers include unclear policies on land use planning and the lack of clarity in carbon/GHG accounting rules and overall a lack of political will.
d. Logisitical barriers include scattered nature of land holdings and conflict of interest among landowners, accessibility to large areas and biological suitability of the land areas for GHG farming.
e. The educational and societal barriers include newer legislations governing the sector, stakeholders' perception and the persistence of traditional practices.

Maximizing the productivity of existing agricultural land and applying best management practices would help to reduce greenhouse gas emissions (Smith et al. 2008). Ideally, agricultural mitigation measures need to be considered within a broader framework of sustainable development. Policies to encourage sustainable development will make agricultural mitigation in developing countries more achievable. The barriers to implementation of mitigation actions in developing countries need to be overcome, if we are to realize even a proportion of the 70% of global agricultural climate mitigation potential that is available in these countries.

15.6 Research Needs

Although GHG emission derived from soil has been researched for several decades, there are still geographic regions and agricultural systems that have not been well characterized. There is an urgent need to estimate GWP across a wide range of agricultural systems. Ideally, a standard or benchmark method to calculate GWP should be established. Methodology to improve the accuracy of determining changes in SOC and GHG emissions would reduce the uncertainty of estimating GWP. Similarly, farmers' participation appears indispensable for technology transfer of any kind, including management changes aimed at sustainable production systems. It is essential to initiate dialogue with the farmers and other stakeholders about GHG concerns and the agricultural practices that would help in mitigating the menace, through various routes:

(i) Improving the understanding of farmers' perceptions and decision making to classify different target groups for specific mitigation strategies.
(ii) Conducting research on farmers' fields or community areas (instead of research stations) as a 'reality check' for suggested improvements.

(iii) Developing alternative management options in close collaboration with farmers preferably derived from indigenous knowledge on sustainable management practices.
(iv) Focusing on farm households rather than individual production systems and evaluating the economic benefit to the farmer, e.g. affordability versus profitability.
(v) Packaging scientific knowledge in practical and user-friendly forms through easy decision-support tools.
(vi) Establishing continuous feed-back on mitigation strategies over longer time spans, e.g. farmers' perception on water pricing may vary according to weather events.
(vii) Educating farmers and rural communities by knowledge initiatives.

15.7 Conclusion

There are significant opportunities for mitigation of GHG in agriculture, however, there are large uncertainties and it is difficult to assess the effectiveness of GHG mitigation measures under the changing environmental conditions. Many recent studies have shown that actual levels of GHG mitigation are far below the technological potential for these measures. However, several barriers, mostly economical, and lack of political will, act as deterrent to achieve this technological potential. The estimated biophysical potential of approximately 5500–6000 Mt CO_2 yr^{-1} would not be realized due to these constraints. With appropriate policies – education and policy initiatives, it may be possible for agriculture to make a significant contribution to climate mitigation by 2030.

Many agricultural mitigation options have both co-benefits (in terms of improved efficiency, reduced cost and environmental benefits) and trade-offs. Many agricultural GHG mitigation options could be implemented immediately without any further technological development, while a few options are still undergoing technological validation. It is important that policy planners understand the issue of climate change vis-à-vis GHG mitigation measures or potential opportunities and get motivated to act and analyze the costs and benefits of mitigation actions. The long-term outlook for GHG mitigation in agriculture suggests that there is a significant potential, but many uncertainties, both price and non-price related, will determine the level of implementation.

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Chapter 16
Attenuating Methane Emission from Paddy Fields

S.N. Singh, Larisha Tyagi and Sadhna Tiwari

16.1 Introduction

Rising trend of earth's surface temperature is today a global threat to mankind. This trend is directly linked to an increasing atmospheric abundance of various greenhouse gases, like CO_2, CH_4, N_2O etc. emanating from man-made activities (IPCC 2007). Among these gases, CH_4 is the most abundant carbon species present in the atmosphere (mixing ratio $\sim$1.8 ppm). Being a highly radiatively active gas, it is a major component of the natural gas after CO_2, accounting for about 20% of the global greenhouse effect (Wuebbles and Hayhoe 2002). Being highly reactive, CH_4 also affects the chemistry and oxidation capacity of the atmosphere by influencing the concentrations of tropospheric ozone, hydroxyl radicals and carbon monoxide (Cicerone and Oremland 1988). Ozone formation further amplifies the methane induced greenhouse effect by approximately 70% (Moss 1992). Global atmospheric concentration of CH_4 has increased from a pre-industrial value of about 715 ppb to 1745 ppb in 1998, and to 1774 ppb in 2005 (IPCC 2007). Once emitted, CH_4 remains in the atmosphere for approximately 8.4 years before removal (Dentener et al. 2003). Although atmospheric abundance of CH_4 is far less than 0.5% of CO_2, but on molecule to molecule basis, it is approximately 23 times more effective in absorbing infrared radiations than CO_2 (IPCC 2007). Dlugokencky et al. (2003) observed that atmospheric methane had been at a steady state of 1751 ppbv between 1999 and 2002 (Fig. 16.1). However, over the last two decades, the concentration of CH_4 in the troposphere is reportedly increasing at the rate of $\sim$ 0.7% each year and is anticipated to modify the global climate, affecting terrestrial ecosystem both functionally and structurally (Houghton et al. 1996).

Interestingly, in the early 1990's, the rate of CH_4 increase has declined markedly from 1.1% to approximately 0.5% per year (Dlugokencky et al. 1994, 1998). The steep increase in CH_4 concentration, followed by a decline in 1990s (Gupta

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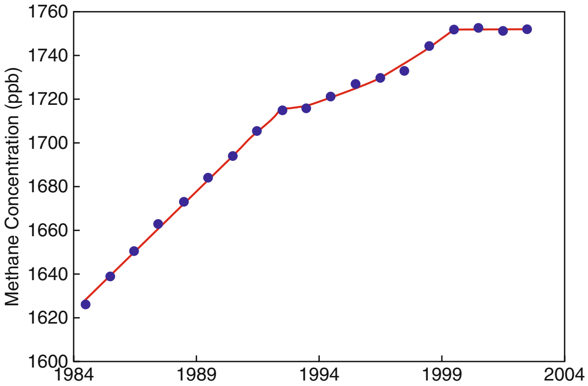
Fig. 16.1 Atmospheric methane concentration



et al. 1996) seems to be related to changes in atmospheric chemistry and temperature induced by the Pinatubo eruption (Dlugokencky et al. 1996).

16.2 Sources

Methane is released into the atmosphere by a number of anthropogenic and natural sources, but anthropogenic emissions used to dominate present-day CH_4 budgets, accounting for more than 60% of the total global budget. The anthropogenic sources include rice agriculture, livestock, landfills and waste treatment, some biomass burning, and fossil fuel combustion, while natural sources are wetlands, oceans, forests, fire, termites and geological sources. Estimates of global CH_4 sources by various workers have been reflected in Table 16.1.

Cultivated wetland rice soils emit significant quantities of CH_4 (Yan et al. 2003). Estimates of rice paddies, as a source of CH_4, range from 20 to 100 Tg yr^{-1} (Houghton et al. 1995), which is equivalent to 5–28% of the total CH_4 from all anthropogenic sources.

Methane emission levels from a source can vary significantly from one country or region to another, depending on many factors, such as climate, industrial and agricultural production characteristics, energy types and usage, and waste management practices. For example, temperature and moisture have a significant effect on the anaerobic digestion process, which is one of the key biological processes that cause methane emissions in both human-related and natural sources. Also, the implementation of technologies to capture and utilize methane from sources, such as landfills, coal mines, and manure management systems, determine the emission levels from these sources.

