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Adequate urban soil occupation planning to face food crisis in the Sahel

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Abstract

Sahel countries are facing at present a threefold crisis, in particular a food crisis with a very weak economy. Non-occupied land areas should be considered as important resources that could be a solution to the problem. Throughout the world, productive land areas have been threatened by urbanization, which has progressively lost potential agricultural land. Sahel's coastal countries are facing this phenomenon with the pressure of increasing population migrations caused mainly by economic issues and climate change impacts. This paper is a case study of two coastal cities from Senegal (Dakar and Saint Louis). It aims to emphasize how well oriented soil occupation should be considered as a solution for enabling populations to overcome the changing climate and economic difficulties, while building a city. Location and evolution of cities are represented using Geographical Information System tools. With appropriate soil quality classification, questionnaires dealing with productivity of ongoing socioeconomic activities provided a convincing argument to support a pedoeconomic approach that concludes on how soil occupation planning could enable people to face efficiently the effects of the food crisis in the Sahel.

Key Words

Food crisis, Sahel, changing climate's effects, soil occupation, pedoeconomic approach, Geographical Information System.

Introduction

Located south of the Sahara desert, the Sahel is a wide semi-arid zone that has suffered tremendous crises that affect Sahelian wellbeing. Food security is an endemic crisis that goes along with periodic drought and flood. The ongoing discussion on global climate change emphasizes the implications of environmental policies and soil occupation, in the Sahel. The Sahel region is part of the world where all economic indicators, rank the countries belonging to this geographic area, as under the minimum human needs to survive (IDH, UNDP 1990/2009). The persistent food crisis that has been revealed to international audiences with the early 1970's drought became a key point of global political and economical cooperation within the region and of increasing interest for researchers. Climate change defined as "long term alteration of global weather patterns, especially increases in temperature and storm activities, regarded as potential greenhouse effects" or "statistical distribution of weather over period of time that range from decades to millions of years" did not cover Sahel rainfall variations from year to year (Ex Saint Louis registered 171 mm in 1986, 340 in 1987, 164mm in 1997, 446 mm on 2000 and 130 mm on 2004). However changing climate is increasingly taken into account by research done on crisis source analyses and their effects in the region. Economic and environmental policies are starting now to be built from results of this research in order to face a decreasing economy and environmental disaster. Floods covering some Sahel countries, from 2005 to 2009 and affecting cities, brought multidisciplinary studies on city building in the region. Urbanization is a global issue. The UN states that more than three quarter of earth's population will live in cities by 2015. Sustainability will depend on how cities are built and how to provide food to the growing population. Soil occupation, particularly when other resources are not available, is one of the solutions to the problem and must integrate ongoing global and local changes. Coastal cities, through their attraction constitute a special case study. The purpose of this paper is to propose a new approach of building cities in a way to overcome the changing climate's effects and economic difficulties. It is based on change observed in Dakar and Saint Louis cities starting in the early 1970's before drought periods and the recent 2000's with recurrent floods. The main objective is to demonstrate how pedoeconomic analyses could positively address climatic and economic problem in Sahelian coastal cities.

Materials

Aerial photos covering the territory taken in 1970 and 1990 at a scale of 1/50 000, have been interpreted for soil mapping and estimation of soil occupation. For soil mapping purpose, scenes of Landsat Thematic Mapper (TM) and Landsat Enhanced Thematic Mapper Plus (ETM+) geometrically corrected at 30 by 30 m resolution have been interpreted for detection of change in soil cover. Geographical Information System

(GIS) tools are used for gathering spatial information from different sources. Soils were described using a regional soil map 1/50 000 and geological map 1/200 000. Soil characteristics were defined using representative profiles. For present soil cover and occupation, Google Earth professional images of the located areas were extracted and read. Rainfall data were extracted from FAO-clim and an increasing number of implemented stations during the last four decades. Gaps in food security as estimated by CILSS, have been extracted from countries annual reports. Decades time series data of agricultural production from city areas were collected from the ground. ENVI software has been used for imagery treatment and ARC GIS for soil types and occupation mapping units.

Methods

Systematic radiometric and geometric correction using standardized methods have been used to connect images and to precisely locate growing cities. All images were corrected for sensor differences and normalized for differences by recalculating pixels into at satellite reflectance (Markham Beker 1986). Regarding interpretation of changes of fast growing cities areas and population, results on NDVI of the Sahel (Jönsson an Eklundh 2004) and precisely of Senegal (Bai and Dent 2007) have been used. Visual colored composed imagery analyses have been done to delimitate built areas. The calculation of NDVI for several periods allowed establishing the vegetation cover progress. Physical characterization of city changes comes from terrain verification by observation and questionnaires. Geographic Information System (GIS) technologies were used with geo-statistic methods and demographic enquiries result analyses to identify the size of cities. With GIS, the work progresses through four steps: i) Scanning to obtain raster data; ii) geo-referencing to adapt map coordinate universal Transverse Mercator (UTM) World Geodesic System 84 datum (WGS); iii) digitalizing raster for vector data; iv) interpreting to identify main soil occupations.

Results

Site location and soils

The specific geographic locations of the areas are shown on Figures 1 and 2 which contain soils types. Comparing soil quality in the global Sahel area and the Senegalese territory in particular, one can notice a relative natural richness due to relatively rich parent materials being volcanic rocks for Dakar city and alluvial silt and clay for Saint Louis. Good amounts of 2/1 clay occur.

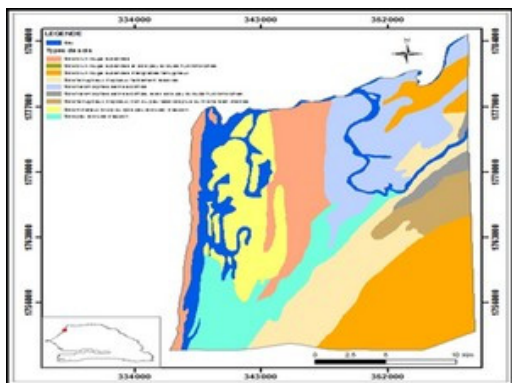


Figure 1. Saint-Louis soil map.

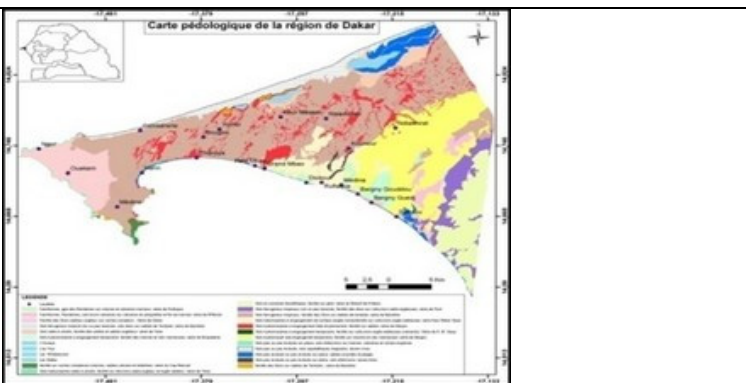


Figure 2. Dakar city soil map.

City building and population distribution

West African countries urbanization, from 1950 to 2020 are hereby described, including the Senegalese ones. In 2000 West African countries had about 1000 cities compared to less than 125, in 1950. Such progression rhythm will bring the number of agglomerations to more than 1430 by 2020. Unbalance between the first agglomeration that generally correspond to capital cities, and other cities, was growing from 1950 to year 2000 (Figures 3 and 4). Unbalance between secondary cities and capitals that characterize present dynamic processes of urbanization will not be in favor of sustainable development; as resources are not equitably distributed across the country. City growing processes induce competition between land functions, as land is needed for buildings, vegetation cover, recreation, water bodies and agricultural production. This competition will reduce cities food self sufficiency. Cities land should preserve best agricultural land. For the studied cases, the urbanization has progressed without taking into account value of natural resource in particular soils and water. Thus, good agricultural land is the first occupied by buildings, while water and food to feed growing populations are brought from more and more longer distances. The particular pedoclimatic zone called Niayes that is represented by a medium large band of land along the northern

Atlantic coast from Dakar to Saint Louis, which represents less than 4% of cultivated land is providing more than 40% of nationwide production.

