CHAPTER THREE

# Sugarcane for Bioethanol: Soil and Environmental Issues

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## Contents



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## Abstract

Cultivation of sugarcane for bioethanol is increasing and the area under sugarcane is expanding. Much of the sugar for bioethanol comes from large plantations where it is grown with relatively high inputs. Sugarcane puts a high demands on the soil because of the use of heavy machinery and because large amounts of nutrients are removed with the harvest; biocides and inorganic fertilizers introduce risks of groundwater contamination, eutrophication of surface waters, soil pollution, and acidification. This chapter reviews the effect of commercial sugarcane production on soil chemical, physical, and biological properties using data from the main producing areas. Although variation is considerable, soil organic C decreased in most soils under sugarcane and, also, soil acidification is common as a result of the use of N fertilizers. Increased bulk densities, lower water infiltration rates, and lower aggregate stability occur in mechanized systems. There is some evidence for high leaching losses of fertilizer nutrients as well as herbicides and pesticides; eutrophication of surface waters occurs in high-input systems. Soil erosion is a problem on newly planted land in many parts of the world. Trash or green harvesting overcomes many of the problems. It is concluded that sugarcane cultivation can substantially contribute to the supply of renewable energy, but that improved crop husbandry and precision farming principles are needed to sustain and improve the resource base on which production depends.

## 1. Introduction

Bioenergy is energy from biofuels. Biofuel is produced directly or indirectly from biomass such as wood, charcoal, bioethanol, biodiesel, biogas (methane), or biohydrogen (FAO, 2006). It is big business. Demand for biofuels is surging because of the rise in crude oil prices and the global search for renewable energy (Valdes, 2007) and global biofuel production tripled between 2000 and 2007. Currently, the most important biofuel crops are corn, rapeseed, soybean, sugarcane, and oil palm whereas suitable trees for bioenergy production include eucalyptus, poplar, and willow. Biofuel production itself needs fossil energy. Currently, agriculture accounts for about 15% of the global energy demands (fertilizers, transport etc.) but it is estimated that agriculture can produce half to several times the current global energy demand (Smeets *et al.*, 2007).

The environmental impact of the shift toward growing crops for energy is still to be assessed. It is a complex matter with economic interests and other factors interacting on several scales. For example, the cultivation of biofuel crops is competing with food crops and may drive up commodity prices (UNEP, 2007)—over the last few years, world food prices have increased because of market demand for corn, wheat, and soybean. There is further concern that the expansion of biofuel crops takes place at the expense of rainforest and has negative effects on biodiversity and the environment.

Sugarcane as a biofuel crop has much expanded in the last decade, yielding anhydrous ethanol (gasoline additive) and hydrated ethanol by fermentation and distillation of sugarcane juice and molasses (Gunkel *et al.*, 2007; Pessoa *et al.*, 2005). By-products are bagasse and vinasse (stillage or dunder), which is the liquid waste sometimes used for fertigation purposes. Bagasse, a by-product of both sugar and ethanol production, can be burned to generate electricity or be used for the production of biodegradable plastic. It provides most of the fuel for steam and electricity for sugar mills in Australia and Brazil.

One hectare of sugarcane land with a yield of 82 t $ha^{-1}$ produces about 7000 liter of ethanol. Brazil currently produces about 31% of the global production and it is the largest producer, consumer, and exporter of ethanol for fuel (Andrietta *et al.*, 2007). The industry employs more than one million people (Pessoa *et al.*, 2005). The value of the sugar and ethanol industry reached $8 billion in 2006, some 17% of Brazil's agricultural output (Valdes, 2007).

Between 1990 and 2005, global average sugarcane yields increased from 61 to 65 Mg $ha^{-1}$ (http://faostat.fao.org). In 1990, global production was 1050 million Mg and in 2005, production of sugarcane was 1225 million Mg. Much is grown on large plantations but in some countries sugarcane is grown by smallholders, for example in Thailand where there are more than 100,000 farmers growing sugarcane (Sthiannopkao *et al.*, 2006). In Brazil, less than 20% of the sugarcane is produced on small farms; most is grown in the southeast with over 60% of the production in the São Pula district (FAO, 2004). In some countries, sugarcane is the main source of revenue and in Mauritius, sugarcane occupies 90% of the arable land (Ng Kee Kwong *et al.*, 1999). Globally, the area harvested increased by 2.6 million ha in the period 1990–2005; the largest expansion was in India and Brazil. It is expected that the area under sugarcane in Brazil will expand by 3 million ha over the next 5 years whereas the area under sugarcane in China is forecast to rise by 5% or more than 100,000 ha $year^{-1}$. Brazil has a long tradition of growing sugarcane. In sixteenth century, it was the world's major supplier of sugar (Courtenay, 1980). In 1975, the area under sugarcane in Brazil was 1.9 million ha (de Resende *et al.*, 2006), now there is about 6.2 million ha under sugarcane in Brazil compared to 21 million ha soybean and 14 million ha corn. Other big sugarcane producers are India (4.2 million ha), China (1.4 million ha), Thailand (1.1 million ha), and Pakistan (0.9 million ha) whereas the sugarcane areas in Australia, Cuba, Indonesia, Mexico, and South Africa cover some 0.5–0.6 million ha in each country. In the United States, there are about 170,000 ha in Louisiana and 167,000 ha in Florida. Against the trend, the area under sugarcane in Hawaii has decreased from about 100,000 ha in the 1930s to 6000 ha in 2007, and also the area under sugarcane in Cuba has been more than halved in the last 15 years.

Traditionally, sugarcane was harvested manually; the senescent leaves (trash) and stalks were removed by people using big knives. Green harvesting was common in Brazil up to 1940s (de Resende *et al.*, 2006), but the large volume of trash makes manual harvesting difficult (Boddey *et al.*, 2003). As labor shortages developed, it became common practice to burn of the dead leaves prior to the harvest (preharvest burning). In the last two decades, preharvest burning has been replaced by mechanical green- or trash harvesting by cutter-chopper-loader harvesters that leave the trash on the field. Most of the sugarcane in Australia and parts of the West Indies is now arvested like this (Graham *et al.*, 2002a). Up to the 1960s, Australian sugarcane was harvested manually but a decade later, following severe labor shortages, nearly all sugarcane was harvested mechanically (Brennan *et al.*, 1997). Currently, about 30% of the Brazilian sugarcane is green-harvested, the rest is harvested manually with preharvest burning. All sugarcane in the United States is mechanically harvested but over 90% of the fields are burned after the green harvesting, to get rid of the trash blanket.

Sugarcane is grown as a ratoon crop: the whole above ground biomass is harvested each year and harvests may continue for a number of years (ratoons). Yields decline with ratooning and, after some years, the land is ploughed and new sugarcane is planted. Much of the world sugarcane is grown with a high degree of mechanization. Also, large amounts of biomass are annually removed with the harvest and herbicides and pesticides are used extensively. Irrigation and large amounts of inorganic fertilizers are often required for high yields. As a consequence, soil properties are likely to change under sugarcane cultivation and the high biocide inputs may affect the environment. Environmental concerns and policies are key factors affecting the future of sugarcane production (Valdes, 2007). There is a also risk that the sugar industry is expanding on marginal lands where the costs or preventing or repairing environmental damage may be high (Arthington *et al.*, 1997). This chapter reviews the main soil and environmental issues under continuous sugarcane cultivation. Most of this work predates the surge of sugarcane production for bioethanol but the results are very relevant for the new situation.

## 2. Changes in Soil Chemical Properties

### 2.1. Data sources and types

There is fair a body of literature on changes in soil properties under sugarcane cultivation, especially in conference proceedings and books. Increasingly, there have been publications on soil and environmental issues in international scientific journals in English. Changes in soil properties under continuous sugarcane have been investigated in two ways. First, soil properties are monitored over time at the same site and this generates Type I data using chronosequential sampling. There are few such data sets because they require long-term research commitment and detailed recordings of soil management and crop husbandry practices. In the second approach, soils under adjacent different land-use systems are sampled at the same time and it is called biosequential sampling (Tan, 1996) generating Type II data (Sanchez *et al.*, 1985). The assumption is that the soils of the cultivated and uncultivated land are the same and that differences in soil properties can be attributed to differences in land use and management (Hartemink, 2003).

A considerable number of studies have focused on soil chemical and physical changes, and there are only few studies that included soil biological changes (Table 1). Several studies have been conducted in Brazil, Australia, and South Africa; although sugarcane is important and extensively grown in many other countries, fewer studies have been reported in the literature. Well-researched soil types are Fluvents, Inceptisols, Alfisols, and Oxisols; less data are available from Vertisols, although they are extensively used for sugarcane (Ahmad, 1983).

## 2.2. Monitoring over time

Few studies have monitored soil chemical properties under continuous sugarcane cultivation. In Fiji, Haplic Acrustox were sampled under native vegetation prior to planting sugarcane, and again 6 years later (Masilaca *et al.*, 1985). Exchangeable K decreased, soil P levels were increased in two of the three topsoils, and in one-third of the Oxisols, the topsoil pH had declined from 5.5 to 4.6 (Table 2).

Schroeder *et al.* (1994) measured soil pH over 5 years on sugarcane farms on soils derived from sedimentary rocks in South Africa. These soils had received about 140 kg N ha$^{-1}$ year$^{-1}$ and pH declined by 0.4 units. Soil pH in the VMC milling district in the Philippines declined from 5.0 to 4.7 over a 19-year period under sugarcane (Alaban *et al.*, 1990). The decline in pH was accompanied by a decrease in organic C from 14 to 10 g kg$^{-1}$; also available P and levels of exchangeable cations decreased (Table 3). In Papua New Guinea, Hartemink (1998a,c) compiled soil data at a plantation on Fluvents and Vertisols. Soil chemical data were available from the early 1980s and early 1990s (Table 4). A significant decrease was found in the pH, available P, and CEC of the Fluvents and even in Vertisols, the pH had decreased significantly. A decrease of 0.2–0.4 pH unit was found to a depth of 0.60 m after 10 years of continuous sugarcane (Table 5).

**Table 1** Studies focusing on changes in soil chemical, physical, and biological properties under sugarcane cultivation

| | | Soil property investigated | | | Data[a] | | |
|---|---|---|---|---|---|---|---|
| Soil order | Country | Chemical | Physical | Biological | Type I | Type II | References |
| Alfisols | Australia | √ | √ | √ | | √ | Blair, 2000; Bramley *et al.*, 1996; Pankhurst *et al.*, 2005a,b; Skjemstad *et al.*, 1999 |
| | Brazil | √ | √ | | | √ | Caron *et al.*, 1996; Tominaga *et al.*, 2002 |
| | India | √ | | | | √ | Sundara and Subramanian, 1990 |
| | Swaziland | √ | √ | √ | | √ | Henry and Ellis, 1995; Nixon and Simmonds, 2004 |
| Andosols | USA Hawaii | √ | | √ | | √ | Zou and Bashkin, 1998 |
| Fluvents | Australia | √ | √ | √ | | √ | Bramley *et al.*, 1996; Braunack *et al.*, 1993; Pankhurst *et al.*, 2005a,b; Skjemstad *et al.*, 1999 |
| | Brazil | √ | | | | √ | de Resende *et al.*, 2006 |
| | Fiji | | √ | | | √ | Masilaca *et al.*, 1985 |
| | USA Hawaii | | √ | | | √ | Juang and Uehara, 1971; Trouse and Humbert, 1961 |
| | Iran | | √ | | | √ | Barzegar *et al.*, 2000 |
| | Mexico | √ | | | | √ | de la F *et al.*, 2006 |
| | Papua New Guinea | √ | √ | | √ | √ | Hartemink, 1998a,c |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Inceptisols | Australia | √ | | √ | | √ | Bramley *et al.*, 1996; Noble *et al.*, 2003; Pankhurst *et al.*, 2005a,b; Skjemstad *et al.*, 1999 |
| | India | √ | √ | √ | | √ | Singh *et al.*, 2007; Srivastava, 2003; Suman *et al.*, 2006 |
| | Iran | | √ | | | √ | Barzegar *et al.*, 2000 |
| | South Africa | √ | √ | | | √ | Dominy *et al.*, 2002 |
| Oxisols | Brazil | √ | √ | √ | | √ | Caron *et al.*, 1996; Ceddia *et al.*, 1999; Cerri and Andreux, 1990; de Souza *et al.*, 2005; Nunes *et al.*, 2006; Razafimbelo *et al.*, 2006; Silva *et al.*, 2007 |
| | Fiji | √ | √ | | √ | | Masilaca *et al.*, 1985 |
| | USA Hawaii | | √ | | | √ | Juang and Uehara, 1971; Trouse and Humbert, 1961 |
| | South Africa | √ | √ | √ | | √ | Dominy and Haynes, 2002; Dominy *et al.*, 2002; Haynes *et al.*, 2003 |
| | Swaziland | √ | √ | √ | | √ | Henry and Ellis, 1995 |
| Spodosols | Australia | | √ | √ | | √ | McGarry *et al.*, 1996a,b |
| | USA | √ | | | | √ | Muchovej *et al.*, 2000 |
| Ultisols | Australia | | | √ | | √ | Pankhurst *et al.*, 2005a,b |
| | Brazil | | √ | | | √ | Ceddia *et al.*, 1999 |
| | Indonesia | √ | | | √ | | Sitompul *et al.*, 2000 |
| Vertisols | Mexico | √ | √ | | | √ | Carrillo *et al.*, 2003; de la F *et al.*, 2006 |
| | | √ | √ | | √ | √ | Hartemink, 1998b,c |

(*continued*)

**Table 1** (*continued*)

| | | Soil property investigated | | | Data[a] | | |
|---|---|---|---|---|---|---|---|
| Soil order | Country | Chemical | Physical | Biological | Type I | Type II | References |
| | Papua New Guinea | | | | | | |
| | South Africa | √ | √ | √ | | √ | Graham and Haynes, 2005, 2006; Graham *et al.*, 2002b |
| | Zimbabwe | √ | | √ | | √ | Rietz and Haynes, 2003 |
| Not specified | Australia | √ | √ | √ | | √ | Garside *et al.*, 1997; King *et al.*, 1953; Maclean, 1975; Magarey *et al.*, 1997; Moody and Aitken, 1995, 1997; Wood, 1985 |
| | India | √ | √ | | | √ | Srivastava, 1984; Yadav and Singh, 1986 |
| | Mexico | √ | | | | √ | Campos *et al.*, 2007 |
| | Philippines | √ | | | √ | | Alaban *et al.*, 1990 |
| | South Africa | √ | √ | | | √ | Schroeder *et al.*, 1994; Swinford and Boevey, 1984 |
| | Trinidad | | √ | | | √ | Georges *et al.*, 1985 |

[a] Type I are data whereby soil dynamics are followed with time on the same site; Type II are data whereby different land use was sampled simultaneously [see Hartemink (2006)].