Table 16.1 Estimates of the global CH_4 sources (Tg CH_4 yr^{-1}) from different references

Reference Base year	Fung et al. (1991) 1980	Hein et al. (1997)	Lelieveld et al. (1998) 1992	Mosier et al. (1998) 1994	Houweling et al. (1999)	Olivier et al. (1999) 1990
Natural sources						
Wetlands	115	237	225[b]		145	
Termites	20	–	20		20	
Oceans	10	–	15		15	
Hydrates	5	–	10		–	
Anthro. Sources						
Energy	75	97	110		89	109
Landfills	40	35	40	80	73	36
Ruminants	80	90[a]	115	14	93	83[a]
Waste treatment	–	[a]	25	25–54	–	[a]
Rice agricultural	100	88	[b]	34	–	60
Biomass burning	55	40	40	15	40	23
Others	–	–	–		20	
Total sources	**500**	**587**	**600**			

Data from IPCC (2001)

– not estimate, blank: no aim to estimate originally

[a] Waste treatment included under ruminants

[b] Rice agriculture included under wetlands

16.3 Rice Paddy as a Source of CH_4

Among the basic needs, food is the foremost necessity of human needs. To meet the growing demand of burgeoning human population, the world's annual rice production must increase from the present 520 million tons to at least 880 million tons by 2025, as rice is a staple food for more than half of the world's population (Lampe 1995). Among the biological sources, paddy fields are a major contributor to CH_4 emission, accounting for about 15–20% of the global atmospheric CH_4. Methan emission from paddy fields of different continents has been reflected in Table 16.2. It is further projected that the CH_4 emission from rice cultivation may increase from 1990 level of 97 to 145 Tg yr^{-1} in 2025 (Anastri et al. 1992), as rice is cultivated world wide between 50° N and 35° S and about 90% of the harvested area is located in Asia. Along with CH_4, CO_2 and N_2O are also emitted from the paddy fields. Tsuruta et al. (1998) calculated the contributions of GHGs (CO_2, CH_4 and N_2O) emissions from Japanese paddy fields to the GWPs with a 100-year time horizon and the effluxes of CH_4, CO_2 and N_2O were accounted for 16.0, 78.2 and 5.8 of the total CO_2-equivalent emission, respectively. Therefore, a decrease in CH_4 emission from paddy fields has far reaching impact on global warming.

Recent studies have estimated CH_4 emission from paddy fields at 53 Tg CH_4 yr^{-1} (Chanton et al. 1997), 25–54 Tg CH_4 yr^{-1} (Mosier et al. 1998), 33–49 Tg CH_4 yr^{-1} (Neue and Sass 1998) or 25.1 Tg CH_4 yr^{-1} (Yan et al. 2003), although studies conducted before 1990 had given higher estimates (Holzapfel-Pschorn and Seiler 1986; Aselman and Crutzen 1989). The methane budget from Indian paddies has been estimated to be an average of 4.3 Tg Yr^{-1} with a range between 2.7 and 5.4 Tg Yr^{-1} in Methane Campaign 1991 (Parashar et al. 1997).

Later on, Methane Asia Campaign (MAC) (1998) computed a CH_4 budget for Indian paddy fields to be 3.61 Tg/yr as depicted in Table 16.3. The recent estimates of paddy emissions account for 4.2–9.0% of the total global CH_4 emission (600 Tg CH_4 yr^{-1}) (Minamikawa et al. 2006). Before that, Kumaraswamy et al. (2000) reviewed CH_4 emission from paddy fields, while Le Mer and Roger (2001) estimated CH_4 production, oxidation, emission and consumption separately in various paddy soils.

Table 16.2 Methane emission from paddy fields of different continents

Ecosystem	Location	Total emission (Tg yr^{-1})
Total emissions		80±59 (Lelieveld et al. 1998)
Irrigated rice	Asia	26–62
	Africa	2–4
	North America and Central America	1–2
	South America	3–6
	Rest	0–1
Rainfed rice (0–100)	Asia	12–25
	Africa	1–2
Rainfed rice (30–100)	Asia	4–11
Deep water rice (>100)	Asia	<1
Uptake		
Upland rice soils	World wide	>20 (Seiler et al. 1984) 20–60 (Reeburg et al. 1994)

Table 16.3 Methane budget estimates from Indian rice paddy fields under Methane Asia Campaign 1998 (co-ordinated by NPL, New Delhi)

Water regime				Harvested area (Mha)	IPCC-96 *Default E_{sif} (gm^{-2})	Indian Seasonal IIntegrated flux E_{sif} (gm^{-2})	Total emission ($Tg\ Yr^{-1}$) *Using IPCC-96 Default E_{sif}
Upland				6.35	0	0	0
Lowland	Rainfed	Food prone		4.23	*16(10–20)	19.0±6.0	0.80±0.25 *0.68 (0.42–0.84)
		Drought prone		6.77	*8(0–10)	6.9±4.3	0.47±0.29 *0.54 (0–0.68)
	Irrigated	Continuously flooded		6.77	*20	15.3±2.6	1.04±0.18 *1.35
		Intermittently flooded	Single aeration	9.92	*10(4–14)	6.9±4.3	0.69±0.43 *0.99
			Multiple aeration	5.74	*4(2–6)	2.2±1.5	(0.40-1.39) 0.13±0.09 *0.23 (0.12-0.34)
	Deep water	Water depth 50–100 cm		2.54	*16(12–20)	19.0±6.0	0.48±0.15 *0.41 (0.31-0.51)-
		Water depth 50–100 cm		–	12(10–16)	–	
	Total Methane Emission	Estimates using Indian CH_4 Estimates data upto 1999 *Estimates using IPCC-96 Default Methane E_{sif}					3.61±1.4 *4.20 (1.25–5.11)

Mha-Million hectares, $Tg = 10^{12}$ g

16.4 Mechanism of Methane Formation

Methanogenesis is the microbial process for formation of CH_4, as the terminal product of anaerobic breakdown of organic matter. Methane is exclusively produced by methanogenic bacteria that can metabolize only in the strict absence of free oxygen and at redox potential of less than $-150\,\text{mV}$ (Wang et al. 1993). In contrast to this concept, Keppler et al. (2006) reported a startling finding that CH_4 could be also produced by the living plants in aerobic conditions without the action of bacteria. This was proven by the laboratory experiments with 30 different kinds of tree leaves and grasses, but they could not explain the processes involved in it. This finding could explain many previous puzzling observations made by a group of physicists of Germany and Paul J. Crutzen, a 1995 Nobel Prize winner and his colleagues about the presence of methane clouds over the tropical forests and Venezuelan savanna, respectively. However, their observations have been questioned by many workers who still believe that CH_4 is the terminal product of anaerobic digestion of organic matter in soils by methanogenic bacteria.

The methanogens are best at work at a pH of 6–8 and a temperature of 30–40°C. These bacteria produce methane (i) by transmethylation or decarboxylation of acetic acid, and (ii) by the reduction of carbon dioxide (Takai 1970). Before methanogens act, acidogens and acetogens break down lipids, proteins, polysaccharides, sugars and amino acids of the organic residues by their extra cellular enzymes in the sediments (Table 16.4).