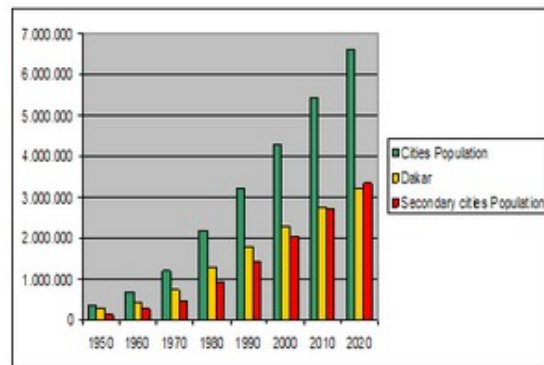
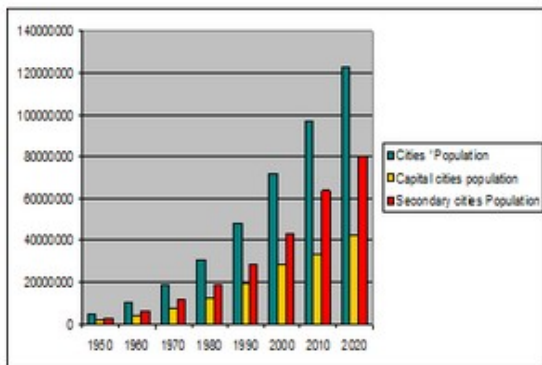


Figure 3. west African statistical population distribution **Figure 4. Senegal statistical population distribution.**

Changes and Soil occupation

Global climate changes appear mostly through rainfall variation rather than warming temperature or sea level increase. This variation was noticed in the early 1970's drought, years before the worldwide concern on climate changes and is nowadays seen through recurrent floods starting in year 2000, and extending to all western Sahel in year 2009. A diachronic study of Dakar soil occupation, through imageries and photos interpretations and field survey based on knowledge of the site, lets one distinguish five major modes of soil occupation of the peninsula with increasing artificial or built areas (grey part) against reduction of agricultural zone, natural vegetation, water bodies and naked land (Fall *et al.* 2009). Dakar city is the most fascinating examples in the Sahel region, but the phenomenon is widespread all along the coastal region of the Sahel.

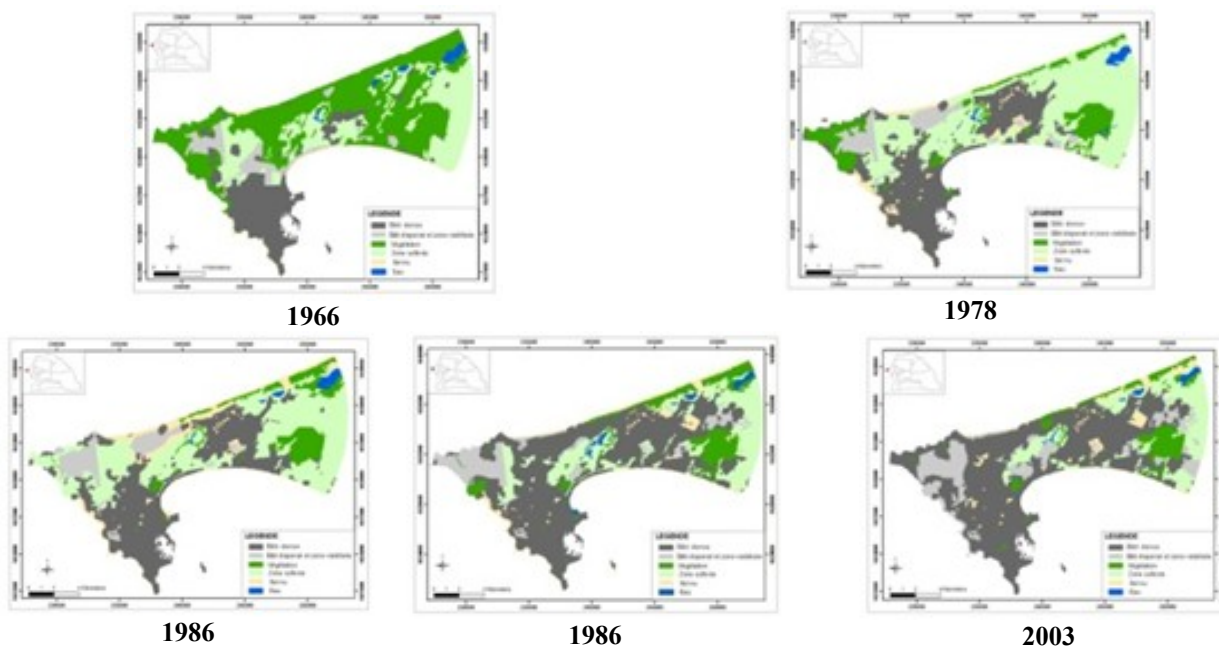


Figure 5. Dakar city diachronic dynamic after Fall and al (2009).

Flooding and food production

Human development needs to explore all land services while building a city. With return to the present rainfall pattern progressively since 2000, all expertise has been concerned by the phenomenon. In particular soils scientist have proposed a new vision on building cities through better consideration of soil quality in relation to land functions and occupation (Fall 2007). For this paper, Dakar and Saint Louis city lands classification have been extracted from a nationwide global soil suitability evaluation (Fall 2009). Based on this soil quality distribution, land building should be directed to land non suitable for agriculture, while suitable lands is dedicated to food production. This does not mean that Dakar and Saint Louis should be rebuilt but taking flooding issue as an opportunity; some part of cities could be reallocated to other occupations.

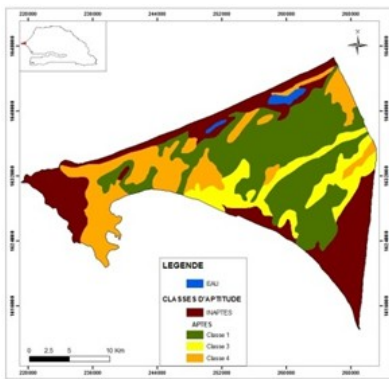


Figure 6. Dakar soil class aptitude (after Fall 2009).

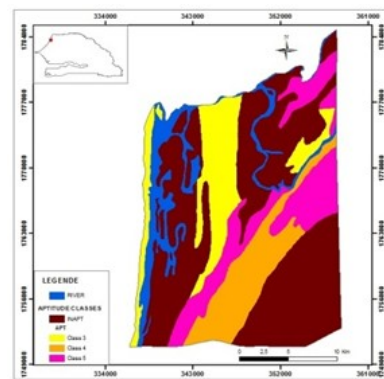


Figure 7. Saint Louis soil class aptitude (after Fall 2009).

Conclusion

All citizens are required to contribute urban plan extension and management in order to guaranty consideration of all sources and effects of global changing. This study open new field of research for city building in the Sahel such as: methods of land evaluation and classification, modeling land spatial occupation evolution, integrating densification of data collection such as meteorological stations, flood occurrence and its effects on agricultural production.

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Assessing macroscopic salinity models for predicting canola response to salinity under bud stage

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Abstract

Plant response to salinity varies during growth stages. Canola is more sensitive to salinity at earlier growth stages but becomes resistant at germination stage. Earlier growth stages including bud stage are very important parts of plant life, because their survival in these stages will determine their final yield. Few macroscopic models have been proposed to quantify the plant response to average root zone salinity during the whole growth period. Since plant response to salinity varies during different growth stages, developing appropriate models for quantitative characterization of plant response to salinity at each growth stage seems to be crucial. To determine the effect of salinity on canola at bud growth stage, an extensive experiment was conducted with a natural saline loamy sand soil, using some salinity treatments including one non-saline water (tap water) and 8 natural saline waters of 3 to 17 dS/m. The Maas and Hoffman (1977), van Genuchten and Hoffman (1984), Dirksen *et al.* (1993), and Homae *et al.* (2002) function were used as macroscopic models to predict relative transpiration (T_a/T_p). To compare the models and their efficiency, some statistics were used. The results showed that the calculated statistical parameters were same for Homae *et al.* (2002) and Dirksen *et al.* (1993) model, but since input parameters for Homae *et al.* (2002) model are easier to obtain, it is recommended to be used for simulating canola response to salinity at bud growth stage.