**Table 2** Changes in soil chemical properties at sugarcane plantations in Fiji

| Site | Sampling depth (m) | pH | N (g kg$^{-1}$) | C (g kg$^{-1}$) | P (mg kg$^{-1}$) | CEC and exchangeable cations (mmol$_c$ kg$^{-1}$) CEC | Ca | Mg | K |
|---|---|---|---|---|---|---|---|---|---|
| A | 0–12 | −0.7 | −26.9 | −2.1 | +62.0 | −96.0 | −19.9 | −1.3 | −1.8 |
| | 30–40 | −0.8 | +3.8 | −0.2 | +3.0 | +0.3 | +2.4 | −0.1 | −0.1 |
| | 70–80 | −0.6 | −0.6 | 0 | −1.0 | −18.0 | +0.2 | −0.2 | −0.2 |
| B | 0–12 | −0.3 | −14.2 | −2.2 | −2.0 | −38.0 | −26.9 | −10.6 | −1.1 |
| | 30–40 | +0.1 | +1.3 | +0.1 | +1.0 | +5.0 | −3.3 | −1.4 | −0.2 |
| | 70–80 | −0.1 | +0.5 | −0.2 | +2.0 | +7.0 | −2.9 | −2.2 | 0 |
| C | 0–12 | +0.2 | −17.0 | −0.3 | +64.0 | −3.0 | −29.6 | +1.8 | +0.5 |
| | 30–40 | +0.1 | +7.8 | +0.3 | +6.0 | +35.0 | −1.4 | −0.4 | −0.9 |
| | 70–80 | +0.1 | −0.2 | 0 | −4.0 | +28.0 | 0 | −0.3 | −0.2 |

Soils were Oxisols and had been under sugarcane for 6 years. Type I data, modified from Masilaca *et al.* (1985).

**Table 3** Changes in soil chemical properties on sugarcane plantations in the Philippines

| Sampling period | pH | Organic C (g kg$^{-1}$) | Available P (mg kg$^{-1}$) | Exchangeable cations (mmol$_c$ kg$^{-1}$) Ca | Mg | K |
|---|---|---|---|---|---|---|
| 1969–1970 | 5.0 | 13.3 | 27.3 | 85.7 | 11.6 | 3.7 |
| 1988–1989 | 4.7 | 9.9 | 17.3 | 47.4 | 11.1 | 3.4 |

Type I data, modified from Alaban *et al.* (1990).

## 2.3. Samples from different land-use systems

One of the longest data sets on soil changes under sugarcane cultivation is from the coastal tableland in Alagoas, Brazil (Silva *et al.*, 2007). Soil samples were taken Oxisols in undisturbed forest and compared with soils that had been under sugarcane for 2, 18, and 25 years. Under forest, soil organic C was about 26 g kg$^{-1}$ in the upper 0.20 m soil layer but had decreased to 19 g C kg$^{-1}$ after 2 years of sugarcane cultivation. After 18 and 25 years, soil organic C levels were similar to those under forest in both topsoil and subsoil.

In South Africa, an experiment established in 1939 on a Vertisol at the Experimental Station at Mount Edgecombe, has trash-burned and unburned treatments and with or without inorganic fertilizers. Fertilized plots received 140 kg N ha$^{-1}$, 20 kg P ha$^{-1}$, and 140 kg K ha$^{-1}$. Soil organic matter was lowest when crop residues (trash) were removed and

**Table 4** Soil chemical properties (0–0.15 m) of Fluvents and Vertisols under sugarcane in the 1980s and 1990s

| | Fluvents ($n = 7$ pairs) | | | Vertisols ($n = 5$ pairs) | | |
|---|---|---|---|---|---|---|
| Soil chemical properties | 1982–1983 | 1991–1994 | Difference | 1982–1984 | 1991–1994 | Difference |
| pH $H_2O$ (1:2.5 w/v) | 6.3 | 5.9 | $p < 0.001$ | 6.4 | 6.0 | $p < 0.001$ |
| Available P (mg $kg^{-1}$) | 37.2 | 29.0 | $p = 0.04$ | 35.4 | 24.6 | ns |
| CEC ($mmol_c$ $kg^{-1}$) | 412 | 354 | $p < 0.001$ | 450 | 403 | ns |
| Exchangeable Ca ($mmol_c$ $kg^{-1}$) | 229 | 213 | ns | 269 | 250 | ns |
| Exchangeable Mg ($mmol_c$ $kg^{-1}$) | 100 | 94 | ns | 109 | 95 | ns |
| Exchangeable K ($mmol_c$ $kg^{-1}$) | 11.0 | 9.5 | ns | 13.0 | 10.1 | ns |

ns = not significant. Type I data, modified from Hartemink (1998c).

highest when residues were retained and inorganic fertilizers were applied. Soil pH decreased from 5.8 under natural grassland to 5.2 under sugarcane with fertilizer applications and also as a result of the trash retention. In soils where there was no trash or inorganic fertilizers, there was no significant decline in pH. Acidification was accompanied by a decrease in the levels of Ca and Mg (Graham and Haynes, 2005; Graham *et al.*, 2002a).

Several studies have been conducted in Australia where Type II data are termed samples from "paired sites" or "paired sampling," sampling "old and new soils," comparing "cropped and undeveloped" land, or comparing "virgin and cultivated" soils (Hartemink, 2006). King *et al.* (1953) compared soil chemical properties of uncultivated soils with those that had been under sugarcane for 22 years in the Bundaberg area. The cultivated soils contained on average 22 g C $kg^{-1}$ whereas the C content of virgin soils was 48 g $kg^{-1}$. In proportion, total N contents of the soils under sugarcane were also less than half of the N contents in virgin soils. Maclean (1975) found significant differences in topsoil pH between sugarcane and uncultivated land and also topsoil P, Ca, and Mg levels were significantly lower in soils under sugarcane. In the subsoil, available P and exchangeable Mg were significantly lower, but below 0.3 m depth, there was no significant difference between soils under sugarcane and uncultivated soils. Wood (1985) sampled cultivated and adjacent uncultivated land at 19 sites in a range of different soil types. The cultivated sites had been cropped with sugarcane for at least 30 years whereas the uncultivated sites were road reserves, cleared

**Table 5** Change in pH $H_2O$ with depth based on samples from the same site at different times and from the different land use sampled at the same time

| Type I data | | | | | Type II data | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Sampling depth (m) | Sample pairs | 1986 | 1996 | Difference | Sampling depth (m) | Sample pairs | Natural grassland | Continuous sugarcane[a] | Difference |
| 0–0.15 | 9 | 6.2 | 5.8 | $p < 0.001$ | 0–0.15 | 5 | 6.3 | 5.8 | $p = 0.02$ |
| 0.15–0.30 | 9 | 6.2 | 5.9 | $p < 0.001$ | 0.15–0.30 | 5 | 6.3 | 6.1 | $p = 0.02$ |
| 0.30–0.45 | 7 | 6.5 | 6.1 | $p = 0.02$ | 0.30–0.50 | 5 | 6.6 | 6.4 | $p = 0.05$ |
| 0.45–0.60 | 7 | 6.6 | 6.4 | $p = 0.01$ | 0.50–0.70 | 5 | 6.7 | 6.6 | ns |
| | | | | | 0.70–0.90 | 5 | 6.9 | 6.8 | ns |

[a] Soils were continuously cultivated with sugarcane for at least 10 years
ns = not significant. Modified from Hartemink (1998a).

**Table 6** Changes in soil chemical properties on sugarcane plantations in North Queensland, Australia

| Land use | Sampling depth (m) | pH | C ($g\,kg^{-1}$) | P ($mg\,kg^{-1}$) | CEC and exchangeable cations ($mmol_c\,kg^{-1}$) CEC | Ca | Mg | K |
|---|---|---|---|---|---|---|---|---|
| Sugarcane | 0–0.10 | 5.0 | 7.0 | 35 | 37.0 | 15.2 | 7.3 | 2.0 |
| | 0.10–0.20 | 4.9 | 6.5 | 26 | 37.0 | 15.5 | 5.1 | 1.4 |
| | 0.20–0.30 | 4.9 | 5.6 | 15 | 39.0 | 17.1 | 5.6 | 1.1 |
| | 0.30–0.40 | 5.0 | 4.0 | 9 | 41.3 | 18.7 | 8.1 | 1.0 |
| Uncultivated | 0–0.10 | 5.2 | 15.0 | 14 | 56.3 | 32.8 | 14.1 | 2.9 |
| | 0.10–0.20 | 5.2 | 8.1 | 8 | 47.5 | 26.1 | 12.3 | 1.6 |
| | 0.20–0.30 | 5.1 | 5.9 | 7 | 46.8 | 23.1 | 12.4 | 1.3 |
| | 0.30–0.40 | 5.1 | 4.9 | 3 | 51.7 | 25.0 | 15.3 | 1.3 |

Average data of various soil types. Sugarcane was cultivated for at least 30 years. Type II data, modified from Wood (1985).

land, or forest. A slightly lower pH was found under sugarcane and differences in soil reaction in the 0.20–0.30 m soil horizon were significant (Table 6). Organic C levels in soils under sugarcane were less than half of the levels in uncultivated soils. Exchangeable cations and the CEC were significantly lower in soils under sugarcane but these soils had significantly higher levels of available P due to high application rates of P fertilizers.

Bramley *et al.* (1996) sampled Dystropepts, Ustropepts, Tropaquepts, Natrustalfs, Haplustalfs, and Fluvents that had been under sugarcane for 20 years or more. Soil fertility decline differed between soil orders and depths. Organic C declined in the Fluvents, but no significant changes were found in the other soils. A significant decline in soil pH was found only in Ustropepts. Skjemstad *et al.* (1999) investigated the same soils and found little changes in total soil organic C and in the light fraction ($<1.6$ Mg $m^{-3}$). Well-established sugarcane sites (20–70 years) had lower soil organic C levels in the subsoils relative to uncultivated soils. No difference was found between Ustropepts, Natrustalfs, and Fluvents, and it appeared that sugarcane production did not lead to an overall decline in total organic C in the soil profile confirming the observations of Bramley *et al.* (1996). However, Noble *et al.* (2003) found that soil organic C declined under continuous sugarcane cultivation and levels were 13 g C $kg^{-1}$ in 1994 and 8 g C $kg^{-1}$ in 2000. The pH under continuous sugarcane was 6.6 in 1994 and 6.0 in 2000.

Caron *et al.* (1996) sampled a Typic Haplorthox and Typic Paleudalf under primary forest and 20-year-old sugarcane near São Paulo, Brazil. Topsoil organic C levels were 34 g $kg^{-1}$ in the Alfisol under forest and 16 g C $kg^{-1}$ soil under sugarcane. In Oxisols under forest, there was 45 g C $kg^{-1}$ compared with 30 g C $kg^{-1}$ under sugarcane; the difference between forest and sugarcane extended to 1.2 m in the Oxisol and up to 0.9 m in the Alfisol. The decrease in soil organic C was accompanied by a significant decrease in soil pH in both soil orders but the drop in pH was larger in Alfisols (Caron *et al.*, 1996).

In Mexico, Vertisols and Fluvents under different periods of sugarcane were sampled (de la F *et al.*, 2006): a significant decline in N, P, and organic matter levels was found after 30 years of sugarcane cultivation but pH changes were less consistent. Henry and Ellis (1995) investigated changes in Oxisols and Natraqualfs under sugarcane in Swaziland. The Oxisol had been under sugarcane for 18 years and the Alfisols had been under paddy rice for 25 years and were 15 years under sugarcane when sampled. In Oxisols, the difference in organic C between sugarcane and uncultivated soils was only 2 g C $kg^{-1}$. Exchangeable K in soils under sugarcane was about half the values found in uncultivated soils in both Oxisols and Alfisols. Levels of available P were much higher in the soils under sugarcane. Changes in soil chemical properties were accompanied by a degradation of soil physical and biological properties.

Both the Type I and Type II studies showed considerable changes in soil fertility under continuous sugarcane. In most soils, the pH dropped, often accompanied by a decrease in exchangeable cations. Soil acidification has been reported from sugarcane areas in Australia (Moody and Aitken, 1995), Brazil (Silva *et al.*, 2007), Hawaii (Humbert, 1959), Papua New Guinea (Hartemink, 1998a), Puerto Rico (Abruña-Rodriguez and Vicente-Chandler, 1967), and Florida (Coale, 1993). An important cause of soil acidification is the application of N fertilizers. Because these contain N in the ammonium form, nitrification results in acidification. The soils under sugarcane in Fiji (Table 2) had acidified following the applications of sulfate of ammonia at rates averaging 150 kg N $ha^{-1}$ $year^{-1}$. In Papua New Guinea (Tables 3 and 4), most of the N fertilizers in the mid-1990s were applied as sulfate of ammonia; previously urea was applied that is less acidifying but most of the N is lost when urea is applied on the trash blanket. The levels of P increased in many soils, also as a result of fertilizer applications and relatively low removal rates (see also Section 5.2). A decline in organic matter has been reported from several sugarcane areas; the dynamics of soil organic matter are discussed below. No study has been found that looked at changes in soil micronutrients under sugarcane.

## 2.4. Soil organic matter dynamics

Soil organic matter is key for the productive capacity of many tropical soils (Woomer *et al.*, 1994). As shown in the previous sections, soil organic matter has declined in many soils under sugarcane but some studies found little change in soil organic matter levels under continuous sugarcane. Because there are different systems of cultivation (trash harvesting, preharvest burning) and sugarcane is grown in different agroecologies that largely affect the soil organic matter status, it is hard to generalize.