Table 16.4 Bacteria involved at different stages of CH_4 production

Group	Bacteria	Extra-cellular enzymes	Conversion
Acidogens	**Clostridia, Micrococci**	Lipases	Lipids to long-chain fatty acids-further oxidized to acetyl Co A
	Bacteroides, *Butyrivibrio* *Clostridium,* *Fusobacterium* *Selenomonas,* *Streptococcus* *Campylobacter,* *Peptococcus*	Proteases	Proteins hydrolysed to amino acids further degraded to fatty acids such as acetate, propionate and butyrate
	Bacterial Consortium	Cellulases Amylase Pectinases	Polysaccharides are hydrolysed in synergistic action and fermented to acetate, propionate, butyrate, ethanol, CO_2 and H_2
Acetogens	*Syntrophobacter* *Syntrophomonas* *Clostridium*		Fermentation of sugars and amino acids both derived from acetogenesis and dehydrogenation to highly volatile fatty acids
Methanogens	*Methanosarcina,* *Methanothrix (now Methanosacta)*		Reduction of CO_2 and transmethylation of acetic acid and formic acid

The mechanism of CH_4 formation involves either reduction of CO_2 with hydrogen ions, where fatty acids and alcohols act as hydrogen donor or transmethylation of acetic acid or methyl alcohol, which does not produce CO_2 as an intermediate (Takai 1970; Vogels et al. 1988). Under flooded paddy soils, after the trapped molecular O_2 **is** used, sequential reduction of the soil oxidants ($NO_3{}^-$, $Mn_4{}^+$, $Fe_3{}^+$, $SO_4{}^{2-}$ and CO_2) progresses in accordance with the thermodynamic theory.

$$
\begin{array}{ll}
CH_2O + O_2 \longrightarrow CO_2 + H_2O & -475\,\text{kj} \\
5CH_2O + 4NO_3{}^- \longrightarrow 4HCO_3{}^- + CO_2 + 3H_2O + N_2 & -448\text{kj} \\
CH_2O + 2MnO_2 + 3CO_2 + H_2O \longrightarrow 2Mn^{++} + 4HCO_3{}^- & -349\text{kj} \\
CH_2O + 4Fe\,(OH)_3 + 7CO_2 \longrightarrow 4Fe^{++} + 8HCO_3{}^- + 3H_2O & -114\text{kj} \\
2CH_2O + SO_4{}^{2-} \longrightarrow H_2S + 2HCO_3{}^- & -77\text{kj} \\
2CH_2O + H_2O \longrightarrow CH_4 + CO_2 + H_2O & -58\text{kj}
\end{array}
$$

The production of methane is preceded by the production of volatile acids. Hydrogen evolution follows the disappearance of oxygen following flooding of the soil into water. In this sequence, carbon dioxide evolution is followed by methane formation and emission (Takai et al. 1956). The main electron-acceptors in submerged soils include dissolved oxygen (O_2), NO_3^-, Fe (III), SO_4^{2-}, and CO_2. The final products of reduction in submerged soils are Fe (II) from Fe (III), H_2S from SO_4^{2-} and CH_4 from CO_2.

Carbon substrates for methanogenesis are supplied from applied organic matter, exudates and sloughed tissues of rice plants, and soil organic matter (Watanable et al. 1999; Kimura et al. 2004). A major part of the CH_4 produced is consumed by methanotrophs under oxidative conditions in the rhizosphere of rice plants and in a layer of soil interfacing with the surface water. Besides, CH_4 generation in the sediment is regulated by various edaphic factors like temperature, redox potential, and moisture regime, length of water logging, sulphate and pH (Grűnfeld and Brix 1999; Singh 2001).

In fact, net CH_4 emission is the difference between production and consumption. Processes that regulate the CH_4 efflux from the site of generation to the atmosphere include ebullition, molecular diffusion and vascular transport by plants. Ebullition of CH_4 gas generally occurs when the partial pressure of the entrapped CH_4 within the sediment results in an upward surge of the gas into the atmosphere. While in molecular diffusion, dissolved CH_4 diffuses according to the concentration gradients through the sediment-water and air-water interfaces (Bartlett et al. 1985) and the diffusion of gases in water is approximately 10,000 times slower than in air (Wang et al. 1995). The actual diffusion of CH_4 from rice fields is a function of CH_4 supply to the floodwater, CH_4 concentration in the floodwater and prevailing wind speed (Sebacher et al. 1983). Diffusion through floodwater is usually less than 1% of the total flux (Conard 1993). Most of the CH_4 emitted to the atmosphere is transported through aerenchyma of rice plants by molecular diffusion or by high-low pressure induced flow. The aerenchyma tissue of aquatic plants helps in transport of atmospheric O_2 to the rhizosphere for the root respiration (Nouchi et al. 1990; Schütz et al. 1991 and Nouchi and Mariko 1993) and also facilitates the reverse transport of CH_4 produced in the anaerobic zone of flooded soil to the atmosphere. The path of CH_4 through the rice plants includes diffusion into the root, conversion into gaseous CH_4 in the root cortex, diffusion through cortex and aerenchyma and release to the atmosphere through micropores in the leaf sheath (Nouchi and Mariko 1993; Aulakh et al. 2000).

In the temperate rice fields, more than 90% of CH_4 is emitted through plants (Schütz et al. 1989), while from the tropical rice fields, significant amounts of CH_4 may evolve by ebullition, particularly during the early part of the season and in the case of high organic input (Wassmann et al. 1998). Thus, rice plants influence the methane dynamics in paddy soils by (1) providing substrate in the form of root exudates to methanogens to enhance the production of CH_4; (2) transporting CH_4 from soil to atmosphere (conduit effect), and (3) creating aerobic microhabitat in rhizosphere, which is suitable for growth and multiplication of methanotropic bacteria responsible for CH_4 consumption (Dubey et al. 2000).

16.5 Microbial Oxidation

Soils, that support production of CH_4 in the anoxic condition, also act as a sink for the same in presence of oxygen. Methane oxidation in aerobic condition is carried out by methanotrophs or methane oxidizers. The enzyme responsible for the initial step in CH_4 oxidation is methane monooxygenase (MMO) enzyme, which requires molecular O_2 to perform its activity. Methane monooxygenase (MMO) has two isoenzymes i.e. soluble MMO (sMMO) and particulate MMO (pMMO), which require Cu and Fe metals for their activity (Dalton et al. 1993). The overall reaction for methane oxidation given by Dalton and Hocknall (1990) is as follows:

$$CH_4 + 2O_2 \longrightarrow CO_2 + 2H_2O \; (\acute{C}G0 = -780\,\text{kj mole}^{-1})$$

During different stages of CH_4 oxidation, Whalen et al. (1990) found that a small fraction (19.1±2.3%) of formaldehyde was assimilated by the plants into cellular biomass. Rhizosphere, being oxic zone, works as a major sink for methane in the sediment. The decrease in CH_4 concentration in the rhizospheric zone of some aquatic plants has been reported by Wagatsuma et al. (1992).

16.6 Edaphic Factors Influencing CH_4 Efflux

Yagi (1997) categorized the factors controlling CH_4 emission into four scales (Fig. 16.2). From microbial to global scales, soil properties, rice plant activity, field management and climate are the factors, which modulate CH_4 efflux from paddy fields.