Key Words

Canola, salinity, relative transpiration, bud stage.

Introduction

The adverse effects of salinity are generally most pronounced in arid and semi-arid regions because of insufficient annual rainfall to flush out accumulated salts from the root zone (Bresler *et al.* 1982). Plants in response to salinity, represents various resistances with respect to its phonologic stages. Most plants are resistant at germination stage. However, at seedling or earlier growth stages, plants usually become more sensitive to salinity but their tolerance increases with age. Salt tolerance of various plants has been extensively studied by different researchers (e.g. mer *et al.* 2000. Keshta *et al.* 1999 and Francois 1994) with the main conclusion that plant response to salinity highly depends on their phonologic stages. Plant water consumption is low at primary growth stage but increases with plant growth and reaches its maximum at bud and rosette stages. Measurement of transpiration or evapotranspiration is necessary for determining plant water demand. Several researchers (e.g. deWit 1958; Hanks 1984) showed a linear relationship between plant growth and transpiration or evapotranspiration rate (Homae and Schmidhalter 2008). Water absorption by plants decreases with increasing salinity. Water movement in unsaturated soils is described with the Richard's equation (Richards 1931). Including the root extraction term S , it reads:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z} \left[K(h) \frac{\partial h}{\partial z} + K(h) \right] - S \quad (1)$$

where θ is volumetric water content (L^3/L^3), t is the time (T), h is the soil water pressure head (L), z is gravitational head, as well as the vertical coordinate (L) taken positive upward, k is the soil hydraulic conductivity (L/T), and S is the water extraction rate by plant roots ($L^3/L^3/T$). Feddes *et al.* (1978) introduced a macroscopic sink term depending on soil water pressure head h only as:

$$S = \alpha(h) S_{\max} \quad (2)$$

where S_{\max} is the maximum rate of water uptake and $\alpha(h)$ is a dimensionless function of pressure head. Analogously, one may introduce a soil salinity reduction term, $\alpha(h_o)$, instead of $\alpha(h)$ in Eq. (2). This salinity function can be put in the form of the Maas and Hoffman (1977) equation. Written in terms of the soil solution osmotic head h_o , this gives (Homae *et al.* 2002):

$$\alpha(h_o) = 1 - \frac{a}{360} (h_o^* - h_o) \quad (3)$$

where h_o^* is the osmotic threshold value and 360 is a factor to convert the salinity-based slope to centimetres osmotic head. Since the linear assumption in Eq. (3) does not fully meet the real field conditions, van Genuchten and Hoffman (1984) proposed:

$$\alpha(h_o) = \frac{1}{1 + (h_o/h_{o50})^p} \quad (4)$$

Where h_{o50} is the soil salinity at which $\alpha(h_o)$ is reduced by 50%, and p is an empirical parameter. Dirksen *et al.* (1993) proposed the following as modification for Eq. (4):

$$\alpha(h_o) = \frac{1}{1 + ((h_o^* - h_o)/(h_o^* - h_{o50}))^p} \quad (5)$$

The most important limitation for both Eqs. (4) and (5) arises from the difficulty involved in obtaining h_{o50} (Homaee and Schmidhalter, 2008).

Homaee *et al.* (2002) proposed:

$$\alpha(h) = \frac{1}{1 + (1 - \alpha_o) / \alpha_o [(h_o^* - h_o)/(h_o^* - h_{o\max})]^p} \quad (6)$$

The reduction in α due to salinity beyond h_o^* continues significantly until a certain degree of salinity ($h_{o\max}$) is reached; beyond $h_{o\max}$, the salinity increase do not cause significant further reduction in α . The exponent p is further defined as (Homaee *et al.* 2002):

$$p = \frac{h_{o\max}}{h_{o\max} - h_o^*} \quad (7)$$

Methods

In this study, natural saline water with electrical conductivity of 600 dS/m was first provided from Hoze Soltan Lake, Qom, Iran. Table 1 shows some chemical properties of lake water. Considering the experimental salinity treatments, the lake water was diluted by adding proportional amount of tap water to gain the desired salinities. The experimental salinity treatments were tap water (control treatment), 3, 5, 7, 9, 11, 13, 15, and 17 dS/m with three replicates. A natural saline soil (4 dS/m) with loamy sand texture was delivered to greenhouse; air dried and sieved through 5mm sieves. Some experimental pots were carefully packed with this saline soil at bulk density of 1.35gr/cm³. Each pot was then irrigated with relevant natural saline water until the desired salinity was reached. The main reason to select loamy sand texture was to apply maximum leaching fraction in order to obtain the designated soil salinities. Then, the leaching fraction (LF) of 0.5 was applied to all experimental soils. To insure the accuracy of applied leaching, the electrical conductivity of drainage water was continuously monitored. To avoid any salinity stress, canola plants were irrigated with tap water before reaching bud stage and the salinity treatments were applied afterward. To minimize water evaporation from soil surface, a 3-Cm sand layer was laid on the top of each pot. Three canola plants were maintained in each pot and the daily transpiration was calculated by weighting the pots at regular times sequence. The Maas and Hoffman (1977), van Genuchten and Hofman (1984), Direksen *et al.* (1993) and Homaee *et al.* (2002) models were used to predict relative transpiration in different salinity levels. The p and o are denoted for predicted and observed data. The Maximum Error (ME), Root Mean Square Error (RMSE), and Modelling Efficiency (EF) statistics were computed to compare the models.

Table 1. Some chemical properties of the lake water.

| (NO ₃ ⁻) | (Na ⁺) | (Mg ²⁺) | (Ca ²⁺) | (B) | (SO ₄ ²⁻) | (CL ⁻) | (HCO ₃ ⁻) | (CO ₃ ²⁻) | (EC) | pH |
|---------------------------------|--------------------|---------------------|---------------------|------|----------------------------------|--------------------|----------------------------------|----------------------------------|------|------|
| mg/L | g/L | g/L | g/L | mg/L | mg/L | g/L | g/L | mg/L | dS/m | |
| 2.75 | 115 | 22.4 | 1.2 | 54.8 | 341.5 | 161 | 8.6 | 0.0 | 600 | 7.25 |

Results

Figure 1 shows the measured and predicted relative transpiration (T_a/T_p) of canola at bud stage by different models. Based on the data reported by Maas and Hoffman (1977), the threshold value for canola is 11 dS/m for the whole growth period. However, the results given in fig. 1 indicate that salinity threshold value for canola (ECm) is 4 dS/m at bud stage. The estimated parameters of all four used models and the related statistics are given in Table 2. As can be seen in Fig.1, the nonlinear models including van Genuchten and Hoffman (1984) Direksen *et al.* (1993) and Homaee *et al.* (2002) models provided better estimation than the linear response model of Maas and Hoffman (1977). Based on Table 2, in spite of having identical R² value for all nonlinear models, a model with less ME and RMSE values and definable parameters is preferred.

Accordingly the Homae *et al.* (2002) model provided lower ME and RMSE values (Table 2). Taking the advantage of its accessible parameters, this model provided the most reliable estimation at bud phonologic stage.