In Brazil, Cerri and Andreux (1990) measured different C fractions of a Typic Haplorthox under forest and at a sugarcane plantation in São Paula State. The natural abundance of the isotope $^{13}C$ was used to identify organic C sources and to determine the changes in soil organic matter when forest is cleared and sugarcane planted. The approach depends on the difference in the natural $^{13}C$ abundance between plants having different photosynthetic pathways: mainly C3 (forest) and C4 (sugarcane). The $^{13}C/^{12}C$ ratio of C3 plants is lower than that of C4 plants. Table 7 presents the C content in soils under forest and sugarcane. Total C levels after 50 years of sugarcane cultivation were 46% of the levels under forest. After 12 years of sugarcane cultivation, more than 80% of the soil organic C still originated from the forest but after 50 years, the forest C formed 55% of the total C contents in the topsoil. The rate of increase in C originating from sugarcane was slower than the decrease in C that had originated from the forest.

The data in Table 7 were used in a regression model for soil organic matter dynamics (van Noordwijk *et al.*, 1997). The decline in forest-derived organic matter continued during the 50 years spanned by the investigation; the apparent equilibrium value of total soil organic C is based on a balance between gradual build-up of sugarcane-derived organic matter, and decay of forest-based organic matter. For comparison, soil from pastures showed a larger stable C pool, a more rapid decline of labile forest C but also a much faster accumulation of labile crop C, which returned the total soil organic C levels to that of the forest before deforestation after about 7 years (van Noordwijk *et al.*, 1997). Some of the differences between the pasture and sugarcane patterns can be explained by the lower annual input of C under sugarcane (<1.0 Mg C $ha^{-1}$) compared with the pasture (7.5 Mg C $ha^{-1}$) and differences in soil mineralogy and climate (Cerri and Andreux, 1990). Soil texture plays a role; 12 years after conversion from forest to sugarcane,

**Table 7** Carbon content of soils under forest and after 12 and 50 years of sugarcane cultivation (Mg $ha^{-1}$, 0–0.20 m depth)

| | | Sugarcane | |
|---|---|---|---|
| | Forest | Soils under 12 years of sugarcane | Soils under 50 years of sugarcane |
| Total C | 71.9 | 44.6 | 38.5 |
| Stable C originating from the forest | 71.9 | 36.0 | 21.0 |
| C originating from the sugarcane | | 8.6 | 17.3 |

Type II data, modified from Cerri and Andreux (1990).

the majority of the C derived from sugarcane is found in the coarse sand fraction. About 90% of the C in the clay fraction still has the forest signature after 12 years, whereas after 50 years, 70% of the forest-derived C persisted in the clay fraction (Vitorello *et al.*, 1989). These data illustrate the importance of clay–organic matter linkages as a C-protection mechanism (Dominy *et al.*, 2002; van Noordwijk *et al.*, 1997).

Another study in Brazil found that soil organic C levels under continuous sugarcane reached the same levels as soils under forest. Soil organic C under forest was about 26 g $kg^{-1}$ in the soils under forest but had decreased to 19 g C $kg^{-1}$ in soils that were cultivated with sugarcane for 2 years. After 18 and 25 years of sugarcane cultivation, levels were similar to those under forest in both topsoil and subsoil. The increase in soil organic C under continuous sugarcane was explained by the input of filter cake and vinasse (Silva *et al.*, 2007). Also Graham *et al.* (2002b) found similar soil organic C levels in natural grassland compared with soils that had been under sugarcane cultivation for 59 years. Soil organic C levels under sugarcane were even higher when the sugarcane was fertilized.

Not only is organic matter decline affected by clay content and soil texture, it is also different for different fractions. On a Grossarenic Kandiudult in Sumatra, Indonesia, Sitompul *et al.* (2000) modeled soil organic matter dynamics under sugarcane using CENTURY. Rates of change differed between particle size fractions. The sum of light, intermediate, and heavy fractions of macro-organic matter (150 $\mu$m–2 mm) showed a decline of about 250 to about 100 g C $m^{-2}$ after 10 years of sugarcane cultivation.

In South Africa, Graham and Haynes (2006) investigated soil organic matter and the microbial community under burned and trash-harvested sugarcane on Vertisols. Soil organic C was lower under burned sugarcane but $K_2SO_4$-extractable C, light fraction C, microbial biomass C, and basal respiration were much lower; changes occurred to a depth up to 0.30 m. Much organic matter is returned to the soil with trash harvesting but in burned sugarcane systems, the main organic return is through root turnover (rhizodeposition). The authors concluded that the effects of agricultural practice on organic matter status are more obvious and first noted when labile C fractions microbial activity is measured. In these Vertisols, soil organic C levels were similar under natural grassland and sugarcane (Graham *et al.*, 2002b).

In Inceptisols and Oxisols in the South African province of KwaZulu-Natal, the organic C content was 40–50 g C $kg^{-1}$ under natural vegetation but it declined exponentially with increasing years under sugarcane (Dominy *et al.*, 2002). After 20–30 years of sugarcane, organic C content had declined to about 33 g $kg^{-1}$ in the Oxisol and to 17 g $kg^{-1}$ in the Inceptisol. In the Inceptisol, it reached a new equilibrium level after about 30–40 years. The higher organic matter content in the Oxsiol was attributed to clay protection of organic matter. The natural $^{13}C$ abundance in Inceptisols was used to calculate the loss of forest-derived, native soil C and the input of sugarcane-derived C. Sugarcane-derived organic C increased over time until it accounted for about 61% of organic C in the surface 10 cm in soils that had been under sugarcane for more than 50 years (Dominy *et al.*, 2002).

Alfisols under sugarcane in Australia contained about 11 g C $kg^{-1}$ whereas under natural grassland, C levels were 34 g $kg^{-1}$. Levels of soil organic C were much higher under trash-harvesting system than when preharvest burning was practiced, but the organic C levels of the soils under grass were not reached (Blair, 2000).

In most studies on soil organic matter dynamics under sugarcane, it was found that the rates of soil organic matter decline differed for different soils (clay protection), soil organic matter fractions, agroecologies (climate), and management (e.g., trash-harvesting, vinasse applications). In most soils, levels decreased in the first years of cultivation and then slowly increased again. The increase is higher with higher levels of organic inputs (trash, vinasse). Rarely, the original soil organic matter levels are reached, typically, the levels settle at 60% of the soil organic matter levels in soils under natural vegetation.

## 2.5. Leaching, denitrification, and inorganic fertilizers

Many studies have investigated the effects of inorganic fertilizer on sugarcane yield, sugar and leaf nutrient content, and the overall response to inorganic fertilizers. The Diagnosis and Recommendation Integrated System, originally developed for rubber, has been adapted to sugarcane in Brazil, United States, and South Africa (El Wali and Gascho, 1984; Reis and Monnerat, 2002; Sumner and Beaufils, 1975). The effects of lime have been well documented. This is important because sugarcane is prone to acidify the soil when ammonia-fertilizers are used. The effects of organic amendments have been studied (e.g., Braunbeck *et al.*, 1999; Ng Kee Kwong and Deville, 1988; Orlando Filho *et al.*, 1991; Sutton *et al.*, 1996) and several studies have followed the fate of applied nutrients. Most have focused on N because sugarcane is a large N consumer (Malavolta, 1994); less attention is given to K as sugarcane is often grown on soils in which the K status may be sufficient for sugarcane (de Geus, 1973). There has been little soil process-oriented research on P, possibly because sugarcane has a low P requirement (Malavolta, 1994).

### 2.5.1. Leaching

Comprehensive N work has been conducted at the Sugar Industry Research Institute in Reduit, Mauritius on Ustic Eutropepts (annual rainfall 1550 mm) and Dystropeptic Gibbsiorthox (annual rainfall 3700 mm). In a study, $^{15}N$-labeled was given as $(NH_4)_2SO_2$ or as $NaNO_3$ at the rate of

100 kg N ha$^{-1}$ (Ng Kee Kwong and Deville, 1984, 1987). The amount of N leached depended more on the duration and intensity of drying preceding rainfall than on the leachate volume. More N was leached from soils with higher organic matter content. Leaching was greater at the drier site but cumulative N loss over one year was similar for both soils; it appears that frequent, shorter, and less-intense drying and wetting cycles are as effective in mobilizing soil N as less frequent but longer and more-intense drying. The Oxisols were able to retain $NO_3$ by absorption, which reduced N leaching, but K and Ca were more readily leached than N. It was concluded that losses of cations might be more acute than the need for measures to minimize N leaching. Also in Australia, leaching losses were low under sugarcane (Chapman *et al.*, 1994; Wei-Ping *et al.*, 1993). A study on Grossarenic Paleudults in Florida (Unites States) showed that leaching losses varied from 6% to 24% of applied N depending on fertilizer type and irrigation level (El Wali *et al.*, 1980). Leaching of the applied N was mainly as $NO_3$ but when irrigation took place before, the N hydrolyzed from urea was completely nitrified, there was substantial leaching in the $NH_4$ form. Losses were lowest with sulfur-coated urea and increased with irrigation. Amounts of N loss ranged from 17% to 24% of the applied N, and upto 15% of the applied N was not accounted for by the plant, leachate, or soil. A study on Vertic Haplaquepts in Louisiana (United States) showed that N losses by leaching could be substantial (Southwick *et al.*, 1995). Average $NO_3$ leaching ranged from 15% to 60% depending on the leaching period and season.

de Oliveira *et al.* (2002) measured leaching of N and cations under sugarcane using lysimeters in São Paulo, Brazil. Inorganic N was applied but loss of N by leaching from the fertilizer ($^{15}N$) was not detected despite the heavy rainfall and irrigation. There were N losses but these originated from crop residues and the amount of N leached in 11 months was less than 5 kg N ha$^{-1}$. However, there were high rates of leaching for K, Ca, and Mg. Nitrogen losses under sugarcane may be low but cation losses may be considerable, which is a problem in acid soils. Moreover, not all N that is leached is lost. In very acid subsoils with an anion exchange capacity, $NO_3$ is adsorbed but it may be below the rooting zone of the crop (Dynia, 2000; Rasiah *et al.*, 2003). The maximum rooting depth of sugarcane is about 2 m, although there are considerable genotypic variations (Smith *et al.*, 2005). When the exchange capacity below the rooting zone is saturated, there may be leaching of $NO_3$ to the groundwater.

### 2.5.2. Gaseous losses

Many studies have shown that an appreciable fraction of fertilizer N invariably remains unaccounted (Allison, 1966). It is generally assumed that denitrification and volatilization of $NH_3$ are the major components of this unaccounted N. Weier *et al.* (1996) studied the potential for biological denitrification of fertilizer N in soils under sugarcane. In field studies on Alfisols and Ultisols, denitrification ranged from 1% to 20% of the applied N but differences between soil orders were considerable. In a glasshouse study, denitrification losses ranged from 13% to 39% of the N applied and the majority of the gaseous N loss occurred as $N_2$. It was concluded that denitrification is an important cause of fertilizer N loss from fine-textured soils, with $N_2O$ the gaseous N product when soil $NO_3$ concentrations are high (Weier *et al.*, 1996).

Fertilizer N uptake and N-use efficiency was investigated in Mauritius using $^{15}$N-labeled $(NH_4)_2SO_2$ at a rate of 100 kg N ha$^{-1}$. Approximately 10–20 kg ha$^{-1}$ of the labeled-N was lost from the green tops (i.e., the aerial parts of the sugarcane). These were gaseous N losses from the plant itself. Fertilizer N use efficiency derived from recovered N at harvest grossly underestimates the ability of sugarcane to use fertilizer N. The study showed that the uptake of fertilizer N varied from 20% to 40% whereas it was 13–18% when the measurements were made at the harvest. Denitrification and volatilization were grossly overestimated because losses of N from the aerial parts constitute a significant proportion of the unaccounted N (Ng Kee Kwong and Deville, 1994).

Most studies in Australia have focused on losses of N fertilizer applied on trash-harvested fields. Such losses can be high under relatively dry conditions when urea is applied. Sugarcane, like all plants, contains the enzyme urease that in trash breaks down urea into $CO_2$ and $NH_3$. The trash blanket cannot bind this $NH_3$ (Ralph, 1992) and volatilization of $NH_3$ can be one-third of the applied N. Wei-Ping *et al.* (1993) found losses of 20–30% of the applied N within 30 days after applications. In heavy rainfall areas, urea was washed from the trash and losses were 17% of the applied N, whereas ammonia losses from ammonia-sulfate were less than 2% of the applied N (Freney *et al.*, 1992). Similar losses were reported from Brazil (Gava *et al.*, 2003).

Chapman *et al.* (1994) investigated the efficiency of fertilizer N uptake using urea, labeled with $^{15}$N, which was either broadcast or buried in different trash management systems. The proportion of the applied fertilizer N recovered was 33% when the urea was buried and 18% when the urea was broadcast. It was suspected that denitrification accounted for the majority of the fertilizer-N loss (Chapman *et al.*, 1994). Drilling urea into the soil decreased the N losses although total losses remained relatively high (Prasertsak *et al.*, 2002). In the Australian research, gaseous N losses from the plant itself were not taken into account. Fertilizer N recovery was about the same as was found in Mauritius (20–40%). Low N-recovery values were also found sugarcane grown on Vertisols in Guadeloupe that is attributed to the higher rates of volatilization on these soils (Courtaillac *et al.*, 1998).

Leaching and denitrification are difficult to measure because of the inherent variability in the governing factors and the time needed for accurate measurements. Modeling contributes to our understanding of these processes and studies have used APSIM-SWIM to estimate N losses under sugarcane in Australia (Stewart *et al.*, 2006). Modeled N-fluxes through 1.5 m soil depth showed that of the 30 kg N $ha^{-1}$, about 27 kg N $ha^{-1}$ was taken up by the crop so only 3 kg N $ha^{-1}$ $year^{-1}$ was leached. In another model study, it was shown that $NO_3$ leaching was lower under rainfed conditions compared to irrigated conditions. Nitrogen leaching was strongly correlated with rainfall when application rates exceeded 200 kg N $ha^{-1}$; it was concluded that careful N management is needed under rainfed conditions (Thorburn *et al.*, 2005). Modeling studies may play an increasing role in the quantification of leaching and gaseous losses, both for experimental work and for decision making (Stewart *et al.*, 2006).