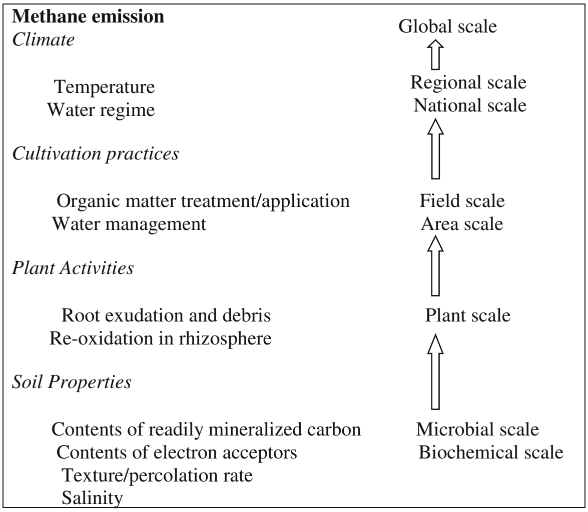


Fig. 16.2 The factors controlling CH_4 emissions from paddy fields on different scales (modified after Yagi 1997; Minamikawa et al. 2006)

Emitted CH_4 from the paddy fields enhances global warming on the largest scale, but the microbiological processes of CH_4 production and oxidation occur on the smallest scale. In terms of environmental factors, temperature (Dunfield et al. 1993; Fey and Conrad 2003) and water regime (Mishra et al. 1997; Rath et al. 1999) determine the potential for CH_4 emission. On the microbial and plant scales, the amount of CH_4 emitted is determined by the population and activity of microorganisms (Mayer and Conrad 1990; Adachi 2001; Eller and Frenzel 2001; Hadi et al. 2001), the physical conditions and chemical properties of the soil (Yao and Zhang 1996; Van den pol-van Dasselar and Oenema 1999; Wang et al. 1999; Watanable and Kimura 1999; Yao et al. 1999), the biomass and activity of rice plants (Chanton et al. 1997) and the kinds and amounts of substrates (Watanable et al. 1999; Kimura et al. 2004).

From the farmer's viewpoint, management on a field scale is the only way to produce rice and increase the rice yield (Minamikawa et al. 2006). Such field management also has physical, chemical and biological effects on CH_4 emission. Statistically during growing period, average methane efflux was found to be significantly affected by water management, organic matter application, and content of soil organic carbon, soil pH, pre-season water status and climate. Therefore, ecological problems on different scales from different view points i.e. CH_4 emission (global warming) and rice production, need to be considered simultaneously.

Irrigated paddy fields were estimated to account for 70–80% of global CH_4 whereas rainfed (about 15%) and deepwater (about 10%) fields had much lower shares (Wassmann et al. 2000).

As evaluated by Wassmann et al. (1999), irrigated rice offers the most options for modifying field management to mitigate CH_4 emission. On the other hand, mitigation options for rain fed and deepwater fields are very limited in both number and potential gain (Wassmann et al. 1999). Therefore, it is certain that irrigated paddy fields are the most promising targets for applying mitigating options.

16.7 Options for Mitigating CH_4 Emission

Reduction in the CH_4 efflux from rice fields can be made either by controlling the production, oxidation or transport processes of CH_4. Since methanogens require highly reducing conditions for their activity (Zeikus 1977; Cicerone and Oremland 1988), arresting the development of soil reduction is one of the most effective ways of decreasing CH_4 production rate in soils. This can be accomplished by aerating soils during the flooding period with altering water management, or by inhibiting the progress of the sequential redox reactions with adding chemicals. These options may enhance the CH_4 oxidation rate in soils. Reducing the amount of labile organic matter in soils by composting organic fertilizer or promoting aerobic decomposition of biomass is another effective way of controlling CH_4 production in soils. Since, rice plants also have a significant contribution on production/oxidation of CH_4 in the rhizosphere and its transport to the atmosphere, selection of rice variety that emits small amount of CH_4 may also mitigate CH_4 emission significantly from rice paddy fields. In summary, the options, that agree with the above strategies, includes altering water management, using application of mineral fertilizer or soil amendments,

Table 16.5 Evaluation of the mitigation options for methane emission form paddy fields

	Problem						
	CH_4 mitigation efficiency	Applicability		Economy		Effects on	
		Irrigated	Rainfed	Cost	Labor	Yield	Fertility
Water management							
Midseason drainage	□	○	●	∼	↑	+	∼
Short flooding	□	○	●	∼	∼	–	–
High percolation	□	○	●	↑	↑	+	∼
Soil amendments							
Sulfate fertilizer	□	○	○	↑	∼	▲	–
Oxidants	□	○	○	↑	↑	▲	–
Soil dressing	○	○	○	↑	↑	–	–
Organic matter							
Composting	□	○	○	↑	↑	+	+
Aerobic decom.	□	○	○	∼	↑	∼	∼
Burning	○	○	○	∼	↑	∼	∼
Others							
Deep tillage	○	○	○	↑	↑	–	–
No tillage	?	○	○	∼	↓	–	∼
Rotation	○	○	▲	∼	↑	–	–
Cultivar	○	○	○	∼	∼	∼	∼

□: very effective; ○: effective/applicable; ▲: case by case; ●: not applicable/require long time; ?: no information; ↑: increase; ↓: decrease; ∼: about equal to previous situation; +: positive; –: negative improving organic matter management, changing tillage, rotation, and selecting rice varieties (Table 16.5).

16.7.1 Water Management

A primary switch turning on the production and emission of methane, is the absence of oxygen in the rhizosphere environment. Removal of oxygen from the rhizosphere is normally through consumption by soil bacteria. The presence of the floodwater impedes the diffusion of oxygen from the atmosphere into the soil and thus keeps it anoxic. Hence, water management is one of the most effective options in decreasing CH_4 emission, because it prevents the development of soil reductive conditions. Water management, such as mid-season drainage and intermittent irrigation, is commonly used in irrigated paddy fields to control surplus tillering and to supply rice roots with molecular O_2 for preventing sulfide toxicity (Kanno et al. 1997). However, the reduction in CH_4 emission achieved through intermittent irrigation varies in a broad range from 7 to 80% and imply a number of constraints due to an inverse effect on N_2O emission, the third most important greenhouse gas. Changes in the soil moisture regime stimulate nitrification (through soil dying) and denitrification (through soil wetting) and thus, enhance emissions of N_2O (Bronson et al. 1997; Zheng et al. 1997 and Abao et al. 2000).

The extent of mitigation is likely to vary appreciably depending on different factors, e.g. soil texture, percolation rate, frequency of drainage, duration of dry period, etc. Field drying at mid-tillering stage has been shown to reduce CH_4 emission by 15–80%, as compared to continuous flooding, without a significant effect on grain yield. The net impact of mid-tillering drainage was diminished when (i) rainfall was strong during the drainage period and (ii) emissions were suppressed by very low levels of organic substrate in the soil (Wassmann et al. 2000).

16.7.1.1 Mid-Season Drainage

Reduction in CH_4 emission by mid-season drainage (aeration) was first observed in the early field measurement in Japan (Yagi and Minami 1990). Mid-season drainage, supplies oxygen into soil, resulting in a reduction of CH_4 production. This practice encompasses a distinct period of one week when irrigation is interrupted. Rate of methane efflux shows a short-term peak at the beginning of soil aeration due to the release of soil–entrapped methane and then followed by persistently low emissions even when the fields are flooded again. A properly timed mid-season drainage appears as a promising option to achieve net–gains in greenhouse gas reduction when the baseline of CH_4 emissions is very high (Wassmann et al. 2000). Single mid-season drainage may reduce seasonal CH_4 emission rates by about 50%, which can be explained by the influx of oxygen into the soil (Kimura 1992; Sass et al. 1992).

CH_4 emission is very sensitive to duration and timing of the drainage period, hence management practice may further be improved to reduce emission, especially in areas where the general concept of mid-season drainage is hardly known as a successful agronomic practice by some farmers. This mitigation practice is, however, not feasible during periods of heavy rainfall and when excess water is not available to flood it again. Therefore, in case of non-availability of water for reflooding, it has limited applicability in time and space. Mid-season drainage supplies oxygen into soil, resulting in a reduced of CH_4 production and induced CH_4 oxidation in soil.