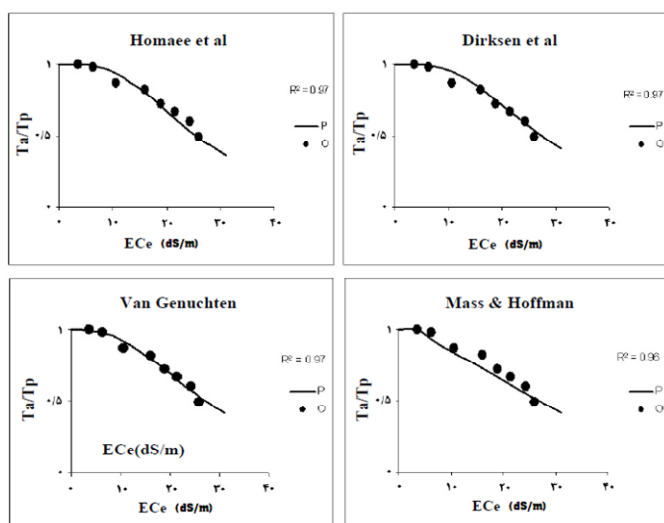


Figure 1. Simulation canola plant response to salinity at bud growth stage.

Table 2. The estimated parameters and calculated statistics for different salinity response models.

| Models | EC _p | EC _o | EC ₅₀ | EC _{max} | <i>b</i> | α | ME | EF | RMSE | R ² |
|--------|------------------|-----------------|------------------|-------------------|----------|----------|-------|------|------|----------------|
| | (-----dS/m-----) | | | | - | - | - | - | - | - |
| 4 | 3 | 4 | - | 18.59 | - | 0.72 | 0.05 | 0.98 | 3.4 | 0.97 |
| 3 | 3 | 4 | 28.1 | - | - | - | 0.05 | 0.98 | 3.4 | 0.97 |
| 2 | - | 4 | 27.3 | - | - | - | 0.046 | 0.98 | 3.4 | 0.97 |
| 1 | 5.2 | 4 | - | - | 0.021 | - | 0.06 | 0.97 | 3.98 | 0.96 |

Conclusion

Unlike data reported by several investigators (e.g. Maas and Hoffman, 1977) the canola threshold value for the whole growth period, depends on growth stage. This threshold value for canola was 4 dS/m at bud growth stage. Simulating canola relative transpiration by four different response models indicated that the nonlinear models provide better estimation than the linear model for this growth stage. The predicted threshold values (EC_p) obtained by these models were less than those reported by Maas and Hoffman (1977). The statistical analysis showed that Homae *et al.* (2002) model can provide more accurate predictions than the other models. Since the required input parameters for Homae *et al.* model are easier to obtain, it is recommended to be used for evaluating canola response to salinity under bud stage.

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Bio-char from sawdust, maize stover and charcoal: Impact on water holding capacities (WHC) of three soils from Ghana

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Abstract

This paper reports part of an on-going investigation into the effects of bio-char on soil physical properties in Ghana. Bio-char from sawdust (B1) and maize stover (B2) were prepared using a muffle furnace. The effect of local charcoal from Ghana (referred to in this paper as B3) was also studied. These three types of bio-char were applied to three soil types from Ghana, at 5, 10 and 15 t/ha. Results indicated that WHC was increased when bio-char was applied at all rates compared to zero application. However, there wasn't much difference in effect on WHC between the rates. It is suggested that water repellency of the bio-char partly explains this behaviour. Improving WHC by bio-char application was more effective in sandy textured soils.

Key Words

Bio-char, water holding capacity, maize stover, sawdust and charcoal

Introduction

This study aims to further the recent research efforts on the evaluation of bio-char as a soil enhancer (e.g. Yeboah *et al.*, 2009) and as a means to mitigate increasing levels of carbon dioxide in the atmosphere. The carbon sequestration benefit results from the fact that bio-char takes carbon from the atmosphere-biosphere pool and transfers it to a slower cycling form that can exist for hundreds of years (Fowles, 2007). The primary benefit of bio-char in Ghana is its positive effect on agricultural productivity, as most soils are acidic and some have problems of aluminium toxicity, a condition amenable to bio-char application (Lehmann *et al.*, 2003). Low soil organic matter content in soils resulting from high temperatures and rainfall, are responsible for the low available water capacity and weak structure of many agricultural soils (Piccolo *et al.*, 1996). Glaser *et al.* (2002) stated that bio-char added to soil may not only change soil chemical properties but also affect soil physical properties, such as soil water retention and aggregation. Hence, there is a need for researchers to gather evidence on the capability of bio-char to improve soil physical properties, notably soil water retention and availability, soil aggregation and infiltration, thereby sustaining agriculture on already converted forest lands in Ghana.

Methods

Soil sampling

The top soil layer (0-15 cm depth) was sampled from three sites in Ghana, followed by the removal of all plant debris. The soil samples were air-dried and sieved through a 2 mm mesh, prior to physical and chemical analysis of the soils.

Soil textural classification

The soil textural analysis was carried out by the Bouyoucos/hydrometer method (Bouyoucos, 1962). After measuring the sand, silt and clay distributions, the soils were assigned to textural classes (Table 1) with the help of a textural triangle.

Soil bulk density

The dry bulk densities (BD) of the soils were determined on intact soil cores (5 cm diameter), sampled from the 1-15cm depth. The cores were oven-dried at 105°C for two days. The bulk densities were calculated using the formula below:

$$\text{Bulk density} \left(\text{gcm}^{-3} \right) = \frac{(W_2 - W_1)}{V \text{cm}^3} \text{g} \quad (1)$$

where W_2 and W_1 are weights of moist and oven-dry soils, respectively, and V is the volume of the cylindrical core.

Charring maize stover and sawdust

Dry maize stover, collected from farmers' fields in Ghana, as well as sawdust obtained from small woodcuts from a local sawmill in Kumasi, Ghana, were exported to Reading University, UK, where they were subjected to carbonization under anoxic conditions using a muffle furnace, operated at atmospheric pressure (method as used by Braadbaart and Poole, 2008, with slight modifications). The stover samples were placed into the furnace at ambient temperature (~20°C) in pre-weighed rectangular aluminium containers and heated to a pre-selected final temperature (420°C for maize stover, being soft wood, and 450°C for the sawdust). The times of exposure were 70 and 75 minutes, respectively. Sawdust (sieved to particle size <0.5 mm) was placed in a cleaned metal syrup tin. A 1mm hole was drilled in the lid and pressed firmly on the tin. This provided a small opening for steam and gas to escape to avoid explosion as the heating progressed. The weight of the filled containers was measured, before placing it in the furnace, so that the initial weight of maize stover and sawdust used could be calculated. As soon as the set temperature was reached and remained over the given time, the furnace was switched off and allowed to cool for about thirty minutes before transferring the samples into a dessicator for further cooling. The samples were then removed for weighing. B3 was not treated in the UK but collected in Ghana from local charcoal makers.

Determination of water holding capacity of soils and soils treated with bio-char

To test the effect bio-char on the soils' water holding capacity – WHC (also known as field capacity), WHC of the soils and soils treated with bio-char were determined. WHC is the maximum amount of water the freely drained soil can hold, which is estimated after a saturated soil has been allowed to drain without allowing its moisture stores to be depleted by evaporation. To do this, 20 grammes of the air-dried soil sample, in triplicate, were put in a plastic container (with a wire mesh at the bottom) and placed in a dish of water. This was allowed to become saturated, for approximately six hours. The container was removed from the water and covered with cling-film to prevent loss of water by evaporation. It was then hanged on a retort stand overnight to allow drainage. All samples were allowed to drain for the same amount of time. Next, soil was carefully removed from the container, put in a pre-weighed container (M_1) and the total weight of moist soil and moisture container (M_2) was taken. The samples were then dried in an oven at 105°C until no further water loss occurred, and reweighed to record the oven-dried sample (M_3). The WHC was determined from:

$$WHC(\%) = \frac{M_2 - M_3}{M_3 - M_1} \times 100 \quad (2)$$

Results

Table 1 summarises the basic physical properties of the soils under study. The relatively high BD of Soil E is consistent with the observed high sand fraction (sandy soils have higher bulk densities compare to loamy and clayey soils, Phogat *et al.*, 1999), low clay fraction and exceptionally low OC content. This condition is typical for Ferric Lixisols (Oguntunde *et al.*, 2004), which have potentially higher infiltration, decreased aggregation and lower water holding capacities compared to Chromic Lixisols (soil A) and Ferric Acrisols (soil K) [Yeboah *et al.*, 2009]). Soils A and K are loams, sandy loam and silt loam respectively.