Low fertilizer-N recovery has been reported from many sugarcane areas. As N rarely accumulates in the soil, leaching and gaseous losses must be considerable with high rates of N applications. Leaching depends on the weather (rainfall), soil physical attributes (wetting, drying), the age of the sugarcane, as well as the type, quantity, timing, and placement of the fertilizer (Prasertsak *et al.*, 2002). The few detailed studies that have been conducted show that leaching losses are generally low despite the low N recovery of inorganic fertilizers. There is also evidence that gaseous losses are generally high from N fertilizers applied on trash-harvested fields and under poorly drained conditions. Related environmental issues are discussed in Section 6.2.

## 2.6. Nutrient balances

Nutrient balances can be used to estimate likely changes in soil chemical properties. In essence, they mimic an accounting procedure that compares inputs (e.g., inorganic fertilizers, manure) to nutrient outputs (e.g., crop removal, leaching) over a given time span. Nutrient balances may give insight into the processes that regulate nutrient cycling and help to formulate system management decisions and direct research (Hartemink, 2005a; Robertson, 1982).

Nutrient balance studies have been influential—agronomically and politically—in many tropical regions, but most suffer from methodological problems. The best quantified budget line is often nutrient removal by the crop. It is usually calculated as yield multiplied by nutrient uptake or removal data but nutrient uptake data are variable. For example, Hartemink (1997) showed that, based on 11 literature sources, nutrient removal of *Agava sisalana* varied from 27 to 33 kg N $ha^{-1}$, 5 to 7 kg P $ha^{-1}$, and 59 to 69 kg K $ha^{-1}$ per Mg of produce. The variation may be attributed to differences in sampling techniques, sampling period, inherent soil conditions, fertilizer applications, and analytical methods. Faerge and Magid (2004)

**Table 8** Partial N-balance (kg ha$^{-1}$ year$^{-1}$) for sugarcane cropping systems in Latin America and the Caribbean start Brazil, Ecuador,Peru etc at the same height as Dominician

| | Brazil | Ecuador | Dominican Republic | Peru | Trinidad | Range of values |
|---|---|---|---|---|---|---|
| *Inputs* | | | | | | |
| N fixation | 15–25 | nd | nd | nd | nd | nd |
| Inorganic fertilizer | 60–100 | 150 | 200 | 200 | 80 | 60–200 |
| Manure | 5 | 5–10 | 5–10 | 5–10 | 5–10 | 5–10 |
| Deposition | 5 | nd | 5–10 | nd | nd | 5–10 |
| Total | 100 | nd | nd | 282 | nd | nd |
| *Outputs* | | | | | | |
| Harvest | 50–60 | 50–60 | 50–60 | 150 | 50–60 | 50–150 |
| $NH_3$-volatilization | nd | nd | nd | nd | nd | nd |
| Leaching | nd | nd | nd | 20 | nd | nd |
| Burning | nd | nd | nd | 45 | nd | 30 |
| Forage | nd | nd | nd | 20 | nd | nd |
| Total | 100 | 100 | 100 | 230 | 100 | 100–230 |
| *Within system* | | | | | | |
| Fertilizer recovery (%) | 50 | 50 | 50 | 70 | 50 | 50–70 |

nd = no data. Modified from Ruschel *et al.* (1982).

concluded that losses are often overestimated and that modeled losses are rarely compared with direct measurements. Also, there are differences between years depending on the weather and other factors. As a result there is large interannual variation in nutrient balances (Sheldrick *et al.*, 2003)

A work group on sugarcane summarized the problems with nutrient balances in sugarcane as follows: A general N balance for this crop is difficult to construct because of widely differing agronomic practices and growing conditions and also a lack of knowledge of certain processes (Ruschel and Vose, 1982). The agronomic variation includes a growing period that ranges from 9 to 22 months, yields that range from 30 to 150 Mg ha$^{-1}$, preharvest burning or trash harvesting, and different inorganic fertilizer regimes and recycling practices for organic wastes. Partial N-balances for the sugarcane in some Latin American and Caribbean countries are presented in Table 8. A wide range of values was found for the sugarcane systems in the different countries. Inorganic fertilizer applications ranged from 60 to 200 kg N ha$^{-1}$ year$^{-1}$. Fertilizer recovery under sugarcane in Latin America is about 50%, which implies that the N balances, shown in Table 8, are negative in most countries. The fertilizer N recovery rate has substantial influence on the overall balance. In Mauritius, a fertilizer N recovery of 20–40% was found (Ng Kee Kwong and Deville, 1994), in Australia 20–50% (Vallis *et al*., 1996; Weier, 1994), and in India 16–45% (Singh *et al*., 2007). Higher recovery is reported in Brazil (Basanta *et al*., 2003). In the Peruvian case in Table 8, total N input after correcting for the fertilizer recovery is 182 kg N $ha^{-1}$ whereas total output is estimated to be 230 kg N $ha^{-1}$. Losses of N by burning were estimated to be 45 kg N $ha^{-1}$ and, in another study in Peru, it was shown that burning losses could account for 30% of the N output in a sugarcane system (Valdivia, 1982).

Denitrification and losses by erosion were not considered in the balances of Table 8 because the data were not available. In Louisiana (United States), erosion losses under sugarcane were about 17 Mg soil $ha^{-1}$ and annual nutrient losses by erosion were 18 kg N, 14 kg P, and 104 kg K $ha^{-1}$ (Bengtson *et al*., 1998). No data are available on nutrient losses with soil erosion from other sugarcane areas, but considerable amounts of nutrients can also be lost with soil erosion (Hartemink, 2006).

A partial N balance for sugarcane on an Entisol in the São Paulo region of Brazil showed that N added in vinasse and urea was insufficient to maintain the N levels in the 0.20 m topsoil, but in the 0–0.60 m soil layer, total N levels increased. The input by biological nitrogen fixation (BNF) caused a positive N balance (de Resende *et al*., 2006). An estimate of the major nutrient inputs and outputs at a sugarcane plantation in Papua New Guinea (Hartemink, 2003) used yield data from 1991 to 1995. The N balance was positive but the P and K balance was negative. The N recovery was not measured but if a 50% recovery assumed the N balance was also negative. In Coimbatore, India, a partial nutrient balance for sugarcane was calculated for sugarcane grown on Alfisols (Sundara and Subramanian, 1990). Data for the plant cane and first ratoon (2 years) are given in Table 9. More N and slightly more P was applied with the inorganic fertilizers than removed with the crop. The difference between K applied and removed was 137 kg $ha^{-1}$. Despite the positive balance of N

**Table 9** NPK balance and soil changes in a sugarcane field at Coimbatore, India

| | Balance (kg $ha^{-1}$ $year^{-1}$) | | | Soil changes (0–0.20 m) (kg $ha^{-1}$) | | |
|---|---|---|---|---|---|---|
| | Applied with inorganic fertilizer | Removed at harvest | Difference | Level at the beginning | Content after two years | Difference |
| N | 225 | 107 | +118 | 182 | 157 | –25 |
| P | 33 | 29 | +4 | 35 | 32 | –3 |
| K | 100 | 237 | –137 | 521 | 341 | –180 |

Calculated from Sundara and Subramanian (1990).

and P, the soil levels of N and P declined because not all outputs were measured and much of the applied N may have been lost. If a 50% recovery is assumed, the balance becomes negative and explains the loss of N from the topsoil.

There has been no study quantifying all nutrient inputs and outputs in sugarcane cultivation systems, and partial nutrient balances should be interpretated with caution (Hartemink, 2006). Although the removal of nutrients with the crop is fairly well-quantified (de Geus, 1973; Malavolta, 1994; Srivastava, 1992), other outputs and inputs of the nutrient balance have not been studied in sufficient detail, with the exception of BNF.

### 2.6.1. Biological nitrogen fixation

In the late 1950s, it was discovered that $N_2$-fixing bacteria of the genus *Beijerinckia* were present in the rhizosphere of sugarcane (Döbereiner, 1961). Most of the work on BNF in the sugarcane rhizosphere has been conducted in Brazil. Evidence for substantial inputs via $N_2$ fixation by sugarcane has been provided by isotope-dilution measurements; these are consistent with observations in the field (Chalk, 1991). In Brazil, an estimate of BNF holds that about 17% of total plant N is fixed by the sugarcane, which is equivalent to 17 kg ha$^{-1}$ at yields of 70 Mg ha$^{-1}$ (Ruschel and Vose, 1982). More recent results of $^{15}$N dilution/N balance studies showed that some sugarcane varieties can obtain larger contributions ranging from 60% to 80% of total plant N, equivalent to over 200 kg N ha$^{-1}$ year$^{-1}$ (Boddey *et al.*, 1991; de Oliveira *et al.*, 2006; Medeiros *et al.*, 2006). N-fixation is high for most Brazilian cultivars as they have been systematically bred for high yields with low N inputs (Boddey *et al.*, 1995).

Depending on the yield, 100–200 kg N ha$^{-1}$ is removed with the harvest. Annual N application rates on sugarcane in Brazil are on average 50 kg N ha$^{-1}$ (FAO, 2004), If over 200 kg N ha$^{-1}$ year$^{-1}$ is fixed biologically, it can be assumed that N levels in the soils under sugarcane are maintained (Boddey *et al.*, 2003; Lima *et al.*, 1987). However, ecosystems with high rates of N fixation often have high loss rates through leaching or possibly denitrification. The relationship is not fully understood but is related to the plant energy requirements when switching from uptake of atmospheric N to soil mineral N (Pastor and Binkley, 1998). It should be noted that the benefits of high N fixation in sugarcane may only be recorded in soils low in mineral N, when no or little inorganic N fertilizers are applied and when the soil P and Mo status is adequate. Cultivar differences in the potential for BNF are considerable (Urquiaga *et al.*, 1992) and water supply needs to be abundant for high fixation rates (Boddey *et al.*, 2003). These effects have been documented in Brazil, India, and Mexico (de Oliveira *et al.*, 2006; Medeiros *et al.*, 2006).

# 3. Changes in Soil Physical Properties

At most commercial sugarcane plantations, heavy machinery is used for land preparation, harvesting, and applications of fertilizers, herbicides, and pesticides. The machinery affects soil physical properties like aeration and porosity and the variation in soil physical properties, which is naturally already large within a field (Cassel and Lal, 1992), may be enhanced. In areas where most field work is done manually like in Mexico, soil physical changes are minimal under continuous sugarcane (Carrillo *et al.*, 2003; de la F *et al.*, 2006). This chapter discusses the effects of mechanized sugarcane cultivation on soil bulk density, aggregate stability, water intake (infiltration), and runoff and soil erosion.

## 3.1. Compaction and aggregate stability

Usually, sugarcane is grown in rows on low ridges (intrarows) with tractors and harvesters passing through the interrow. On Spodosols in Australia, McGarry *et al.* (1996a) found a topsoil bulk density of 1.55 Mg $m^{-3}$ in the intrarows as compared to 1.85 Mg $m^{-3}$ in the interrow. An adjoining uncultivated site had a topsoil bulk density of 1.40 Mg $m^{-3}$. Maclean (1975) and Wood (1985) reported significant increases in bulk density of 0.15–0.18 Mg $m^{-3}$ in the topsoil compared with uncultivated land; several other reports have confirmed compaction under mechanized harvesting in Australia (Braunack, 2004; Braunack and McGarry, 2006). In South Africa, Dominy and Haynes (2002) sampled Oxisols that had been cultivated for over 30 years with sugarcane and compared these to soils under native grassland. The topsoil bulk density was 1.17 Mg $m^{-3}$ under grassland but had increased to 1.37 Mg $m^{-3}$ under sugarcane. Below 0.10 m, there was little difference in the bulk densities of these soils and the increased bulk densities and lower water stable aggregates have negative effects on the growth and yield sugarcane. They also found that bulk density is generally higher in soils with burned sugarcane compared with soils under trash-harvested sugarcane (Graham and Haynes, 2006). Also in Brazil where much of the sugarcane is burned before harvesting, it was found that the topsoils of Oxisols after 6 years of cultivation had an increased bulk density (Ceddia *et al.*, 1999; Silva *et al.*, 2007).

Absolute and relative increases in soil bulk density are different for different soils. In Papua New Guinea, bulk densities under natural grassland and within the sugarcane rows were similar for all depths of both Fluvents and Vertisols (Table 10). The bulk densities in the interrow were significantly higher and roots were absent. The absolute increase in the topsoil bulk density of the interrow as compared to natural grassland was 0.22 Mg $m^{-3}$

**Table 10** Difference in bulk density between Fluvents and Vertisols under sugarcane and natural grassland for three depths

| Sampling depth (m) | Sugarcane | | | Natural grassland | | |
|---|---|---|---|---|---|---|
| | Fluvents | Vertisols | Difference | Fluvents | Vertisols | Difference |
| 0–0.15 | 1.19 | 1.09 | $p < 0.05$ | 1.07 | 1.00 | ns |
| 0.15–0.30 | 1.28 | 1.15 | $p < 0.01$ | 1.17 | 1.02 | $p < 0.05$ |
| 0.30–0.50 | 1.37 | 1.18 | $p < 0.001$ | 1.26 | 1.12 | $p < 0.05$ |

ns = not significant. Data from Hartemink (1998c).

(+21%) in the Fluvents and 0.18 Mg m$^{-3}$ (+18%) in the Vertisols. In Fluvents, the bulk density of the interrow increased to 0.50 m soil depth.