16.7.1.2 Alternate Flooding/Drying

The water regime refers to alternate flooding and aeration (drying) of the soil throughout the vegetation period. Methane emission from this water regime in northern India, are generally very low, but N_2O emissions from this regime vary in a broad range. In contrast to mid-season drainage, the time intervals between wet and dry conditions appear to be too short to facilitate the shift from aerobic to anaerobic soil conditions. Yagi et al. (1994) observed CH_4 emission reduction of approximately 50% in intermittent drying plots.

16.7.2 Nitrification Inhibitors

Nitrification inhibitors (NI) prevent the conversion of NH_4^+-N into NO_3^-–N (Bronson et al. 1991), thereby reducing emissions of N_2O *via* nitrification directly (Majumdar et al. 2000; Kumar et al. 2000) and then conserve by reducing the availability of NO_3^- for denitrification (Aulakh et al. 1984; Bronson et al. 1992). Nitrification inhibitors are also increasingly recommended for rice agriculture to minimize fertilizer N losses (Prasad and Power 1995) by limiting the formation of nitrate from ammonium.

In addition to their role in controlling various processes of N losses, nitrification inhibitors have been also reported to inhibit CH_4 emission from flooded soil planted with rice. Besides, artificially synthesized materials, some plant products have been also found suitable to mitigate N_2O from rice fields (Majumdar et al. 2000).

Nitrification inhibitors, such as dicyanamide (DCD), acetylene and nitrapyrin, mitigated emission of CH_4 gas from rice fields by inhibiting the growth of nitrifers, methanogens and methanotrophs (McCarty 1999). In a study at Cuttack, application of nitrification inhibitor i.e. dicyanamide (DCD), inhibited CH_4 emission, while nimin (a plant product), another nitrification inhibitor, stimulated its emission from flooded rice field. Besides DCD, calcium carbide, ammonium thio sulphate and thiourea have also been shown to inhibit CH_4 emission from flooded soil planted to rice (Bronson and Mosier 1991 and Keerthisinghe et al. 1993). Bronson and Mosier (1991) reported a significant reduction in CH_4 emissions from rice paddies, following the application of urea in combination with encapsulated calcium carbide. Lindau et al. (1993) found that CH_4 emissions from rice fields were decreased by 35 and 14%, following the application of encapsulated calcium carbide and dicyandimide, respectively. According to Keerthisinghe et al. (1993), wax coated calcium carbide and nitrapyrin effected a significant reduction in CH_4 emission from rice paddies. The decrease in CH_4 emissions in plots treated with wax-coated calcium carbide was attributed directly to the slow release of acetylene, a known inhibitor of methanogenesis (Bronson and Mosier 1991).

In a rice field study, Rath et al. (1999) observed that CH_4 emissions were lowest in plots treated with a mixture of prilled urea and Nimin, a nitrification inhibitor which inhibits the autotrophic oxidation of NH_4^+ to NO_2^- (Sahrawat and Parmar 1975). However, the mechanism of the inhibitory action of Nimin on CH_4 emission is not known. On the contrary, nitrifications inhibitors are also known to inhibit CH_4 oxidation (Bronson and Mosier 1994). Kumaraswamy et al. (1997) showed that nitrification inhibitors (thiourea, sodium thiosulphate and dicyandiamide) inhibited the CH_4-oxidizing activity of flooded rice field soil samples. Both surface soil and rhizospheric samples, which harboured higher populations of CH_4-oxidizing bacteria than the sub-surface soil, showed inhibition of CH_4 oxidation.

Several benzene-ring compounds (Patel and Roth 1977) and N-containing compounds (Bollag and Czlonkowski 1973) are also known to suppress methanogensesis in pure cultures and in soils. Chemicals, known to inhibit CH_4 production as well as CH_4 oxidation, include DDT (2, 2-dichlorodiphenyltrichloroethane) (McBridge and Wolfe 1971) and the nitrification inhibitor, acetylene (Sprott et al. 1982). Availability of these specific as well as general inhibitors of microorganisms holds a promise for their use along with chemical fertilizers or other agro-chemicals to mitigate CH_4 emission from rice soils. Bharati et al. (2000) also studied impact of 6 nitrification inhibitors on CH_4 emission and reported reduction in CH_4 emission in the decreasing order: Sodium azide > DCD > Pyridine > aminopurine > ammonium thiosulphate> thiourea (Table 16.6).

Sodium azide, a potent NI, is also microbial respiratory inhibitor, causing a reduction in methane production by 75% over that of unamended control (Aulakh and Rennie 1985). HQ (Hydroquinone) is also found to decrease N_2O as well as CH_4 emission via inhibition of the methanogenic fermentation of acetate (Wang et al. 1991).

Nimin, a plant product, also mitigated methane emission when added to rice fields with prilled urea as compared to pilled urea alone (Rath et al. 2000). Although slow release fertilizers mitigated N_2O emission (Majumdar et al. 2000; Abao et al. 2000), but no significant difference in methane emission was found between

Table 16.6 Effects of various nitrification inhibitors on CH_4 production ($\mu g\,kg^{-1}$) in an alluvial soil under flooded conditions (Bharati et al. 2000)

	Days after flooding							
Treatments	5	10	15	20	25	30	40	Mean
Unamended control	47	126	157	168	573	2929	4426	1204
Sodium azide	40	42	48	58	135	541	1795	380
Aminopurine	56	82	133	140	569	2066	3540	941
Pyridine	52	87	92	130	346	1558	4094	908
Dicyandiamide	39	50	55	63	276	1112	2844	634
Thiourea	95	103	117	110	399	1843	4791	1065
Ammonium thiosulphate	71	90	103	125	491	1443	5098	1060

slow release (Coated urea) and fast release (Compound fertilizer) of N sources (Sugii et al. 1999).

16.7.3 Pesticides

In modern rice culture, pesticides are being increasingly used. Most pesticides at field-recommended doses are considered to be harmless to beneficial soil microorganisms and their activities. However, some pesticides may effect inhibition or stimulation of certain microbial transformations in the rice fields. There is an evidence for the inhibition of nitrification (Ray et al. 1980) and nitrogenase activity (Patnaik et al. 1994) by an organochlorine insecticide, hexachlorocyclohexane (HCH), and stimulation of autotrophic nitrification of NH_4^+ by a carbamate insecticide, carbofuran (Ramakrishna and Sethunathan 1982). However, there is less information available on the effects of pesticides on bacteria involved in the production or consumption of CH_4. The organochlorine insecticide DDT is known to inhibit CH_4 production as well as CH_4 oxidation in the culture media (McBridge and Wolfe 1971). Topp (1993) found that the pesticides, bromoxynil and methomyl as well as nitrification inhibitor, nitrapyrin, were inhibitory to CH_4 oxidation at 50 μgl^{-1}, and the inhibitory effect lasted for <3 weeks. None of these agrochemicals were inhibitory to CH_4 oxidation when applied at a concentration of 5 μgl^{-1}. Arif et al. (1996) reported a negative effect of the herbicide, 2, 4-dichlorophenoxyacetic acid (2, 4-D) on CH_4 oxidation in an arable soil. The degree of inhibition of CH_4 oxidation increased with increasing levels of 2, 4-D, with 100 μg 2, 4-D g^{-1} soil being completely inhibitory. Satpathy et al. (1997) found that the application of a commercial formulation of a widely used organochlorine insecticide, HCH to flooded rice soils, or its technical-grade isomers (α^-, β^-, and δ^-), to laboratory-incubated flooded soils, retarded the production and emission of CH_4, even at a field application rate of 1–2 kg active ingredient (a.i.) ha^{-1} to control insect pests. In contrast, HCH inhibited CH_4 oxidation significantly at 5 $\mu g\, g^{-1}$ soil, and almost completely at 10 $\mu g\, g^{-1}$ soil. Similarly, a commercial formulation of carbofuran, a carbamate insecticide, when applied at rates of 2 kg and 12 kg a.i. ha^{-1} to flooded field planted to rice, resulted in a significant inhibition of CH_4 emissions. Interestingly, in laboratory incubation studies, carbofuran inhibited net CH_4 production when applied at low rates (5 and 10 $mg\, g^{-1}$ soil), but stimulated when applied at a rate of 100 $mg\, g^{-1}$ soil. The oxidation of CH_4 was, however, stimulated by carbofuran when applied at low rates, and inhibited when carbofuran was applied at a rate of 100 $\mu g\, g^{-1}$ soils (Kumaraswamy et al. 1997).