Table 1. Characteristics of soils from the study sites.

| | Soil A | Soil K | Soil E |
|-----------------------------------|-----------------|----------------|----------------|
| Textural class | Sandy Loam | Silt Loam | Loamy Sand |
| Sand % | 60.0 | 30.6 | 79.8 |
| Silt % | 36.6 | 53.3 | 18.1 |
| Clay % | 3.4 | 16.1 | 2.1 |
| Classification (WRB) | Chromic Lixisol | Ferric Acrisol | Ferric Lixisol |
| Bulk Density (kgm ⁻³) | 1293.9±32.7 | 1378.9±48.9 | 1448.8±20.23 |

Effect of bio-char on water holding capacity of three soils from Ghana

Figures 1-3 show the effects of applying bio-chars (B1, B2 and B3) at 5, 10 and 15 t/ha to three Ghanaian soil types differing in classification and texture (Table 1).

Figure 1 shows that the maize stover bio-char (B2) increased the WHCs of all three soil types ($P = 0.001$), most significantly for soil E, although WHC at the highest application rate for all soils was lower than that at 5 and 10 t/ha, but still considerably higher than at zero application. The textural class of soil E was earlier described as loamy sand (Table 1), with about 80% sand fraction. Even though soil A and K were also improved in their water retention capabilities (at all rates), the rate of improvement was far greater for soil E. The rate of increase in WHC for soil E ranged from 349-481%, compared to 36-56% for soil A and 27-41% for soil K, clearly confirming observations by Tryon (1948) that moisture retention increased particularly when charcoal (bio-char) was applied to sandy soil. Furthermore, this is also proof of Downie *et al.*'s (2009) proposition that because small pores in bio-char retain moisture, and because small pores (with a relatively large water holding capacity, as this scales with the pore radius) are largely absent in coarser-textured soils, bio-char should have the greatest effect on water retention in sandy soils. Similar observations can be made for sawdust bio-char (B1) as shown in Figure 2.

Similarly, the WHCs of the three soils were improved by B1 bio-char (Figure 2), and very much so in the coarser textured soil E. However, the rate of increase is slowed after 5 t/ha. For all soils, increasing the application rate from 5 to 10 t/ha, decreases the WHC, whereas increasing the application rate to 15 t/ha brings WHCs up to values that are higher than at any of the other application rates. Even though the statistical report (using Duncan multiple test range) indicated a marked difference ($P = 0.001$) between zero bio-char and other rates, no significant difference was reported between 5, 10 and 15 t/ha, suggesting that the optimum rate of bio-char application to improve soil moisture retention was 5 t/ha. Water repellence might have occurred at higher concentrations of the bio-char. Another possible explanation is that addition of more bio-char negatively affected the soil structure and hence the soil's WHC.

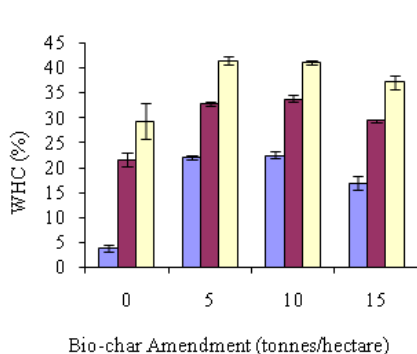


Figure 1. WHC of three soils amended with maize stover bio-char (B2) at three rates.

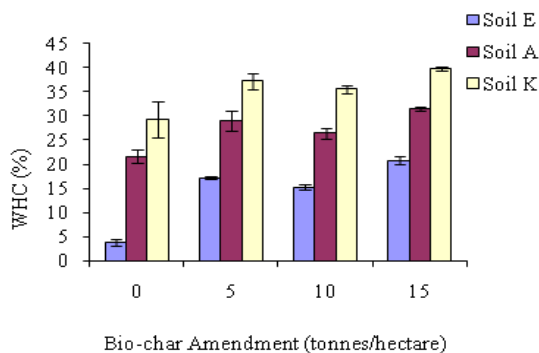


Figure 2. WHC of three soils amended with sawdust bio-char (B1) at three rates.

As evidenced in Figure 3, the WHC of the sandy textured soil E was the least improved by charcoal B3 (36.69% increase at 5 t/ha); after reaching its optimum at 5 t/ha the water retention capabilities of the soil decreased by 31.78% and 0.005% at 10 and 15 t/ha, respectively, as compared to the untreated soil. Many factors may have attributed to this rather unusual observation. Among them is the hydrophobicity due to the hydrophobic polymers [Glaser *et al.*, 2002] present and in such quantities as to cause water repellence after 5 t/ha application rate. These hydrophobic substances might have been redistributed within the soil, much more in sandy soil and at application rates larger than 5 t/ha. Hence, there is an urgent need to investigate and ascertain the type of feedstock used for making B3 in Ghana at the collection site.

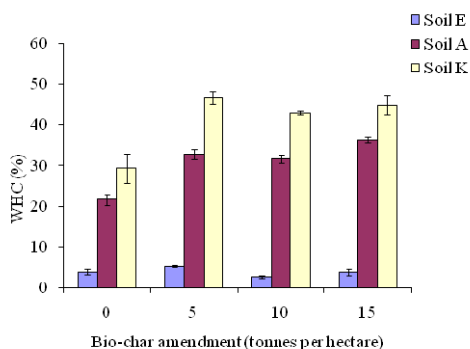


Figure 3. WHC of three soils amended with charcoal (B3) at three rates.

Conclusions

Based on the results, the study could conclude on the following:

WHC was increased when bio-char was applied at 5, 10 and 15 t/ha, compared to zero application of bio-char. However, there was no significant difference between the rates. These results suggest that 5 t/ha is the optimum application rate; Water repellence of the bio-char appears to start playing a role when bio-char application exceeds 5 t/ha; the magnitude of this effect depends on the feedstock characteristics; Improving WHC by bio-char application is more effective in sandy textured soils. It is, however, worthy to note that this may not apply to bio-char from all feedstock. Some feedstock may have hydrophobic substances whose effects may be felt more for a different soil texture, at rates higher or lower than 5 t/ha.

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Changes in the cation composition of a Barossa Chromosol irrigated with wastewaters of contrasting monovalent cation concentrations

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Abstract

Reusing wastewater, including recycled municipal water and winery wastewater, for irrigation has become increasingly commonplace as an efficient use of water and effective means of disposal. One of the major agricultural concerns related to the use of recycled water is the accumulation of exchangeable sodium (Na) in the soil profile and the potential impacts this has on soil structure. Potassium (K), like Na, has a high affinity for clay minerals and therefore has the potential to cause clay swelling and dispersion. Here, we describe the dynamics of cations in two contrasting wastewaters, a Na-rich municipal wastewater and K-rich winery wastewater (Na:K ratio ~ 1:1) following irrigation to a Barossa Chromosol. Accelerated annual irrigation, drying and rainfall cycles enabled long-term predictions to be drawn. With the passage of approximately 1000 pore volumes through the core over the course of 7 months, greater accumulation of K was evident in soils irrigated with winery wastewater irrigation, which is likely to be a result of adsorption to specific binding sites in the illite dominated clay. The high mobility of Na combined with the flow through nature of the columns has prevented accumulation of Na under both wastewater types despite the high initial Na concentration in both water sources. High Magnesium (Mg) concentrations in both wastewaters have led to a greater retention of Mg in favour of Calcium (Ca), conforming to Schofield's Ratio Law.

Key Words

Winery wastewater, municipal wastewater, basic cations.