Soil compaction under sugarcane has been reported worldwide, including India (Rao and Narasimham, 1988; Srivastava, 1984), South Africa (Swinford and Boevey, 1984), Swaziland (Nixon and Simmonds, 2004), Mexico (Campos *et al.*, 2007; Vera *et al.*, 2003), Iran (Barzegar *et al.*, 2000), Brazil (Souza *et al.*, 2004), and Fiji (Masilaca *et al.*, 1985). It is a common problem (Yates, 1978) and it is likely to increase with higher rates of mechanization. Bulk density is higher in ratoons compared with soils that have just been planted. The fraction of water stable aggregates declines with increasing age of the sugarcane (Srivastava, 2003) and declines in soils where the sugarcane is burned before harvesting (Blair, 2000). Soil compaction may occur at once during field operations at moist soil conditions or may be cumulative during the years of cropping. Trouse and Humbert (1961) have shown that the topsoil bulk density of an Oxisol in Hawaii increased from 1.25 Mg m$^{-3}$ after 10 tractor passes to 1.43 Mg m$^{-3}$ after 20 passes, and to 1.53 Mg m$^{-3}$ after 50 tractor passes. Much depends on the ground pressure exerted by the tires of the field equipment and the soil moisture content at the time of field operations. Georges *et al.* (1985) found that water content was the most important factor affecting soil compaction and that equipment type had only a significant effect at high soil moisture contents. Similar findings were reported by Braunack *et al.* (1993) who found differences in bulk density between conventional tires and so-called high flotation equipment. In general, bulk densities were 0.1–0.3 Mg m$^{-3}$ higher under conventional equipment but conditions under which the experiments were conducted were fairly dry (Braunack *et al.*, 1993).

Compaction commonly results in an increase in soil strength. In South Africa, Swinford and Boevey (1984) found a penetrometer resistance of 220 N cm$^{-2}$ in fully compacted topsoils that reduced the root density from about 4 to 2.5 Mg m$^{-3}$ in uncompacted soils. Uncompacted soils had resistance values of about 140 N cm$^{-2}$. McGarry *et al.* (1997) observed soil resistance values in Spodosols in North Queensland of about 2500 kPa in the top 10 cm of the interrow whereas the resistance was less than 800 kPa in the intrarow. In Trinidad, Georges *et al.* (1985) found an increase in penetration resistance from 21 to 26 kg $cm^{-2}$ after wheel traffic on a clay soil.

Water intake is commonly reduced by an increase in topsoil bulk density. On Fluvents in Australia, Braunack *et al.* (1993) found differences in infiltration rates of 15–60% between the use of conventional and high flotation equipment. In Papua New Guinea, Hartemink (1998c) observed a negative exponential relation between topsoil bulk density and water intake of Vertisols and Fluvents. Bulk densities causing slow water intake (<50 mm $h^{-1}$) were about 1.20 Mg $m^{-3}$ in Fluvents and 1.16 Mg $m^{-3}$ in Vertisols. For both soil types, an increase of about 0.2 Mg $m^{-3}$ drastically reduced the water intake. Water intake in the interrow was less than 10% of the soils under natural grassland. The slow water intake in the interrows may result in soil erosion, which can be high on Vertisols.

Decreasing aggregate stability following loss of soil organic matter (see Section 2.4) may also cause increased soil bulk densities. In Vertisols in South Africa, it was found that aggregate stability was decreased following many years of inorganic fertilizer applications, particularly K. There was an increase in the proportion of monovalent cations (K, Na) and less Ca and Mg, which were leached. It favored dispersion, lowered stable soil aggregates, and increased soil bulk density (Graham *et al.*, 2002a).

Under sugarcane, the bulk density of different soils increases at different rates so it is difficult to establish a threshold bulk density value that affects the movement of air and water. Juang and Uehara (1971) mentioned that bulk density, in itself, is not a particularly useful index for predicting crop performance. It is, however, a good indication of what happens to the soil under continuous sugarcane cultivation. Although soil compaction is common, it can be relatively easily reversed. After 3 or 5 years when the sugarcane is plowed out and a new crop is planted, compacted soil layers may be broken up. Tillage usually lowers the bulk density, and sugarcane soils under zero tillage tend to have higher bulk densities than when the soil is tilled. Trash harvesting could lead to lower soil bulk densities because of increased soil organic matter contents (Srivastava, 2003).

## 3.2. Soil erosion

Soil erosion is a common problem under sugarcane. Some soils under sugarcane are heavy textured, for example, Vertisols that are erodible due to their low water infiltration rates after wetting (Ahmad, 1996). When planted, after harvesting, or with excessive furrow irrigation, soils may erode even if the land is nearly flat. In other soils, compaction may be accompanied by surface sealing that reduces infiltration and increases the likelihood for runoff and erosion. Also, the heat of preharvest burning makes the topsoil hydrophobic that decreases soil hydraulic conductivity (Robichaud and Hungerford, 2000) and increases the potential for runoff.

Putthacharoen *et al.* (1998) measured runoff and soil erosion under different arable crops on Quartzipsamments in Eastern Thailand. The experimental site was located on a 7% slope with annual rainfall 1300 mm. Runoff and sediment load were measured in ditches. Over a 50-month period, average annual soil erosion losses were 47 Mg ha$^{-1}$. Erosion was particular severe during the first 3 months after planting but once the crop was established, there was little erosion in the successive 2 years when the canopy protected the soil and contour rows reduced runoff. After 18 months, the sugarcane was trash harvested and erosion was minimal (Putthacharoen *et al.*, 1998).

A soil erosion study in the sugarcane areas of Australia, where the industry is largely confined to the high rainfall coastal zones (Johnson *et al.*, 1997), monitored soil erosion at seven sites with slopes ranging from 5% to 18% (Prove *et al.*, 1995). Soils were Oxisols and annual rainfall was 3300 mm. Soil erosion losses from conventionally cultivated ratoons were in the range of 47–505 Mg ha$^{-1}$ year$^{-1}$ with an average soil loss of 148 Mg ha$^{-1}$ year$^{-1}$. The variation was largely explained by the variation in the rainfall. Analyses of *in situ* and eroded soil indicated that sediment from no-tillage practices may be transported further from the erosion site and carry a more mobile fraction of nutrients (Prove *et al.*, 1995). A time series analysis of remote sensing imagery, daily rainfall, digital soil, and terrain maps combined with the universal soil loss equation and field observations showed that average erosion rates under sugarcane in Australia are 16 Mg ha$^{-1}$ (Lu *et al.*, 2003). Soil loss is particularly high in newly developed sugarcane lands (Brodie and Mitchell, 2005). In Louisiana (United States), soil erosion losses under sugarcane were on average 17 Mg ha$^{-1}$ (Bengtson *et al.*, 1998) but rainfall was lower than in the Australian study and ranged from 1300 to 1600 mm year$^{-1}$.

Various studies have been conducted in which soil erosion under sugarcane was not measured but modeled, based on remotely sensed images or models like Universal Soil Loss Equation (USLE) (Sparovek *et al.*, 2000). Erosion under sugarcane in Piracicaba in Southeastern Brazil was estimated to be 31 Mg soil ha$^{-1}$ (Sparovek and Schnug, 2001b). In South Taiwan, multitemporal remote sensing images and numerical simulation models were used to investigate soil erosion and nonpoint source pollution (Ning *et al.*, 2006). Total N and P measured in the runoff of sugarcane fields were six times larger than in the runoff of soils under forest. However, sugarcane made only a small contribution to total erosion and nutrient input into the river systems. In the upper northwest region of Thailand, sugarcane is an important crop and the area is expanding. Forest conversion to sugarcane accelerated soil erosion and, in some farms, the topsoil was completely eroded within 30 years of sugarcane cultivation (Sthiannopkao *et al.*, 2006).

### 3.2.1. Erosion control

The available evidence shows that soil erosion under sugarcane can be high, when it is immature, after burning and harvesting, and when the soil is compacted and infiltration reduced. Annual soil loss levels per hectare ranges from 47 Mg (Thailand), 16–505 Mg (Australia), 17 Mg (United States), and 31 Mg (Brazil). On most plantations, erosion control measures are taken: drains, bunds, ridges, strip cropping, and on heavy clays, strip tillage has proven successful to control erosion (de Boer, 1997). In Brazil, bench terracing following the contour is common practice to avoid runoff and soil erosion but the interest in reduced tillage and soil cover based methods to control erosion is increasing (Sparovek and Schnug, 2001a). As much of the sugarcane is cultivated on sloping land, the advantages and lower costs of harvesting mechanically on nonterraced and noncontoured fields do not encourage anti-erosion measures (Sparovek and Schnug, 2001a). Mechanical harvesting can be hindered by hilly relief but also by low labor costs (Gunkel *et al.*, 2007). It may restrict antierosion measures like terraces and contour farming.

In Australia, no-tillage practices reduced the rates of erosion to less than 15 Mg $ha^{-1}$ $year^{-1}$, and the effect of no-tillage was greater than the effect of a groundcover from trash harvesting (Prove *et al.*, 1995). A recent study in Louisiana focused on the effects of polyacrylamide (PAM) and crop residues to reduce erosion. In the area, agriculture accounts for up to two-thirds of the nonpoint source pollutions and sediments with absorbed pesticides, metals, and nutrients deteriorate aquatic life in the rivers. The addition of PAM to the irrigation water had no effect on sediment load, whereas sugarcane residues significantly reduced soil erosion. Adding PAM as a water solution had no effects on the erosion in the drains, possibly as PAM is degraded by exposure to UV radiation (Kornecki *et al.*, 2006). However, when PAM was applied directly to the primary quarter-drains, soil erosion was significantly reduced (Kornecki *et al.*, 2005).

## 4. Changes in Soil Biological Properties

Changes in the soil physical and chemical properties as a result of continuous sugarcane cultivation affect the biological properties of the soils. Increasing acidity and decreasing soil organic matter as well as increased bulk density and reduced porosity and aeration cause changes in the quantity and diversity of soil life. Likewise, a change in the soil biological properties influences the chemical and physical properties of the soil. Only a few studies of this interrelationship are available (Table 2).

## 4.1. Macrofauna

The abundance of fire ants was investigated in different soil types in the sugarcane-growing areas of Louisiana, United States (Ali *et al.*, 1986). The ants were found in highest numbers in Vertisols possibly related to the higher soil fertility and moisture content, and lower bulk density. In these clay soils, herbicides are better degraded and sorbed, which favors ants. Increasing ground cover of weeds and trash increased the number of ants, which are predators of the insect pests in sugarcane.

In Hydrandepts under long-term sugarcane in Hawaii, no earthworms were present but earthworms were present and increased after the land was reforestated (Zou and Bashkin, 1998). This was attributed to an increase in soil organic C and N and a higher pH. Earthworm abundance and diversity has also been researched in the sugarcane fields on Oxisols of Paraná state, Brazil (Nunes *et al.*, 2006). Almost 300 earthworm species have been recorded in Brazilian soils but in the sugarcane soils only 6 species were identified. Fewer individuals and species were found in soils under sugarcane compared with pastures, but the lowest number of earthworms were found under forest. Dearth of earthworms under sugarcane was the effect of tillage (plowing, disking). Under sugarcane, native species are lost and exotic species dominate (Nunes *et al.*, 2006).

In South African Oxisols, earthworm abundance, biomass, and number of species were investigated under sugarcane and several other land uses (Dlamini and Haynes, 2004; Haynes *et al.*, 2003). Numbers of earthworms, biomass, and the number of species were lowest under sugarcane compared to soils under pasture or forest. Under sugarcane, twice as many worms were found in the plant rows as the interrow is more compacted that lowers earthworm activity as roots were absent and there was low C turnover. Earthworm numbers and biomass were closely correlated with soluble C, microbial biomass activity, and the pH. There were more worms under trash-harvested sugarcane. As was found in Brazil, the earthworms in the soils under sugarcane were mostly exotic species (Dlamini and Haynes, 2004; Haynes *et al.*, 2003). Accidentally introduced worm species dominate in many agricultural soils (Fragoso *et al.*, 1997).

The effect of burning on the insect community was investigated in Oratorios, Brazil (Araujo *et al.*, 2005). In this area, fire is used to control pests and diseases but the effects on insect populations are poorly understood. The number of insects was reduced by burning but the insect population soon recovered after the sugarcane was burned.

The few available studies suggest that both the population and abundance of the macrofauna are changed under sugarcane cultivation. Tillage, decreased C input, and burning may be the primary causes. The effects of these changes on overall soil functioning as well as on sugarcane production are yet to be quantified. Also the effects of trash harvesting and pesticide and herbicide applications on the soil macrofauna have not been well studied.

## 4.2. Microbes

Measurements of microbial biomass have been made in cultivated and uncultivated sites in Australia: McGarry *et al.* (1996a) found large reductions in microbial biomass following cultivation; they suggested that the decrease was a result of the use of pesticides. Holt and Mayer (1998) quantified microbial biomass in new and old sugarcane fields in Australia. Significantly lower microbial biomass was found in soils under long-term sugarcane (Table 11). Microbial biomass rapidly reduces after the introduction of sugarcane. Garside *et al.* (1997) observed that soil microbial biomass was significantly lower on old sugarcane land than on new land, again concluding that there is a rapid loss of soil microbial biomass under sugarcane, which was also observed in Oxisols in Swaziland (Henry and Ellis, 1995). The cause for such decline is not established but may be related to the use of inorganic fertilizers and biocides, and the reduction in soil organic matter. In South Africa, the effects of inorganic fertilizers on microbial biomass have shown mixed results. In some cases, microbial biomass increased whereas the fertilizer N-induced soil acidification reduced the microbial activity and the activity of exocellular enzymes (Graham and Haynes, 2005).

In Australia, Pankhurst *et al.* (2005a) investigated the effects of soil organisms on sugarcane yield. Root rot fungus and nematodes increase with continuous sugarcane cultivation but long fallows increased biological suppression of soil organisms that may cause yield decline. Root lesion nematodes decrease under fallow but the effects are short-lived (Pankhurst *et al.*, 2005b). Magarey *et al.* (1997) sampled soils continuously cropped with sugarcane and from land that has never been cultivated (Table 12). Higher levels of some fungal pathogens as well nematodes were found under permanent sugarcane but no clear picture emerged of relationships between fungi, bacteria, and actinomycetes and land use. It was concluded that yield

**Table 11** Microbial biomass carbon at six sites in Queensland, Australia

| | Microbial biomass ($\mu$g C g$^{-1}$ soil) | | |
|---|---|---|---|
| Site | New land[a] | Old land[b] | Difference |
| Tully | 591 ± 155 | 357 ± 45 | $p < 0.05$ |
| Costanzo | 590 ± 279 | 519 ± 295 | ns |
| Harney | 192 ± 20 | 216 ± 11 | ns |
| Fortini | 372 ± 57 | 125 ± 14 | $p < 0.001$ |
| Ingham | 732 ± 73 | 313 ± 65 | $p < 0.001$ |
| Kalamia | 336 ± 134 | 160 ± 70 | $p < 0.05$ |

[a] New land is land that not been under sugarcane before or had been cultivated less than 6 months.
[b] Old land is land that has been cultivated with sugarcane for prolonged periods.
Type II data, modified from Pankhurst 2005b.