16.7.4 Chemical Inhibitors and Electron Acceptors

Oxygen, alternate acceptors and some chemicals inhibit methanogenesis by different mechanisms. A central electron carrier (co-enzyme F^{420}) is irreversibly dissociated by oxygen (Vogels et al. 1988). Exposure to low levels of O_2 (a few ppm) lowers the adenylate charge of methanogens and cause their death (Robertson and Wolfe 1970). Alternate electron acceptors other than O_2, inhibit methanogensis in mixed microbial ecosystem by channelizing electron flow to microorganisms that are thermodynamically more efficient than methanogens (e.g. denitrifers or sulphate reducers).

Chemical inhibitors have also been used to lower the emission of CH_4 from agricultural fields. 2-bromoethane sulphonic acid, because of working as structural analogue of co-enzyme M of methanogenesis, inhibits methane formation in the soil. Chlorinated CH_4 (e.g. chloroform and methyl chloride) is also found to be competitive inhibitor of CH_4 formation (Cicerone and Oremland 1988).

Methanogenesis may also be inhibited by some metals, like copper and cadmium (Drauschke and Neumann 1992). Inubushi et al. (1990) also studied the effect of cadmium (Cd) on CH_4 emission from the rice soil in both laboratory and pot experiment and reported lower CH_4 emission.

16.7.5 Cultural Practices and Crop Diversification

Mechanical disturbances of flooded soils caused by cultural practices (e.g. land preparation, transplanting, weeding, fertilization and harvest) increase ebullition of soil-entrapped CH_4 (Neue et al. 1994). Less cultural disturbances of reduced soils and shorter flooding period in direct-seeded rice, also lower CH_4 emissions. The adoption of direct seedling (wet and dry seedling), instead of transplanting, is highly dependent on the ability to manage water regimes. Often rice fields cannot be drained in the wet season and in rain-fed areas, by the farmers because of uncertain rainfall.

Crop diversification is a feasible option to reduce total CH_4 emissions in line with economic benefits. In rice growing areas with year around irrigation, production can be shifted by adopting a rice upland cropping system (i.e. sequential cropping of one upland crop before or after one or two crops of rice). Such cultural practices would drastically cut CH_4 emission from paddy fields.

16.7.6 Soil Amendments

Soil amendment is essential for intensive cultivation. Nitrogen is an element that is fundamental to crops and strongly affects growth and yield of crops. However, the application of large amounts of nitrogen can cause over-luxuriant growth, spindly growth, lodging of rice plants, and N_2O emission (Cai et al. 1997; Hua et al. 1997). Nitrogen fertilizers are also known to influence the activities of many groups of microorganisms. The source, mode and rate of application of nitrogen fertilizers influence CH_4 production and emission from flooded rice paddies. Even under the best possible fertilization practices, substantial amounts of the N applied to the field are emitted to the atmosphere. In irrigated rice, gaseous losses of N may account for up to 48% of the N applied (Reddy and Patrick 1980).

Adding any electron acceptor to the soil, can arrest soil reduction. The main electron-acceptors in submerged soils include dissolved oxygen (O_2), NO_3-, Fe (III), SO_4^{2-} and CO_2. The final products of reduction in submerged soils are Fe (II) from Fe (III), H_2S from SO_4^{2-} and CH_4 from CO_2. SO_4^{2-} is a plant nutrient and can supply the much-needed nutrient in S deficient soil. Sulphate aided fertilizers are known to decrease CH_4 emission because of the competition between sulphate-reducing bacteria and methanogens for the substrates, hydrogen and acetate (Hori et al. 1990, 1993). Methane emission has been significantly reduced by 20% by the application of sulphate through ammonium sulphate (Schütz et al. 1989; Lindau et al. 1993), sodium sulphate (28–35%) (Lindau et al. 1993) and gypsum (55–70%) (Denier van der Gon and Neue 1994), while a non-significant reduction with potassium sulphate was reported by Wassmann et al. (1993). In ammonium sulfate, although ammonium can reduce CH_4 oxidation and thereby, increase CH_4 emission (Hűtsch et al. 1993; Willison et al. 1995), but the presence of sulphate overwhelms this effect and reduce overall CH_4 emission. In a Beijing rice field, an organic amendment plus $(NH_4)_2SO_4$ as topdressing, applied in different amounts and at different growth stages, reduced methane emission by about 58% and increased rice yield by top dressing (Kumar et al. 2000). Minamikawa et al. (2005) reported that increases in the application rate of ammonium in the form of ammonium sulphate from 45 to 135 kg N/ha (compared to a convential rate of 90 kg N/ha) decreased CH_4 emission and increased rice yield under field condition (Table 16.7).

Addition of single super phosphate (which also contains sulphur) as phosphorus (P) source to flooded rice soils, not only supplied P to wetland rice, but also inhibited methane production. On the other hand, application of P as K_2HPO_4 was found to stimulate methane production in a P-deficient soil. Also application of rock phosphates, which contained sulphur, retarded methane production, while rock phosphate sources, that did not contain sulphur, stimulated methane production. It was concluded that the application of single super, phosphate to wetland rice could be used as a methane mitigation strategy in P-deficient soils where addition of P is necessary for sustaining increase rice production and productivity (Adhya et al. 1997). Lindau et al. (1994) evaluated the efficacy of application of calcium sulphate for mitigating methane evolution in irrigated rice field. Calcium sulphate was applied at 0, 1000, and 2000 kg ha^{-1} to rice plots treated with urea (128 kg N ha^{-1}). It was

Table 16.7 Total CH_4 emission and rice yield at different application rate of ammonium sulphate (Minamikawa et al. 2005)

Application rate (kg N/ha)	CH_4 emission (g CH_4/m^2)	Grain yield (g DW/m^2)	Straw yield (g DW/m^2)
135	2.67b	682a	775a
90	6.73a	646a	693b
45	7.86a	514b	494c
0	Not determined	389c	394d

The same letters in each column are not significantly different at $p = 0.05$ by Tukey's HSD test found that calcium sulphate application reduced methane evolution by 29 and 46% compared to control (no calcium sulphate applied) at the low (1000 kg ha^{-1}) and high rate (2000 kg ha^{-1}), respectively, over the 70 days sampling.

Application of salts to wetland rice soils can affect methane production and emission. For example, addition of sodium chloride at relatively high concentration (0.18 M) retards methane production (Patel and Roth 1977). Application of seawater also retards methane formation at relatively low rates of salt concentrations, because of sulphate content in the seawater (Koyama et al. 1970). The presence of sulphate provides the opportunity for sulphate reducers to compete with methanogens for hydrogen. Methane emission from wetland rice fields on saline, were lower than CH_4 emission from otherwise comparable non-saline rice fields (Denier van der Gon and Neue 1995).