Introduction

Agricultural land surrounding Adelaide is highly productive, while accounting for only 1.5 % of South Australia's total agricultural land, this region accounts for more than 18 % (A\$800 million) of the state's gross agricultural earnings (ABS 2003). Currently 2340 hectares of vines around Adelaide are irrigated with recycled water, however, interest has been expressed by the viticultural industry to utilise a greater percentage of recycled water from municipal and winery industry sources to overcome water shortages, rising mains water prices and enable the safe and sustainable disposal of wastewaters generated in the winery. Concern however is raised over the unfavourable concentration of salts in these waters and the ratio of monovalent cations (Na and K) to divalent cations (Ca and Mg).

A high concentration of monovalent Na in soils has the potential to disrupt soil structure, leading to changes in many key soil physical properties such as hydraulic conductivity, infiltration rate, bulk density and soil aeration (e.g. Rengasamy and Olsson 1991). Based on the large hydrated ion size and its affinity to clay minerals, high levels of exchangeable K in soil also has the potential to cause clay swelling and dispersion (Levy and Feigenbaum 1996). Research on the soil structural effects of K in irrigation wastewaters has received less attention due to the typically low abundance of K in most waters (Arienzo *et al.* 2008). Winery processing wastewater however tends to have elevated concentrations of both K and Na therefore posing a risk of clay dispersion when irrigated to land. Unlike Na, exchangeable K in soils is adsorbed by specific and non specific binding, the former being dependent on the soil clay mineral. Illite and other mica clay minerals are particularly abundant in specific binding sites for K within their structural layers (Arienzo *et al.* 2008).

In the Barossa Valley, South Australia, Municipal Sewage Wastewater and Winery wastewater are treated independently and distributed to growers for the irrigation of grape vines. Here, we investigate the effect of these two waters on the soil cation properties of a Barossa Chromosol containing illite clay mineral

Methods

Intact columns (150 mm length x 100 mm width) of a duplex red chromosol soil profile were collected from the Viticultural Research Station in Nuriootpa, Barossa Valley, SA. This soil is described as a non restrictive duplex red chromosol with well structured topsoil and was selected due to its good representation of soils widely used for viticulture in the Barossa region and where winery wastewater is used for irrigation. Sixteen columns were used in this experiment and contained soil from 0-150 mm depth in the soil profile. In order to assess the bio-availability fraction of constituents applied in wastewater, ryegrass (Diplex Italian Hybrid) was established in each column. A fibre-glass wick was mounted at the base of each column to maintain a fixed tension and thereby create a 'hanging soil-water column' similar to field conditions.

Winery wastewater (WIN) was sourced from the North Para Environmental Control Treatment Plant (NPEC) in Barossa Valley and municipal wastewater sourced from the Tanunda Community Wastewater Management Scheme (CWMS). Irrigation of wastewater was applied to a depth of 60 mm per day for ten days and then allowed to drain for 4 days before applying a low ionic water solution, akin to rainfall, at a rate of 60 mm per day for 2 days. This schedule represents the volume of water applied to vineyard soils under drip irrigation over a 1 year period, thereby allowing us to apply the equivalent of 1 year's irrigation to soils in the space of 16 days. This cycle was repeated 9 times in continuous fashion, representing 9 years of irrigation. Care was taken to balance the time over which irrigation was applied each day, typically 8 hours, with the period of free draining, 16 hours, to ensure adequate diffusion of soluble ions throughout the soil matrix. For each treatment and replicate, a bulk sample of soil leachate was collected every 5 days of consecutive wastewater irrigation, approximately 8 pore volumes (p.v.), and every 2 days following rainfall irrigation (approximately 3.2 pore volumes). Analysis of basic cations was carried out on filtered (0.45 µm) samples using ICP-OES.

The sodium adsorption ratio (SAR) of the wastewaters is defined as $\text{Na}/(\text{Ca}+\text{Mg})^{0.5}$ where the concentration of Na, Ca and Mg is in mmol_c per litre, while the potassium adsorption ratio (KAR) is defined as $\text{K}/(\text{Ca}+\text{Mg})^{0.5}$

Results

The composition of the two wastewaters, municipal CWMS and WIN, and rain water are shown in Table 1. It is evident that both wastewaters are dominated by monovalent cations. In the case of CWMS sodium is the dominant cation while in the WIN there is near equal concentrations of both sodium and potassium. The electrical conductivity of the two wastewaters is similar.

Table 1. Chemical composition of Municipal CWMS water, NPEC winery wastewater and rainwater.

| Chemical parameter | Municipal CWMS Water | | NPEC Winery Wastewater | | Rainwater | |
|--------------------------------|----------------------|--------|------------------------|--------|-------------|--------|
| pH | 8.49 | ± 0.31 | 8.69 | ± 0.20 | 6.09 | ± 0.20 |
| Electrical Conductivity (dS/m) | 1.66 | ± 0.15 | 1.38 | ± 0.13 | 0.43 | ± 0.03 |
| Ca (mmol _c /L) | 1.05 | ± 0.24 | 1.61 | ± 0.24 | 0.28 | ± 0.11 |
| Mg | 1.50 | ± 0.11 | 0.89 | ± 0.08 | 0.23 | ± 0.07 |
| Na | 13.64 | ± 1.09 | 7.50 | ± 0.61 | 0.79 | ± 0.23 |
| K | 1.19 | ± 0.07 | 3.94 | ± 0.24 | 0.05 | ± 0.02 |
| SAR | | | 7.0 | | 1.72 | |
| KAR | | | 3.8 | | 0.06 | |

Cation balance for soils irrigated with CWMS water and WIN is shown in figure 1. On the basis of cation chemical charge balance, soils irrigated with CWMS displayed a net reduction in cations (i.e. chemical charges lost exceed those gained). This was principally due to the loss of Ca, despite relatively stable pH. In both soils, the loss of Ca occurred rapidly with the introduction of wastewater and changes after passing of the first 8 p.v. (first sampling point) accounted for 31 and 49 % of all Ca lost for CWMS and WW respectively. Such disproportional losses of Ca may reflect the solubilisation of CaCO₃ precipitates that have accumulated in these soils prior to trial commencement. Soils irrigated with winery wastewater showed a net increase in cations which is consistent with an increase in soil pH from 7.38 (initial) to 8.04 (final). Increase in pH noted here, is likely a result of the high bicarbonate content of winery wastewater (300 mg /L) and the effect of this on soil pH.