**Table 12** Soil biological properties under permanent sugarcane, grassland, and rainforest in northeast Australia

| | Sugarcane | Grassland | Rainforest |
|---|---|---|---|
| Total fungi ($\times 10^6$ g$^{-1}$) | 4.2 | 2.2 | 3.4 |
| Total bacteria ($\times 10^8$ g$^{-1}$) | 4.1 | 3.7 | 4.1 |
| Total actinomycetes ($\times 10^6$ g$^{-1}$) | 5.4 | 48 | 21.8 |
| Fungal pathogens | | | |
| *Pachymetra chaunorhiza* (spores g soil$^{-1}$) | 36 | 0 | 0 |
| *Pythium* spp. (% baits colonized) | 17 | 33 | 17 |
| Nematodes | | | |
| *Pratylenchus zeae* (nematodes kg$^{-1}$) | 273 | 0 | 0 |
| *Helicotylenchus* spp. (nematodes kg$^{-1}$) | 273 | 0 | 0 |

Type II data, modified from Magarey *et al.* (1997).

decline has a major biological component (Magarey *et al.*, 1997), possibly as reduced microbial activity results from decreased soil organic C, which was also found in South Africa (Dominy and Haynes, 2002; Graham and Haynes, 2005) and India (Suman *et al.*, 2006).

In several parts of the world, sugarcane is irrigated. This affects the soil moisture regime and thus the microbial activity. A study in Zimbabwe investigated the effects of irrigation-induced salinity on microbes. Soils were sampled under dead and dying sugarcane, poor, satisfactory, and good cane growth, and from adjacent sites under native vegetation. Increasing salinity and sodicity resulted in a progressively smaller, more stressed microbial community that was less metabolically efficient. Agriculture-induced salinity and sodicity influences the chemical and physical characteristics of soils and greatly affects soil microbial and biochemical properties (Rietz and Haynes, 2003).

Changes in soil microbial biomass are closely related to changes in the soil organic matter status; the microbial biomass is governed by the same factors. A reduction is commonly perceived to negatively affect the soils productivity through, for example, reduced organic matter mineralization. The soil biological component of sugarcane cultivation has been stressed in various studies and is of importance for improved and sustained production.

## 5. Environmental Issues

Environmental issues resulting from continuous sugarcane cultivation for bioethanol production can be explored at different scales. Energy production is important but also the energy consumption of the production system, now more and more sugarcane is being harvested mechanically and increasing rates of inorganic fertilizers are being used. The production of greenhouse gasses is of concern, including the release of methane and $NO_x$ by preharvest burning and inorganic fertilizers, and it may be enhanced in trash-harvested systems because of higher soil moisture contents (Macedo, 1998; Tominaga *et al.*, 2002).

Locally, contamination of the soil and water resources may occur. Commercial sugarcane is usually grown with herbicides that represent about 50% of all biocides used in many countries (Lanchote *et al.*, 2000). Most studies on the environmental impact of sugarcane have focused on the off-site effects including deterioration of surface water and air quality. Nearly all studies have been conducted in the United States and Australia, although a few have been conducted in Barbados and Brazil, reflecting differences in research priorities and capacities between economic regions (Bouma and Hartemink, 2002; Hartemink, 2002).

The Australian sugarcane industry is adjacent to the environmentally sensitive areas Great Barrier Reef and rainforests. The industry is intensifying with fewer and larger farms, using more fertilizers, continuous cropping, and utilizing more marginal soils (Gourley and Ridley, 2005). As reported in Section 3.2, soil erosion rates can be high but the precise rate and impact of sediment delivery to estuarine and marine environment is not well understood (Johnson *et al.*, 1997). Sugarcane production has significant impact on riverine water quality compared to grazing or forestry (Brodie and Mitchell, 2005). This is mainly because of higher N, P, and suspended solids in streamwater draining from highly fertilized sugarcane lands. Nutrient levels may have increased over the years and Rayment and Bloesch (2006) compiled soil acid P soil tests data of 105 sugarcane sites in Australia; they found that since the 1950s, P levels had increased from about 40 to over 100 mg $kg^{-1}$. High rates of P applications resulted in high levels of P in the soil, with risks for leaking to the groundwater.

Both the sugarcane industry and the broader community realize the potentially adverse ecological effects of discharges to the Great Barrier Reef lagoon. The deterioration of surface and groundwater quality is perceived by the farmers to be a consequence of new farm management strategies (Arakel *et al.*, 1993). Several measures have been taken following concerns about the downstream effects (Bunn *et al.*, 1997). In Australia, the policy has been toward a voluntary rather than regulatory approach and the industry has drawn up a national program to raise awareness among growers and introduced a "Sustainability in sugar" checklist (Gourley and Ridley, 2005). An acute concern in coastal areas is the drainage and oxidation of acid sulfate soils. Almost 10% of the soils under sugarcane in Australia are underlain by acid sulfate soils: 18,000 ha in New South Wales, 20,900 ha in South Queensland, and an unknown area in far North Queensland. Drainage of these soils has lead to acidity discharge carrying heavy metals and arsenic into aquatic ecosystems. Such discharge has extremely negative effects on the environment (Dent and Pons, 1995; Kinsela and Melville, 2004). The industry in Australia has responded with management practices to minimize the hazard, notably by avoiding deep drainage, and national legislation now prohibits development of these soils. Such soils are also found in Guyana (Dent, 1986) and any expansion of sugarcane for bioethanol (and oil palm for biodiesel) on coastal plains needs to consider this issue.

### 5.1. Herbicides and pesticides

Herbicides like atrazine, diuron, 2,4-D, and alachlor are extensively used in sugarcane cultivation. Atrazine is probably the most widely used herbicide in sugarcane. These herbicides are water soluble and there is a concern that they may contaminate the soil, vadose zone, and surface and groundwaters. They may leach from sugarcane fields and that may take place along preferential flow paths and cracks in clay soils (McMartin *et al.*, 2003). The herbicides may also wash off the land by runoff and soil erosion. There is some difference in the risks in using these herbicides: atrazine and ametryne are mostly degraded by sunlight and alachlor dissipates faster than atrazine (Javaroni *et al.*, 1999).

Southwick *et al.* (1992) measured atrazine leaching on Vertic Haplaquepts under sugarcane in Louisiana. Maximum concentrations, found within 11 days after application, ranged from 82 to 403 $\mu$g liter$^{-1}$. The lifetime health advisory limit for drinking water in the United States is 3 $\mu$g liter$^{-1}$; this concentration was reached in 20–30 days after application (Southwick *et al.*, 1992). Southwick *et al.* (1995) also measured leaching of atrazine and metribuzin: leaching of both herbicides was high directly after application but decreased after some weeks. Total losses ranged from 0.4% to 2.0% for atrazine and from 0.4% to 1.7% for metribuzin. Atrazine concentrations in the drainage water were again above the United States health advisory levels but the lifetime health advisory limit for metribuzin was not reached (Southwick *et al.*, 1995). Water quality data collected over several years in the sugarcane area showed that one in five detections of atrazine is above the maximum contaminant level for drinking water (Southwick *et al.*, 2002). Although many factors are involved, the method of application is a main factor determining the rate of herbicide loss (Bengtson *et al.*, 1998).

In Brazil, Lanchote *et al.* (2000) measured residues of atrazine, simazin, and ametryne in surface and groundwater collected in a sugarcane area near São Paulo. Ten water-sampling points were selected in a watershed, of which nine were taken from surface water and one from groundwater. In total, 250 samples were collected but atrazine residues were detected in only 17 samples. The concentrations were below those recommended as safe by international agencies of environmental control. Leaching and half-life of the herbicide tebuthiuron were examined under sugarcane in Santa Rita do Passa Quattro, São Paulo State. Soils were Typic Quartzipsamments. The herbicide was applied and soil samples were taken at different depths at regular intervals up to 300 days after the application. No herbicide residues were found and there was much more rapid degradation and less mobility than previously assumed; half-life was 20 days and after 180 days, there was no measurable residue in the soil (Cerdeira *et al.*, 2007).

In Mauritius, the leaching of herbicides was measured under sugarcane on Vertisols and Andosols. There was a lower risk of herbicide leaching than in temperate regions due to the high temperatures and the highly adsorbent soils. The herbicides were moderately to very immobile, although there was a considerable difference between the herbicides and the two soil types. Overall, the potential for leaching was considered very low, but during the 40 days per year when there are cyclones with high rainfall intensities, there may be considerable leaching losses (Bernard *et al.*, 2005).

Most environmental impact studies in sugarcane have focused on herbicides and few on pesticides because they are used less frequently. On an active ingredient basis, about 11 times more herbicides than pesticides are used in the Australian sugarcane industry (Arthington *et al.*, 1997). Environmental regulations caused a shift in the use of biocides in Australia: from the early 1950s until the late 1980s, organochlorine pesticides were widely used but were banned in the 1980s. A survey has investigated residues in sugarcane soils and in the coastal alluvial floodplains. Marine surface sediment samples and three sediment cores had no detectable levels of organochlorine pesticides, but easily detectable concentrations were found in the soils under sugarcane. It is likely that these pesticides move from the sugarcane soils to the near-shore marine environment by runoff and soil erosion (Cavanagh *et al.*, 1999).

Until 1985, persistent organochlorine compounds such as aldrin and heptachlor were also commonly used as insecticides on sugarcane in Brazil. Traces of these insecticides were investigated in soils, colluvium, submerged sediments, and organisms (worms, larvae) in a watershed in a sugarcane area. Most insecticides applied in the past were not detected, but organochlorine compounds that remained on the market after 1985 were detectable in significant amounts. It was concluded that a complete ban is probably the only solution for avoiding the dispersion of these products into the environment (Sparovek *et al.*, 2001).

Leaching of herbicides and pesticides is of serious concern wherever high input agriculture is practiced. Earlier work in Hawaii has shown that leaching of herbicides under sugarcane is negligible because of high adsorption rates in the soil (Hilton and Yuen, 1966). The available evidence shows no serious leaching losses under sugarcane because many soils have a high clay content (Vertisols), organic matter content (Histosols), or organic matter content. More herbicides and pesticides are lost through erosion and runoff but trash harvesting and zero-tillage reduces the risk for such losses. As zero tillage is often combined with higher levels of herbicide use, further studies are needed to investigate how such systems affect herbicide losses.

## 5.2. Inorganic fertilizers

Section 2.5 summarized reports on nutrient losses from fertilized sugarcane systems. High amounts of fertilizer N in the sugarcane and the contamination of surface and groundwater are a concern (Baisre, 2006). Nitrogen concentrations are often not considered to be an important criterion of surface water quality because of denitrification and biological cycling in an open system (Anderson and Rosendahl, 1997). However, there could be increased use of inorganic fertilizers to boost sugarcane production following high ethanol prices and increasing environmental impact regulations for sugarcane growers may follow.

### 5.2.1. Nitrogen

Leaching of N is very likely as in most sugarcane areas application rates are high, rainfall or irrigation are abundant, and N use recovery is low. Studies on N leaching under sugarcane in Brazil have been limited—possibly as N applications are low (40–60 kg N ha$^{-1}$) and much of the N in sugarcane is derived from BNF (de Oliveira *et al.*, 2006)—see also Section 2.6. In Australia, N is applied at rates of 150–300 kg N ha$^{-1}$ and excess N has been linked to $NO_3$ contamination of water supplies as well as contributing to greenhouse gas emissions (Dalal *et al.*, 2003; Gourley and Ridley, 2005; Weier, 1998). Surface water quality in forested wetlands of Louisiana is being reduced by nutrient input from adjacent agricultural production areas. A $^{15}$N study was undertaken to assess the input of fertilizer N applied to sugarcane fields and to forested wetlands (Lindau *et al.*, 1997). The major soil orders were poorly drained Vertic Haplaquepts and Aeric Fluvaquents. Fertilizer N draining into adjacent forested wetlands was estimated to be only a small fraction of the amount applied and concentrations of $NH_4$ and $NO_3$ were low. About 3–4% of the applied N was removed by runoff. Even after anhydrous $NH_3$ application, no increase was observed in the $NH_4$ and $NO_3$ concentration. This was explained by the high clay contents of the soil and the injection of the anhydrous $NH_3$ at 0.10–0.15 m below the soil surface. In another study in Louisiana, it was found that $NO_3$ and P were present in the surface water but not at high levels and it could also not be directly linked to sugarcane cultivation (Southwick *et al.*, 2002).

In order to reduce N losses on sugarcane plantations in Mauritius, research has focused on the use of drip-fertigation (Ng Kee Kwong *et al.*, 1999).

Applications of fertilizer N could be reduced by 30% from 120 to 80 kg N $ha^{-1}$ $year^{-1}$ without a reduction in growth pattern or sugarcane yields. However, investments in drip-fertigation are large and it may not be economically and technically feasible for all sugar producing areas. Alternatively, increased plant densities may reduce the leaching of N in sugarcane systems (Yadav and Prasad, 1997).