Urea [$(NH_2)_2CO$] is the most used nitrogen fertilizer in the world, accounting for 47% of the total as of 2002 (FAO 2005). The CH_4 emission from rice fields was decreased by 18% with the shallow incorporation (5 cm deep) of urea (100 or 200 kg N ha^{-1}) compared to that of the control plots, and by about 40% with the deep incorporation (20 cm deep) of urea (200 kg N ha^{-1}) compared to that of shallow incorporation (Schütz et al. 1989). But CH_4 emissions increased by about 19% with the surface application of urea (200 kg N ha^{-1}) compared to those of untreated plots. In a laboratory incubation study, addition of urea (100, 200 and 300 mg N kg^{-1} soil) to flooded soil stimulated CH_4 production (Wang et al. 1992). Kimura et al. (1992) suggested that foliar applications of N fertilizers could suppress CH_4 fluxes from flooded paddy soils. In a recent study, Rath et al. (1999) found that the sub-surface application of urea super granules was marginally effective in reducing the CH_4 flux relative to that in the untreated control. Compared to the effects of ammonium sulphate, the effects of urea on CH_4 emission are very complex, bringing conflicting results among different fields and soils. Many researchers have reported that CH_4 emission increased with increasing application of urea as compared to no application under field conditions (Lindau and Bollich 1993; Matsumoto et al. 2002; Dubey 2003). This result was attributed to an increase in soil pH caused by the hydrolysis of urea (Wang et al. 1992), an increase in rice biomass (Banik et al. 1996) and the inhibition of methanotrophs by ammonium (Conrad and Rothfuss 1991; Kumaraswamy et al. 1997; Dubey 2003).

In principle, three different reasons have been suggested for the inhibitory effect of nitrogenous fertilizers, especially NH_4-N fertilizers on CH_4 oxidation: (1) an intermediate inhibition of the methanotrophic enzyme system (Bedard and Knowles 1989); (2) secondary inhibition through the NO_2^- production from methanotrophic NH_4^+ oxidation (Megraw and Knowles 1989); and (3) dynamic alterations of microbial communities of soil (Powlson et al. 1997).

Addition of other oxidants may also influence CH_4 emission from rice paddy fields as they delay the development of soil reductive conditions. There have been several reports on the effects of Fe-containing materials on decreasing CH_4 emission (Furukawa et al. 2001; Qu et al. 2004). Addition of Fe-containing slag increased the soil Eh and pH (Yoshiba et al. 1996; Nozoe et al. 1999). Kitada et al. (1993) showed that irrigation water containing 40 mg L^{-1} nitrate decreased CH_4 emission by 23%

as compared to normal irrigation water (less than 1 mg L^{-1}). Addition of iron containing materials as bauxite, iron ore and residues of iron manufacture, probably reduces CH_4 emission. This option, as well as mid-season draining practice, have been adopted to prevent *Aki-ochi*, a disorder in Japanese rice attributed to hydrogen sulphide toxicity due to low soil redox potential (Table 16.8). In this case, the effect depends on the contents of free iron oxide in the soil dressing. Yoshiba et al. (1996) also reported impact of Mn_4^{++} and SO_4^{2-} on reducing CH_4 emission. Since excessive use of oxidants would damage the physiology of rice plants, these substances could be used judiciously as a mitigating option.

16.7.7 Organic Matter Application

The application of organic matter conserves soil fertility for sustainable rice production. In Japan, organic matter is applied together with chemical fertilizer. Because of the slow release of organic nitrogen during decomposition, the use of organic matter prevents temporal and spatial variations in the rice yield. Since, the organic matter supplies substrates for the development of soil reductive conditions and CH_4 production, the kind, rate, timing and degree of maturation affect the magnitude of CH_4 emission.

Methane emissions are generally enhanced by organic inputs in to the soil, such as straw or manure amendment. The increment in CH_4 emission following organic inputs depends on quantity, quality and timing of the application (Yagi and Minami 1990; Sass et al. 1991). Rice straw and manure are typically applied before transplanting, resulting in an emission peak during the first half of the growing season. High temperature in the weeks, following the incorporation of these material, results in a pronounced emission peak, whereas low temperature during this period diminishes this peak (Wassmann et al. 2000). Incorporation of organic material also creates a pool of readily available N and therefore, often stimulates N_2O emission (Rolston et al. 1982; Flessa and Beese 1995). However, the observed increments in N_2O emission were not as pronounced as for CH_4 emission.

16.7.7.1 Fermentation of Manure

Several field studies have compared different types of organic amendments with regard to CH_4 emissions. While the differences between fresh materials either straw, animal manure or green manure, have been relatively small, field records showed a big disparity between emissions triggered by fresh pre-fermented material (Yagi and Minami 1990; Wassmann et al. 1993; Corton et al. 2000). During fermentation processes, the incorporation of fermented material into the soil entails a lower emission potential. Applying residue from a biogas generator could reduce emission by approximately 60% as compared to fresh organic amendments and 52%, compared to the locally practiced combination of urea and organic amendments (Wassmann et al. 1993). The combustion of biogas will also save fossil fuel consumption, hence this mitigation option could be considered as a win-win solution.

Table 16.8 Effects of various options against *Aki-ochi* on the yield of rice (Yanagisawa 1978)

Option	No. of tests	Distribution of yield index [a]									Average yield index
		81–90	91–100	101–110	111–120	121–130	131–140	141–150	151–160	>160	
Bauxite	74	1	1	43	11	4	1				108
Iron ore	18	1	5	5	5	1	–	–	–	–	108
Residue of iron manufacture	19	–	–	16	2	1	–	–	–	–	109
Soil dressing [b]	247	4	48	123	48	15	4	3	2	–	103
Sediment dressing [c]	62	–	–	20	28	7	4	2	–	1	116
Deep tillage	40	–	–	22	9	1	–	–	–	–	108

[a] Yield index: Percentage of brown rice yield after the treatment against that before the treatment

[b] In most cases, soils corrected from B horizon in the mountainous area were dressed

[c] Dressed with lake or river sediment

Composting of organic inputs may offer another agricultural practice that could reduce GHG emission (Yagi and Minami 1990; Corton et al. 2000). Moreover, compost application also results in very low N_2O emissions during the rice crop, that are approximately 50% lower than the emissions from rice fields treated urea (Zheng et al. 2000). However, CH_4 emissions during anaerobic composting process could counterbalance gains observed after the incorporation in to the soil. This emission during composting can be reduced to a greater extent through aerobic composting techniques. Organic amendments derived from aerobic composting of rice straw, significantly reduced emission as compared to fresh straw (Corton et al. 2000).

16.7.7.2 Adjusting Straw Incorporation

Crop production inevitably results in large amounts of straw residues that are typically left in the fields. Straw is often burnt to prepare the field for the next cropping cycle; especially the wheat straw is burnt almost all over the Asian rice-wheat belt (Ladha et al. 2000). Removal or burning of residues ensures farmer's quick seedbed preparation and avoids the risk of N immobilization during decomposition of residues with wide C: N ratio (Beri et al. 1995). Incomplete combustion of carbon, which is generic to smoldering fires of harvest residues generates substantial amounts of carbon monoxide (CO) and other pollutants and thus, have adverse effects on local air quality. Moreover, the burning process also releases methane and nitrous oxide into the atmosphere.

Straw incorporation influences methane emission, in two ways depending on the amount of straw added. It either increases methane emission during 2–3 week period following permanent flooding season (Sass et al. 1991) or induces early seasonal increase in CH_4 emission with incorporated straw probably due to direct transformation of straw carbon to methane through microbial activity in the soil. When straw incorporation causes an increase in methane emission over the whole season, rice grain yield decreases proportionately. It may be due to the fact that an addition of straw causes an increase in root biomass, exudation and root decomposition. Addition of rice straw compost increased CH_4 emission by 23–30% as compared to the 162–250% increase in emission with the use of fresh rice straw (Corton et al. 2000). In contrast, composts of cow dung and leaves decreased CH_4 flux (Agnihotri et al. 1999). Inducing aerobic degradation through the addition of organic matter may significantly reduce CH_4 emission (Yagi et al. 1997), but at the some time, this might increase N_2O emission by nitrification of released ammonium.