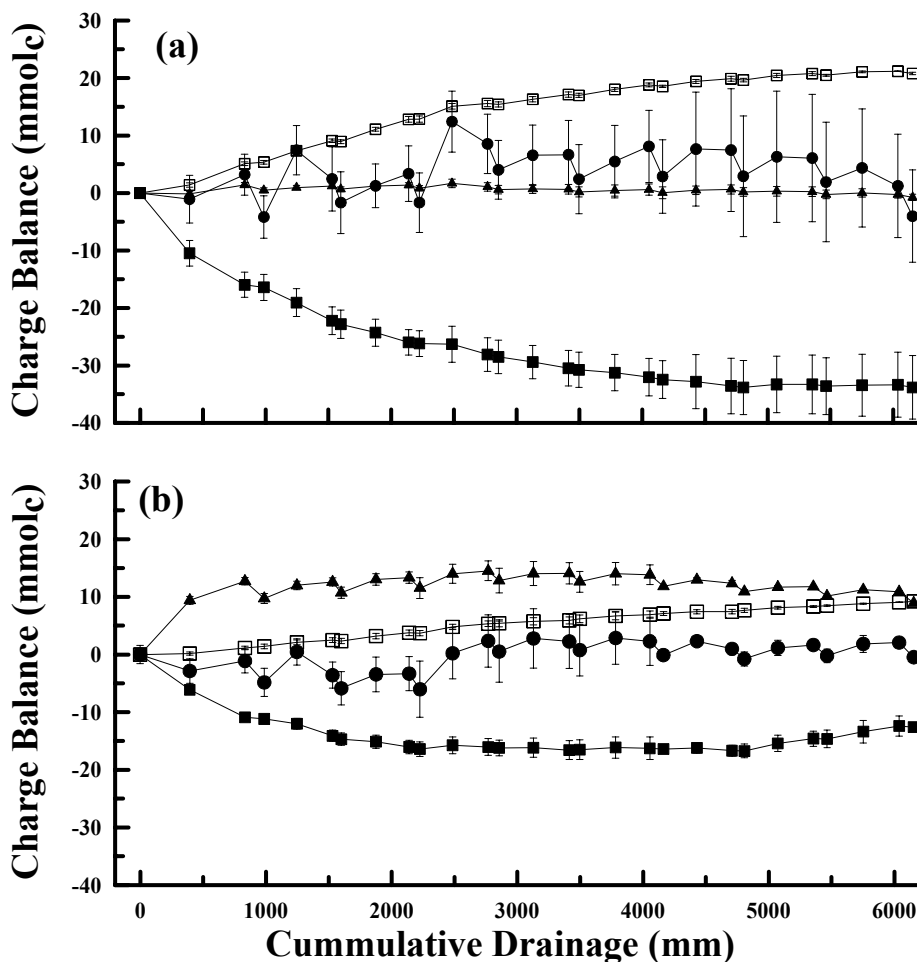


Figure 1. Cation Balance for a Chromosol soil irrigated with (a) CWMS water and (b) winery wastewater. Data is expressed in terms of cumulative additions and losses of Ca (■), Mg (□), Na (●) and K (▲) within the soil profile based on net mmol_c added in water source minus net mmol_c removed by pasture and lost in drainage as a function of the cumulative volume of drainage.

Magnesium is a divalent cation and is likely to have similar binding behaviour to Ca. Although not as effective as Ca in maintaining soil structure, Mg can mitigate the risk of soil dispersion posed by the high abundance of exchangeable monovalent cations. The retention of Mg has increased in soils irrigated with both CWMS and WIN in a similar linear fashion during the course of the experiment. Prior to irrigation, the ratio of soil exchangeable Ca to Mg was 9:1, while in the wastewaters this ratio was 0.68:1 in CWMS and 1.8:1 in WIN. Following irrigation, exchangeable cations undergo adsorption/desorption processes in order to maintain a common ratio between solid and liquid phase. Presumably therefore, Mg will continue to bind in favour of Ca until a Ca:Mg ratio similar to the given wastewater is reached.

In both soils, Na appears to be highly mobile and, given adequate percolation of water, does not appear to accumulate with on-going irrigation, despite the high Na concentration in both wastewaters. Due to the rapid loss of Na following irrigation, it is unlikely that this ion plays a role in the displacement of Ca, rather, the lesser valency, higher solubility and rapid percolation of water enable Na to pass through the column relatively unperturbed. Where evaporation (i.e. upward movement of water) exceeds downward movement (irrigation+rain) Na will have a greater propensity to build-up.

The retention of K in soils is markedly different between irrigation sources and is likely to reflect differences in the initial water concentrations. An accumulation of K is evident in soils irrigated with WIN, this appears to have occurred rapidly following commencement of irrigation. Clay content in the topsoil of this profile is approx. 17%, and is dominated by illite that contains specific binding sites for K within structural layers. With the introduction of K in WIN a considerable amount (10 mmol_c) of K is likely to be retained on specific binding sites in the soil. Following the passage of approximately 16 p.v there is no further retention of K suggesting that specific binding sites are saturated after which time K appears to be readily leached through the column.

In instances where the level of specifically bound potassium in illitic clays is low, such additions of K may in fact improve structural properties of the colloid (Ravina and Markus 1975; Chen *et al.* 1983) and where flow through conditions occur there appears to be no risk of adverse K build up. Pasture irrigated with WIN also showed a corresponding increase in dry matter K concentration (not shown).

Conclusion

Maintaining infiltration of wastewater and rainfall through the soil ensures adequate leaching of monovalent cations that may otherwise accumulate in soils and pose a risk of soil dispersion. It seems evident that although the initial concentration of Na in the CWMS water is high, greater mobility relative to other cations allows it to readily leach through soil. Potassium on the other hand is less mobile in soils and will be retained on both specific and nonspecific binding sites. Following saturation of specific binding sites K appears to be highly mobile. The loss of Ca we suggest is caused by a redistribution of exchangeable and soluble cations to equilibrate the high Mg:Ca ratio in both waters applied.

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Design, construction, and operation of a wetland/reservoir system receiving agricultural drainage water

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Abstract

This study serves as a complete and detailed case study of the entire wetland reservoir system (WRS) design and construction process. Challenges presented by the cold climate and geology of Nova Scotia were identified and addressed during the design, construction, and operation of this WRS so that future systems can be optimized. The main challenge of integrating individual system components into the WRS were grade and land availability. A high water table impacted CTW construction and operation. A detailed field investigation should identify this challenge before construction, but limited land availability may force the CTW to be constructed in a less than ideal area. Installing tile drains around the perimeter of the CTW helped alleviate this challenge. Background hydraulic and water quality data was important for addressing episodic hydraulic and pollutant loading and localizing the design. Treatment is assessed and should help assess the design method, which used monthly data to determine the CTW area. Data from this study may contribute to the development of a design model specific to CTWs receiving drainage water. The WRS continued to operate without difficulty during the winter months. It is importance of establish vegetation cover immediately after construction to mitigate erosion, and to have emergency spillways to remove excess water during storm events.

Key Words

Subsurface drainage, water quality, constructed wetlands.

Introduction

Artificial subsurface or “tile” drainage is used to remove excess water from agricultural lands. Removing excess water lowers soil water content to field capacity, which permits earlier field trafficability and enhances growing conditions, and ultimately results in a more productive crop system. However, agricultural non-point source pollution, including subsurface drainage water, is a major source of surface and ground water degradation. Agricultural subsurface drainage water periodically contains concentrations of nutrients, pathogens, pesticides, and sediment that exceed water quality guidelines. The export of these contaminants can have major ecological, health, and socio-economic consequences. Tile drainage is used extensively throughout Nova Scotia and, in most cases, un-treated effluent is discharged directly into surface waterbodies.

Another water management issue affecting Nova Scotia is water availability during the growing season. Nova Scotia has endured droughts in recent years despite an abundance of groundwater, lakes, and rivers and receiving, depending on the region, annual precipitation of less than 1000 mm to more than 1600 mm (Environment Canada 2007). This is due to the timing of precipitation; there is often a surplus of precipitation during the non-growing season and a deficit during the growing season. Periods of water deficit in Atlantic Canada are projected to become more frequent and severe due to climate change and increased water demands (Nova Scotia Department of Environment and Labor 2005).

Treatment and reuse systems have the potential to address both pollution from agricultural drainage water, and water supply issues. One type of systems captures surface runoff and/or tile drainage water, uses a constructed treatment wetland (CTW) to improve water quality, and stores the treated water in a reservoir. This water can be reused for irrigation, upon which the cycle of drainage, capture, and treatment continues. These systems need to be assessed in a cold climate so that their location, design, construction, and operation can be optimized in Nova Scotia. Specifically, a better understanding of system hydraulics, CTW treatment efficiencies, and pathogen management is required. The purpose of this paper is to describe the design, construction, and a preliminary assessment of a drainage water treatment and reuse system that consists of a tile drainage system, a CTW system, and an irrigation reservoir.

Methods

Site Description

The study site is located at the Bio-Environmental Engineering Center in Truro, Nova Scotia, Canada (N 45E 23' and W 63E 15'). Truro has a daily average temperature of -6.6 °C in January and 18.6 °C in July and receives 1170 mm of annual precipitation, with peaks amounts during the fall (Environment Canada 2007; 2008). The tile drainage system that supplies the wetland-reservoir system (WRS) is underneath 1.8 ha of the field. It is comprised of 0.10 m diameter tile lines installed at a depth of 0.80 m and spaced every 12. The lines converge at a monitoring hut where water quality and flow measurements are collected. The tile drainage system design, manure type and spreader, and tillage systems are typical throughout Nova Scotia and therefore drainage water quality was expected to be representative of the region.