### 5.2.2. Phosphorus

In Australia, P is applied at rates of 15–50 kg $ha^{-1}$. Application rates are often in excess of recommendation to avoid the risk of P-limited yields (Bramley *et al.*, 2003; Thorburn *et al.*, 2005). Application rates do not take into account the differences between different soils in their ability to release or sorb P; industry recommendations do not consider soil properties (Edis *et al.*, 2002). Also, many sugarcane soils have considerable mycorrhizal density that may enhance the P supply (Kelly *et al.*, 2005). As a result, many soils under sugarcane are well supplied with P. This is not necessary an advantage—in the United States, it was found that high soil P levels may increase rust severity (Johnson *et al.*, 2007a). An evaluation of 105 sugarcane sites in Australia showed that 84% of all soils sampled had excessive P levels, following annual applications of 20 kg P $ha^{-1}$. It will take a long time to deplete the high soil P levels (Rayment and Bloesch, 2006). Also in Brazil, continual fertilization with P has lead to high levels of organic and inorganic P in the topsoils (Ball-Coelho *et al.*, 1993).

Soil erosion, fertilizer P loss, and groundwater flow result in blue-green algae and excessive growth of aquatic macrophytes (Arthington *et al.*, 1997). Algal blooms are also increased by reduced water flow—a problem that occurs in the sugarcane areas of Everglades (United States) (Anderson and Rosendahl, 1997) where regulatory program with best management practices was introduced and has considerably reduced the P level in drainage waters of sugarcane farms (Rice *et al.*, 2002). It needs to be ascertained whether current P levels are acceptable for South Florida wetlands. Such assessment may be hard to make because there is a difference between freshwater and marine water on the response to increased P input from agricultural drainage waters. In Australia, it was found that environmental problems posed by P attached to sediments from sugarcane land is likely to be greater in freshwater than in marine ecosystems (Edis *et al.*, 2002).

Several cultural practices could reduce the loss of biocides from sugarcane fields: Southwick *et al.* (2002) reported variable success in the reduction of runoff losses of biocides as a result of conservation tillage; subsurface drains that increase infiltration seem to be more effective to reduce runoff and reductions up to 25% have been reported; filter strips or water settling areas may also reduce runoff, soil erosion, and sediment loss (Southwick *et al.*, 2002).

### 5.2.3. Heavy metals and rare earth elements

Little is known about heavy metal accumulation in sugarcane systems although flora and fauna are affected by even low concentrations. Reports from Australia have shown a sevenfold increase in Cd in the topsoils under sugarcane compared with uncultivated sites (Arthington *et al.*, 1997), probably caused by Cd contamination in P fertilizers, which is common (Kirkham, 2006). It is suspected that preharvest burning could dissipate Cd while trash blankets may concentrate Cd at the soil surface, where it could be eroded (Arthington *et al.*, 1997).

In the sugarcane areas of southern China, Chua *et al.* (1998) investigated the accumulation of cerium (Ce), a rare earth element (REE), nonradioactive, and moderately toxic. It was shown that Ce entered the sugarcane plants via the leaves exposed to atmospheric contaminants, via the roots in soils contaminated by Ce and other REEs, or applied with inorganic fertilizers (Rodriguez-Barrueco, 1996). Official limits to residual concentrations are not available but high REE concentrations in the soils under sugarcane could lead to harmful effects for humans consuming sugarcane products (Chua *et al.*, 1998).

In the Everglades of Florida, serious mercury contamination of freshwater fish in 1989 was related to preharvest burning of sugarcane (Patrick *et al.*, 1994). Soils are Histosols and the average mercury content of the Histosols was only 0.15 mg $kg^{-1}$; Hg concentrations in the sugarcane stalks were also low. The study concluded that direct emission of Hg from sugarcane fields during preharvest burning was only a minor source (2%) of atmospheric Hg and left open the question on the origin of the Hg contamination.

## 5.3. Air and water quality

The bioethanol program in Brazil started after the oil crisis in 1973 with the aim to make the country less dependent on imported oil. A big industry developed and gasoline has been replaced in large measure by ethanol from sugarcane. The cleaner air in the cities has been at the expense of increased smoke from preharvest burning in sugarcane areas. Depending on the amounts of crop residues, over 3000 kg $ha^{-1}$ of C is released as $CO_2$; the smoke is a health problem in many areas and of particular concern in newly developed suburban areas adjacent to plantations (Kornecki *et al.*, 2006). The smoke contains respirable particles that have a size less than 10 $\mu$m (Boopathy *et al.*, 2002). Research in Brazil has shown that increases in total suspended particles generated from preharvest burning were associated with asthma hospital admissions (Arbex *et al.*, 2007). Also in Louisiana, smoke from burning sugarcane accounts for much air pollution (Kornecki *et al.*, 2006) and a link was found between asthma admissions hospital visitations and sugarcane burning. As the prevalence of asthma in both adults and children is rising in many parts of the world, detailed studies on the effects of weather, pollen counts, and air pollution from sugarcane burning and the pollution from other sources are needed (Boopathy *et al.*, 2002).

Water quality is affected by sugarcane cultivation through loss of biocides and inorganic fertilizers (see Sections 2.5 and 5.1) and through sugarcane processing plants. These plants produce waste waters (stillage, vinasse) that are used for fertigation but some are discharged in streams and rivers like in the sugarcane areas in Cuba (Rosabal *et al.*, 2007). In Pernambuco in the northeast of Brazil, the waste water heats the riverwater, contains organic acids, and has a high biological oxygen demand (Gunkel *et al.*, 2007)—all of which deplete aquatic life. A number of treatment options exist including wastewater lagoons, trickling filters, and activated sludge systems. In areas with high risks of water pollution, changes in land use and reforestation may be the only options (Gunkel *et al.*, 2007).

# 6. Discussion and Conclusions

Sugarcane is a major cash crop, increasingly used for bioethanol production. Given the increase in oil prices coupled to the demand for renewable energy sources, it is likely that the area under sugarcane will further expand. Both expansion and further intensification (fewer and bigger farms) affect the soil and wider environment.

## 6.1. Sugarcane for bioethanol

Sugarcane is an ideal crop for renewable energy because of its rapid growth and high energy production per hectare. Fossil energy is needed for growing of the crop and the production of bioethanol, which partly offsets the energy produced. In Brazil, fossil energy costs are minimized by the use of processing products like bagasse for energy. The energy balance (yield over fossil energy) of such systems may range from 9 to 11 (Macedo, 1998), which compares very favorably to many other biofuel crops. In part, this favorable balance is explained by the relatively low N application rates to sugarcane in Brazil because of the high rates of BNF. In many agricultural systems, inorganic fertilizers are a major budget line. Overall, BNF can be considered one of the principal reasons for the success of the bioethanol program in Brazil (Medeiros *et al.*, 2006).

Several cultural practices that reduce the energy demand for growing sugarcane. Tillage before planting requires about one-third of the total operational energy. Zero tillage seems to have little effect on crop yield whereas mechanical trash harvesting increases the energy demand as compared to preharvest burning (Srivastava, 2003). However, preharvest burning is increasingly criticized because of public health issues related to the smoke and because of the loss of beneficial crop residues. Thus, the use of bioethanol has effectively reduced air pollution in many cities but the urban areas surrounding sugarcane areas are largely affected by the preharvest burning, which takes place about 6 months per year (Arbex *et al.*, 2007).

Another aspect that deserves discussion is the expansion of sugarcane in relation to land used for food production. It is projected that in many industrialized regions, the area under agriculture will decrease whereas the area under agriculture in developing regions is increasing (Smeets *et al.*, 2007). Most of the human population increase takes place in the developing regions (Fischer and Heilig, 1997) where the need to increase crop production is largest (Sanchez, 2002; Swaminathan, 2006). It has been calculated that 55% of the present global agricultural land will be needed for food production in the year 2050, if high external input agriculture is practiced (Wolf *et al.*, 2003). The remaining 45% can be used for other purposes including biofuels. There will be no land available for biomass production when low external input agriculture is practiced (Wolf *et al.*, 2003). Little new land is available in developing regions (Young, 1999) so crop production for food and biofuel competes for the same land area. Some expansion is possible through the clearing of forest or savanna, but most of the increased biomass production needs to come from intensification of the present systems.

According to Hill *et al.* (2006), a biofuel should provide a net energy gain, have environmental benefits, be economically competitive, and able to be produced in large quantities. Sugarcane for bioethanol can fulfill these criteria. The net energy gain is several times the input and it is economically grown in many countries without the subsidies that other biofuel crops receive (e.g., corn in the United States or rapeseed in Europe). It is not affecting staple food production in the United States or Australia—where it is grown for sugar. In Brazil and some other tropical countries where sugarcane is mainly grown for bioethanol, a further increase may compete with food production; that assessment is yet to be made.

## 6.2. Effects on the soil

Most studies have shown that soil acidification takes place under sugarcane, principally due to the use of N fertilizers containing or producing $NH_4^+$. All ammoniacal N fertilizers release protons when $NH_4^+$ is oxidized to $NO_3^-$ by nitrifying microorganisms. Also, mineralization of organic matter can contribute to soil acidity by the oxidation of N and S to $HNO_3$ and $H_2SO_4$ (Sumner, 1997). Because organic matter declined in most soils under sugarcane, it may have contributed to the increase in soil acidity. Acidity is reversible; liming readily restores productivity but if acidification has also taken place in the subsoil, amelioration is much more difficult. There is only a small response of sugarcane to lime on moderately acid soils (Turner *et al.*, 1992) whereas in other studies, a decrease in the sugar content was found after lime applications (Kingston *et al.*, 1996). Sugarcane is fairly tolerant of acidity and high concentrations of exchangeable and soluble Al (Hetherington *et al.*, 1988); avoiding strong soil acidification might be a better option than the use of lime to correct for high acidity inputs.

Soil organic C dynamics have received much attention in sugarcane, but there are some conflicting reports. A part of the problem is that total soil organic C determined by the Walkley and Black or the dry combustion method is not very sensitive to short-term changes in land use. Long-term observations are required to pick up statistically significant differences in soil organic C levels. It is also related to the spatial variability in total soil organic C. Notwithstanding these methodological problems, total soil organic C decreased in most topsoils and in most soil types. This may be the effect of tillage that causes increased soil organic matter decomposition compared with soils under natural ecosystems, but, also, because of lower inputs of organic matter in sugarcane systems. Soil texture plays an important role in the rate of change in soil organic C and this change also differs for different size fractions. An equilibrium is reached after many years but it is generally lower than the initial level in the soil under forest. In a number of soils, it was found that levels of soil organic C increased in the subsoil. The decrease in soil organic matter under continuous sugarcane reduces soil biological activity and increases the susceptibility of the soils to physical degradation.

Soil compaction is a common problem in mechanized systems, mainly due to the heavy machinery used for field operations at the wrong soil moisture levels. Also, frequent applications of inorganic fertilizers may lower soil aggregate stability of some soils (Graham *et al.*, 2002a) and increase the bulk density and lower the rates of water infiltration (Mills and Fey, 2003). Erosion losses up to 505 Mg soil $ha^{-1}$ $year^{-1}$ have been reported under sugarcane. Erosion can be high after the harvest and with replanting, especially on sloping land (Blackburn, 1984). Sugarcane is more prone to soil erosion than other perennial crops because the periodic harvesting removes almost all vegetation from the field (Hartemink, 2005b). On the other hand, sugarcane covers the soil in most parts of the year so reduces the risk for soil erosion. Erosion means loss of productive topsoils but also sedimentation in the lower part of the catchment that may cause environmental problems. In Australia, there seems to be little evidence to support claims that sediment deposition resulting from sugarcane cultivation has had a major impact on the characteristics of the rivers and sugar catchments over the last 50–100 years (Johnson *et al.*, 1997). However, there is increasing concern about the erosional effects and green harvesting methods have been advocated to reduce soil erosion (Wood, 1991).

The advantages of green or trash harvesting also include soil water conservation, reduced soil temperature, increased soil fertility and soil organic matter, and improved soil structure. Residue burning leads to a loss of N, reduced soil organic matter with deterioration of physical and microbiological properties, and an increase in greenhouse gases (Basanta *et al.*, 2003). The trash has nematocidal properties (Akhtar, 1993), combats weeds, and there are also more roots in trash-harvested systems that increase nutrient uptake, particularly P (Ball-Coelho *et al.*, 1993). Several reports have indicated that the trash harvesting has the potential to maintain or increase soil organic carbon contents (de Resende *et al.*, 2006; Noble *et al.*, 2003; Razafimbelo *et al.*, 2006) and reduce the susceptibility of the soil to compaction (Barzegar *et al.*, 2000).

There are also some disadvantages. Trash may hinder tillage, reduce nutrient availability through immobilization, and cause waterlogging resulting in N losses through denitrification especially on poorly drained soils (Wood, 1991). Some studies have found that sugarcane trash is allelopathic (Viator *et al.*, 2006). In cooler areas (e.g., Louisiana, United States, or North South Wales, Australia), the trash results in increased soil moisture and lower soil temperatures that not only delay the reemergence of a ratoon crop but can also increase sugarcane infection by parasitic soil fungi (Viator *et al.*, 2005). Consequently, sugarcane yield may be reduced by 4.5–13.5 Mg ha$^{-1}$ (Johnson *et al.*, 2007b).

Trash quantities are large (7–12 Mg DM ha$^{-1}$) and contain high amounts of C (3–5 Mg C ha$^{-1}$) and N (28–54 kg N ha$^{-1}$). In the United States, a trash-harvested field may contain up to 24 Mg DM ha$^{-1}$. The C/N ratio is typically over 70. In some studies, it was found that the trash is decomposed within a year (Vallis *et al.*, 1996), but it may also take longer (Robertson and Thorburn, 2007a). In the longer term, it may improve soil N levels but that depends on the climate, soil, and management practices (Meier *et al.*, 2006). For example, in South Africa and Australia, it was found that sugarcane yields were higher under trash retention because of better moisture conservation (Thorburn *et al.*, 2005), but soils in trash-harvested systems are more acid (Hartemink, 1998a; Noble *et al.*, 2003). Research has shown that fertilizer N applications should not be reduced in the first 6 years after trash harvesting has started and small reductions (15–40 kg ha$^{-1}$) may be possible after 15 years of trash harvesting but that is site dependent (Robertson and Thorburn, 2007b). Overall, sugarcane trash is N source of slow availability to the crop (Basanta *et al.*, 2003).