In addition, an increase in methane emission with additional straw amendments depended on the method of application. Addition of rice straw also stimulates CH_4 emission from flooded rice paddies. Application of rice straw at $5\,t\,ha^{-1}$ and $12\,t\,ha^{-1}$ increased CH_4 efflux rates by factors of 2.0 and 2.4, respectively, relative to those of unamended control plots; but in plots amended with rice straw at $24\,t\,ha^{-1}$ or composted rice straw at $60\,t\,ha^{-1}$, no further increase in CH_4 emissions was noticed (Schütz et al. 1989). Kludze and DeLaune (1995) reported that rice straw applied at the rate of $11\,t\,ha^{-1}$ enhanced CH_4 emissions, whereas $22\,t\,ha^{-1}$ of rice straw retarded CH_4 emissions. In a study, Rath et al. (1998) showed that the addition of rice straw distinctly increased CH_4 production in alluvial soil at all moisture levels [−1.5 MPa and −0.01 MPa (non flooded), 0 MPa (saturated) and at a 1: 1.25 soil: water ratio (flooded)]. These results also raise the importance of many currently unknown sources of CH_4, such as organic-amended non-flooded soils, as contributing to the global CH_4 budget. Bossio et al. (1999) reported that straw incorporation practices altered the organic matter availability, CH_4 pool and flux dynamics of rice soils, and increased emissions per unit available organic matter. However, the use of compost, among the different sources of organic materials, is considered as one of the effective means of mitigating CH_4 emissions from rice fields (Neue 1993; Wassmann et al. 1993; Minami and Neue 1994).

No doubt, many studies have reported that rice straw increases CH_4 emission (Nugroho et al. 1994; Yang and Chang 1997; Chidthaisong et al. 1999), but in Japanese paddy fields, because of aerobic decomposition during the fallow period, the application of straw in the previous autumn or winter decreased CH_4 emission compared to just before transplantation (spring) (Hadi et al. 2001; Matsumoto et al. 2002). However, there was no difference in CH_4 emission between winter application (application two-month prior) and no application (Minamikawa and Sakai 2006). Composting of rice straw is also effective in decreasing CH_4 emission because of aerobic decomposition outside the field (Yagi and Minami 1990; Chidthaisong et al. 1999). Burned straw also decreases CH_4 emissions from fields (Bossio et al. 1999), but CO_2 emissions are increased due to burning of straw.

16.7.8 Cultivars Choice

Since there is a wide variation in CH_4 emission among the rice cultivars, selection of low CH_4 emitting rice cultivar may be one of the strategies to contain CH_4 emission from paddy fields. Usually different cultivars of rice are selected to obtain higher yields with better quality. Characteristics of several rice cultivars, like rice growth and yield, plant conductance of CH_4 transport and concentration of dissolved organic carbon in the soil play several important roles in CH_4 emission. Selecting and breeding the rice cultivars, that emit lower CH_4, are a desirable approach because of easy adoption. Therefore, four different points that should be kept in mind for the selection of rice cultivars that emit low CH_4 are (1) varieties that has low exudation of carbon from roots, (2) that have the efficiency of a low CH_4 transport and a high CH_4 oxidation in the rhizosphere depending on the specific aerenchyma system in roots and shoots, (3) that have a higher harvest index in order to reduce organic matter into the soil after harvest and the last one (4) that are suitable and have a high productivity when the other mitigation options are performed.

Difference in cultivars can lead to an order of magnitude difference in methane emission (Parashar et al. 1996). In a study of five rice cultivars in irrigated fields near Beijing, China, it was observed that methane emission during the tilling-flowering stage varied by a factor of two (Lin 1993). These studies show a significant variation in methane emission, which is solely dependent on cultivars choice. Cultivars choice by individual farmers could thus greatly influence regional and global estimates of methane emission from the rice fields. The wide variation of traits and related emission rates among cultivars open the possibility for the choice of existing cultivars and the breeding of new cultivars as a method for mitigation of methane emission. Some cultivars may have more or less efficient conduits for the removal of methane from the soil through the rice plant, others may deposit more or less organic matter in the soil during the growing season or may be able to transmit more or less oxygen to the rhizosphere, thus raising the redox potential of the soil system or in other ways altering the bacterial response of the rhizosphere. In other cultivars, differential allocation of translocated carbon may even promote higher grain yield in preference to root processes and eventual methane production and emission. Methane emission may also be affected by differences among cultivars in the number of tillers, root biomass, rooting pattern, root respiration and other physiological variables.

16.7.9 Role of Microbes

All above discussed mitigating options are mostly applicable on a field scale and feasible for the farmers. These options basically affect the ecology of microorganism, but insights into microbial ecology have been disregarded in field-scale mitigation.

Variations in CH_4 emission from paddy fields are attributable mostly to variations in methanotrophic activity (Schütz et al. 1989), indicating that methanotrophs are suitable organisms for controlling CH_4 emission (Adachi 2001). Therefore, stimulation of the populations and/or activity of methanotrophs can decrease CH_4 emission. Anaerobic CH_4 oxidation occurs in ecosystems other than paddy soils (Alperin and Reeburg 1985; Iversen 1996). Although there have been no reports of the isolation of bacteria that affect anaerobic CH_4 oxidation (Kumaraswamy et al. 2000), the possibility of anaerobic CH_4 oxidation in paddy fields has been suggested by Miura et al. (1992) and Murase and Kimura (1994).

Amann et al. (1995) reported that probably less than 1% of bacterial species have ever been isolated. The microorganisms, that have been isolated, are not necessarily those that are the only, or the most active, ones in the soil (Le Mer and Roger 2001). As suggested by Kumaraswamy et al. (2000) and Schimel (2000), further microbial studies will enable us to predict ecosystem behaviors and the ecological significance of diversity and community structure.

The production of CH_4 is controlled by flow of carbon (C) and electrons to the microbial community of methanogens during anaerobic organic matter degradation process. Also, thermodynamic constraints of in situ reactions involved and changes in the composition of microbial community affect methane production (Conrad 2002). An understanding of the in situ processes involved on a microscopic level provides leads for developing strategies for controlling methane production and emission. Conrad (2002) has reviewed the existing knowledge of microbiological data, microscopic processes, and other factors relevant for the control of methane production in wetland rice fields.

In principle, mitigation in methane production can be achieved by adding electron acceptors and oxidants. However, electron-acceptors, such as ferric iron or sulphate, are preferred for their role because they allow iron reducers to successfully compete for substrates, hydrogen and acetatae, with methanogens. This stops methane production and emission.

16.8 Conclusion

Paddy fields are one of the major anthropogenic sources of CH_4 emission. For a long time, India – a large producer of staple rice food, was surmised to be major contributor to global methane budget. However, this myth was cleared when the Methane Campaign 1991 and subsequently Methane Asia Campaign 1998 categorically proved that India contributes only one tenth (3.61–4.0 Tg/Yr) of the EPA projections (37.8 Tg/Yr). But still, there is scope to reduce CH_4 emission from paddy fields through various mitigation strategies, like water management, use of nitrification and methane inhibitors, pesticides, changes in cultural practices and crop diversification, soil amendments, selection of cultivars, and use of fermented manures. If such efforts are made globally, it will cut drastically CH_4 emission from paddy fields and certainly help to postpone the danger of climate change.

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