## 6.3. Effects on air and water

Sugarcane is either grown under rainfed conditions with high rainfall or irrigated areas which may enhance leaching of fertilizers. In most sugarcane areas, N applications are high and the recovery of fertilizer N ranges from

20% to 50%. Research in the United States showed that leaching losses were up to 60% of the applied N; in Australia, leaching, runoff, and denitrification caused loss of 60% of the applied N (Vallis *et al.*, 1996). There is concern about the effects of rising $NO_3$ levels in groundwater resulting from intensive cropping in relation to environmentally sensitive areas. Gaseous losses are also important and there are indications that sugarcane may loose some of its N through the aerial parts of the plant. Nitrogen losses from denitrification and ammoniacal losses from N applied on trash contribute to greenhouse gas emissions. A quantitative link between gaseous and leaching losses of inorganic fertilizers and the wider environment has not been clearly established for sugarcane—this applies to most agricultural crops. Increasing public and political concern and refined measurement techniques might lead to new regulatory measures to minimize nutrient losses to the environment. This also applies to herbicide and pesticide use. Further development of integrated pest management practices that minimize the use of pesticides is needed to reduce the environmental impact of sugarcane cultivation (Joshi and Viraktamath, 2004; Reay-Jones *et al.*, 2005; Robertson *et al.*, 1995).

Sugarcane cultivation affects the balance of $CO_2$ and other greenhouse gas emissions. Methane is emitted with preharvest burning, when stillage is applied as a soil conditioner, and when fertilizer and bagasse is burned. Also $NO_x$ is emitted from the soil. Growing sugarcane fixes atmospheric $CO_2$ by photosynthesis but there are emissions from the combustion of fossil fuel for field operations, transport, agrochemical production, irrigation, as well the processing plants. The carbon benefit comes from substituting gasoline by ethanol bagasse for fossil fuels in the processing plants (Macedo, 1998). The net contribution of the sugarcane-bioethanol industry to atmospheric $CO_2$ has not been assessed.

Largely unquantified, and ignored in the $CO_2$ footprint discussion, is the net changes in soil organic C. This chapter has shown that soil organic C declines under sugarcane cultivation. The decline is different for different soil types and much depends on the original C level and the period of cultivation. Some soils under sugarcane have released very large amounts of $CO_2$ when cultivated. In the Everglades of Florida (United States), some 167,000 ha of mainly peat (Histosols) is under sugarcane (Muchovej *et al.*, 2000). More than half of the wetlands have been drained (Schrope, 2001) and since 1900, many areas have lost 2.7–4.0 m of surface elevation due to subsidence upon drainage. Clearly, the drainage of such peat lands has emitted very large amounts of $CO_2$ that will not be sequestered by growing sugarcane. In most mineral soils, organic C levels reach equilibrium after many years of sugarcane cultivation, and once soil organic C levels have stabilized, sugarcane cultivation can be seen as a net atmospheric $CO_2$ fixer.

The effects of sugarcane cultivation on the atmosphere include smoke from preharvest burning and processing factories. The smoke affects the health of people living in the surrounding areas. Trash harvesting is the obvious solution but under many conditions (steep slopes, terraces), and for many small-scale farmers, mechanized harvesting is not feasible and pre-harvest burning is the only option. An indirect effect of the sugarcane for bioethanol program is that the air quality in Brazilian cities has much improved since use of ethanol for cars (Arbex *et al.*, 2007), as almost one-fifth of all cars in Brazil run on ethanol. Sugarcane ethanol is a relatively clean fuel as it contains no sulfur oxides, solid microparticles of carbon, benzene, and lead (Pessoa *et al.*, 2005).

## 6.4. Sugarcane yields

Few of the many reports in the scientific literature on the effects of continuous sugarcane cultivation on the soil and the environment quantify the effects of changes on sugarcane yields. This is, perhaps, not surprising as such relations are hard to establish, or may not be directly measurable—they occur gradually and there is considerable within-field variation of both yield (e.g., Johnson and Richard, 2005a,b) and soil properties (e.g., Tominaga *et al.*, 2002) that may mask the effects of soil changes. Sugarcane yields are indeed highly variable and range from 36 to 134 g $ha^{-1}$ in Louisiana, from 65 to 150 Mg $ha^{-1}$ in Australia, and from 70 to 200 Mg $ha^{-1}$ in Brazil. Many factors other than trends in soil chemical properties may explain yield patterns. One of the factors that have received some attention in the literature is the relation between imbalanced plant nutrition and pests and diseases. Recent work in Louisiana, based on earlier studies in Florida, has shown that sugarcane rust is related to excess P and S levels and soil acidity.

A simple way of starting to investigate the relationships between a yield pattern and the trends in soil chemical fertility is to present yield data and soil data from differently producing sugarcane fields. Muchovej *et al.* (2000) measured soil chemical properties in ‘‘good’’ and ‘‘poor’’ spots in Florida. In the study area, sugarcane often exhibits areas of reduced growth that can comprise up to 25% of a field. Dominant soils were Spodosols. Soil pH, organic C, and exchangeable cations were significantly higher for the areas of good sugarcane growth (Table 13). Nutrients, organic C, and microbial populations were less with increasing depth. Although moisture appeared to be an important factor in the areas of reduced growth, a lower or higher water table was not associated with low-yielding areas in the field. Differences in soil chemical properties may be an important explanation for the differences in sugarcane growth.

At a sugarcane plantation in Papua New Guinea, annual sugarcane yields have ranged from 28 to 88 Mg $ha^{-1}$ over 15 years (Hartemink and Kuniata, 1996). This wide variation was explained by sudden and catastrophic infestation of pests and diseases; to a lesser extent, yields were also affected by weed competition. Changes in soil properties under continuous cultivation

**Table 13** Soil chemical properties from Spodosols under "good" and "poor" sugarcane growth at two sites in Florida, United States

| Sampling depth (m) | Soil chemical property | Site I | | Site II | |
|---|---|---|---|---|---|
| | | Good | Poor | Good | Poor |
| 0–0.15 | pH | 6.4 | 5.8 | 6.8 | 5.9 |
| | Organic C (g $kg^{-1}$) | 7.3 | 4.6 | 5.8 | 4.5 |
| | Available P (mg $kg^{-1}$) | 200 | 166 | 38 | 23 |
| | Exchangeable Ca ($mmol_c$ $kg^{-1}$) | 162.7 | 90.7 | 61.9 | 23.5 |
| | Exchangeable Mg ($mmol_c$ $kg^{-1}$) | 10.2 | 10.2 | 1.8 | 1.6 |
| | Exchangeable K ($mmol_c$ $kg^{-1}$) | 3.1 | 2.0 | 2.0 | 1.4 |
| 0.15–0.30 | pH | 6.7 | 6.3 | 6.6 | 6.3 |
| | Organic C (g $kg^{-1}$) | 5.3 | 2.9 | 3.5 | 2.7 |
| | Available P (mg $kg^{-1}$) | 55 | 54 | 40.0 | 27.9 |
| | Exchangeable Ca ($mmol_c$ $kg^{-1}$) | 35.9 | 21.9 | 30.8 | 11.9 |
| | Exchangeable Mg ($mmol_c$ $kg^{-1}$) | 3.2 | 3.4 | 1.5 | 0.5 |
| | Exchangeable K ($mmol_c$ $kg^{-1}$) | 1.5 | 1.0 | 2.3 | 0.8 |

Modified from Muchovej *et al.* (2000).

included decreases in pH, available P and exchangeable K, and soil compaction (Hartemink, 1998c). Leaf nutrient concentrations of N, P, and K also declined (Hartemink, 1998b). It was concluded that the yields were largely influenced by insect pests and diseases but that the management of soil fertility is increasingly important once those problems have been solved.

In Australia, there is widespread evidence that sugarcane yield has been declining. Higher yields are usually obtained on soils that have not been cultivated before (Lawes *et al.*, 2000; Wood, 1985). The yield decline is thought to be caused by a combination of enhanced soil-borne pests and diseases (McGarry *et al.*, 1996a), the frequent tillage, and the use of heavy machinery (McGarry *et al.*, 1996b; Pankhurst *et al.*, 2005a). Soil management practices such as excessive cultivation, insufficient fallowing, the burning of crop residues, and the application of large amounts of N fertilizers are believed to be partially responsible for the decline in sugarcane yield in Australia (Wood, 1985). Pankhurst *et al.* (2005b) investigated the effects of fallow periods and different fallows on sugarcane yield and soil biological properties (Table 14). A fallow period and fumigation resulted in significant higher yields compared with continuously cropped fields, although the effect differed between sites and soil types. The increase in yield was most likely due to reduced populations of soil organisms (e.g., lesion nematodes) that cause yield decline in sugarcane (Pankhurst *et al.*, 2005b).

The Australian studies had considerable impact on the way sugarcane was cultivated and several practices evolved to improve soil: less tillage is being practiced and preharvest burning has been replaced by trash

**Table 14** Effect of fallow and fumigation on plant crop sugarcane yield (Mg $ha^{-1}$) at five sites in Australia

| | Tully | Ingham | Burdekin | Mackay | Bundaberg |
|---|---|---|---|---|---|
| Soil type | Alfisols, Ultisols | Alfisols | Inceptisol | Alfisol | Alfisol |
| Continuous sugarcane | 44 | 38 | 118 | 60 | 112 |
| Continuous sugarcane fumigated | 83 | 83 | 152 | 101 | 143 |
| Pasture (grass/legume) | 73 | 76 | 153 | 104 | |
| Pasture (grass) | | | | | 121 |
| Pasture (legume) | | | | | 116 |

Modified from Pankhurst *et al.* (2005b) based on the work of A. Garside.

harvesting. Preharvest burning is only possible in dry weather (Wood, 1991) and it may cause about 30% of the annual N removal in a sugar crop (Valdivia, 1982). There may be considerable K losses as ash being blown off the plots following burning (Graham *et al.*, 2002a). In South Africa, loss of soil organic matter under sugarcane as a result of preharvest burning is assumed to be a major contributor to soil degradation and may have yield effects (Graham and Haynes, 2006). In Brazil, it was found that over a 16-year period, trash harvesting increased sugarcane yield by 25% (de Resende *et al.*, 2006); in India, trash harvesting improved crop yields from 49 to 73 Mg $ha^{-1}$ (Srivastava, 2003); and in the Philippines, yields are increased by trash harvesting (Mulkins, 2000). Fallow periods after the plowing-out of the sugarcane is common in some areas and may give yield increases: in Alfisols in Swaziland, sugarcane increased from 129 to 140 Mg $ha^{-1}$ after a fallow period (Nixon and Simmonds, 2004), but it is not known whether the increase is the effect of improvement in soil physical or biological soil attributes, or a decline in pests and diseases, or a combination of factors.

The relation between soil changes and sugarcane yield is not always clear. In Mexico, a decline in soil fertility after 30 years of sugarcane cultivation was accompanied by an increase in sugarcane yields. This was explained by improved crop husbandry, although no details are given (de la F *et al.*, 2006). In Bangladesh, sugarcane yields have declined from about 43 Mg $ha^{-1}$ in the early 1970s to 39 Mg $ha^{-1}$ in the 1990s. Based on farmers' interviews, this decline is perceived to be caused by organic matter depletion and a general decline in soil fertility (Hossain, 2001).

This chapter has shown that soils are much changed under continuous sugarcane cultivation but the effects of these changes on yield are hard to quantify. In many parts of the world, yields have increased whereas in many fields, the soils had adversely changed (i.e., lower pH, loss of soil organic matter, increased bulk density). These yield increases are attributed to better crop husbandry, new cultivars, and higher rates of external inputs, particularly N fertilizer. Sugarcane growers are concerned about soil changes. Management techniques have been adopted to improve soil conditions or reduce the negative effects of continuous cultivation including trash harvesting and zero tillage. Improved soil management strategies should be targeted toward the genetic potential of the sugarcane.

## 6.5. The potential for precision farming

Most environmental impact studies in sugarcane areas have been conducted in the United States and Australia, where the crop is cultivated with high levels of inputs (herbicides, pesticides, inorganic fertilizers etc). The heavy use of agrochemicals is a concern but these inputs also guarantee high yields. Precision agriculture has great potential in sugarcane monocropping systems; it may result in increased yield, savings in fertilizers and biocides, and reduced potential for off-farm environmental damage (Wood *et al*., 1997). Sugarcane is an ideal crop for precision agriculture as it is capital-intensive, grown on a large scale, and monocropped for several years. Various studies have investigated the possibilities, mainly in Brazil (e.g., Cora *et al*., 2004; Galvao *et al*., 2005; Magalhaes and Cerri, 2007; Sparovek and Schnug, 2001a), United States (Johnson and Richard, 2005b), and Australia (Bramley and Quabba, 2002; Everingham *et al*., 2007) but also in Mauritius, India, and South Africa. However, the economic and ecological benefits of precision agriculture that have so widely been advocated and used in other cropping systems (Cox, 2002; Pierce and Nowak, 1999; Robert, 2002; Swaminathan, 2006) have not been fully exploited in sugarcane. Given the rapid expansion of the crop in many parts of the world, such technologies are needed for maintaining high yields and sustaining a healthy environment

Several growth models have been developed for sugarcane—particularly in Australia (e.g., Cheeroo-Nayamuth *et al*., 2000; Stewart *et al*., 2006; Thorburn *et al*., 2005; Wood *et al*., 1996): SUCROWS, AUSCANE, CANEGRO, and APSIM-Sugarcane. The two main models are APSIM-Sugarcane and CANEGRO and are based on the CERES maize model (Thorburn *et al*., 2005). These models have increased the understanding of sugarcane physiology and served to identify knowledge gaps and research areas. They have also influenced sugarcane farming systems and policy (Lisson *et al*., 2005). Models can be coupled to studies on soil variation and yield mapping (Johnson and Richard, 2005a; Timm *et al*., 2003) and be combined with newly developed yield-monitoring equipment (Magalhaes and Cerri, 2007).

Mechanized precision farming technologies are beyond the financial and technical capacities of smallholders. However, many smallholders have high skills that can match and substitute high-technology principles used in capital-intensive precision farming. Fertilizers may be too expensive or unavailable but biofertilizers that combine mineral rock phosphate, organic amendments, and soluble fertilizer (Stamford *et al.*, 2006) could be an appropriate nutrient management strategy.

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