Constructed Treatment Wetland System

A surface flow wetland was selected because they have been successfully implemented in Nova Scotia (Smith *et al.* 2006; Wood *et al.* 2008). Steady state, first-order plug flow models, such as the k-C* model (Equation 1; Kadlec and Knight 1996), have been shown to adequately describe treatment in CTWs receiving episodic hydraulic and pollutant loading, such as that of drainage water (Carleton *et al.* 2001).

$$A = -\frac{Q}{k} * \ln\left(\frac{C_{out} - C^*}{C_{in} - C^*}\right) \quad (1)$$

Where: A = Wetland area (m²), Q = Annual inflow (m³/yr), k = First order reaction or areal rate constant (m/yr), C_{out} = Outflow concentration (mg/L), C_{in} = Inflow concentration (mg/L), and C* = Background concentration (mg/L). The k-C* model was applied monthly in 2005 to each contaminant. The maximum recorded C_{in} and the coinciding daily tile flow, extrapolated over 365 days to calculate Q, were used. C_{out} values were based on drinking water quality guidelines for NO₃⁻-N (FPTCDW 2008), irrigation water quality guidelines for *E. coli* (CCME, 2005). The k values used in this design were calculated by Jamieson *et al.* (2007) from a CTW receiving livestock wastewater in Nova Scotia. This yielded a maximum surface area (A) of 1025 m². Two independent CTW cells, were selected to allow (i) data replication, (ii) a higher length to width ratio while fitting within limited land availability and site topography, and (iii) flexibility to make repairs or modifications to one wetland while the other continues to operate. CTW specifications were based on several design manuals (NRCS 2002; USEPA 1988;1999). To address the concerns presented by a high water table a 12 mil woven polyethylene liner was installed. The wetland floors were sloped 0.2% and two 0.05 m perforated pipes, wrapped in filter fabric, were laid across the last two zones of each wetland to drain any water trapped beneath the liner. Tile drainage water is transported to the CTW by a 200 mm underground pipe. A vertical plastic fin glued inside a T splits flow to each cell. Knife valves allow flow to be shut-off to each cell. An in-line water level control structure at the outlet of each cell allows the water level to be controlled by adjusting plates. An emergency spillway at the outlet of each cell is used to safely discharge excess water if the control structure cannot. Water is then directed to the reservoir by a 30 m surface channel, which may provide additional treatment. Construction began in November 2006 and took approximately 14 days. Cattail (*typha spp.*) shoots were transplanted with a spacing of 1/m² into the shallow zones from a natural wetland in May 2007 and spread to cover the entire shallow zone by August 2007.

Reservoir/Dam Design and Construction

The function of the reservoir as part of the WRS is to store the water treated by the CTW until it is used for irrigation. An on-stream reservoir, which is formed by constructing a dam across the downstream end of a gully, was selected rather than a dugout reservoir primarily because of land availability and cost. The proximity of the gully to the field is close enough to allow for a reasonable length of irrigation main pipe. A field investigation consisting of test pits, permeability tests, and a review of soil maps was conducted to assess seepage potential of the reservoir and the suitability of site soil for use as dam fill. In-situ and Guelph permeability tests indicated that the site is characterized as slowly permeable, raising a concern about water retention by the reservoir. Test pits dug at the dam foundation site revealed a sandstone layer below the topsoil, capable of bearing the dam load. A contour survey of the gully was conducted and reservoir volumes were calculated for various dam locations. A maximum volume of 5000 m³ was calculated when the dam was situated at the downstream limit of the gully. The reservoir has a surface area of 0.3 ha and a maximum depth of 4.5 m, deep enough to provide fish habitat and limit aquatic plant growth. This volume is less than the potential volume of 8500 m³ but was limited by land availability, topography, and cost, as experience by Allred *et al.* (2003).

System Costs

A major obstacle in the adoption of the WRS is cost. Cost depends on factors such as existing components, irrigation requirements, drainage area, water quality, topography, and site geology. System costs for our system are listed in Table 1.

Table 1. System costs

| | Cost (\$CAD) |
|--------------------------------------|--------------|
| <i>Irrigation System</i> | |
| Pump | 4800 |
| Pipes | 9000 |
| Sprinklers | 2400 |
| Fittings | 2400 |
| <i>Constructed Treatment Wetland</i> | |
| Excavation | 10000 |
| Synthetic liner | 5700 |
| Control structures | 2000 |
| Pipes | 2500 |
| Valves | 600 |
| Spillway construction | 400 |
| Spillway rock | 400 |
| <i>Reservoir</i> | |
| Soil tests | 1500 |
| Engineering fees | 4700 |
| Grubbing and stump disposal | 10400 |
| Dam construction | 40000 |
| Compaction monitoring | 600 |
| Bentonite | 3000 |
| Spillway geotextile | 800 |
| Spillway rock | 2400 |
| Site seeding | 2200 |
| Total | 105800 |

Cost should be lowered as recommendations for optimizing WRS location, design, construction, and operation are implemented. It is also difficult to put a financial value on the environmental benefits. Wichelns (2005) investigated the economic feasibility of an integrated on-farm drainage management system in California and concluded that economic incentives and drainage water disposal regulations may be required to encourage farmers to bear the significant cost of the system. Operational challenges generally occurred during high flow events. During a few storm events, before vegetation was established, erosion caused significant damage to instrumentation, CTW berms and dykes, unknown hydraulic and pollutant loading, and sedimentation in the ditch between the CTW and reservoir. This challenge was addressed once vegetation was established. Vegetation debris, transported during fall high flow events, frequently obstructed control structures. This could be addressed through the use of larger control structures or intake screens.

Preliminary water quality results

Water quality, hydrological, and meteorological data were collected since November 2007. Preliminary data indicate that annual nitrate-nitrogen (NO_3^- -N) and *E. coli* reductions were 52% and 33%, respectively. Significant monthly variation was observed, and is attributed to the episodic hydrologic and pollutant loading of drainage water. Total phosphorus and soluble reactive phosphorus concentrations were typically below detectable levels (0.10 mg/L and 0.05 mg/L, respectively) at all sampling locations. Reservoir water quality exceeded irrigation water quality for *E. coli* guidelines (100 CFU/100 mL) during summer months and is attributed to environmental factors. This project provides a detailed case study of the design and construction process of a wetland-reservoir treatment and reuse system. Recommendations focusing on managing episodic hydrologic and pollutant loading are also made to optimize future systems in Nova Scotia.

Conclusion

Nutrient and pathogen loading from the disposal of agricultural drainage water is a major source of surface water quality degradation. Common effects of this type of degradation include eutrophication, and the contamination of drinking and irrigation water supplies. Climate change and increasing water demands also

threaten the availability of good quality water. Integrated water management systems have been used to address these challenges by treating and reusing drainage water. Before these systems can be implemented in Nova Scotia their location, design, construction, and operation need to be optimized to the cold climate and geology of the region. This project provides a detailed case study of the design and construction process of a wetland-reservoir treatment and reuse system. Recommendations focusing on managing episodic hydrologic and pollutant loading are also made to optimize future systems in Nova Scotia. The most likely implementation of wetland-reservoir drainage water treatment and reuse systems in Nova Scotia is on farms that do not have access to enough water. The benefits of creating an irrigation water source may help offset the capital costs of constructing the system. The system did supply more than enough water to irrigate the drainage area and it does significantly improve water quality. However it may not be able to consistently supply water that meets the irrigation water quality guidelines for *E. coli*. This is still an improvement over the typical case where no irrigation water treatment occurs. Regulations for discharging drainage water into the environment may be necessary to encourage farmers to adopt this type of system for the protection of the environment.

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Drainage under permanent beds in a furrow-irrigated Vertisol

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Abstract

Comparative studies of drainage and leaching under tillage systems in irrigated tropical and sub-tropical Vertisols are sparse. The objective of this study was to quantify drainage under cotton-based cropping systems sown on permanent beds in an irrigated Vertisol. Drainage and soil water storage were measured with the chloride mass balance method and neutron moisture meter, respectively, during the 2002-03, 2004-05, 2006-07 and 2008-09 cotton seasons in an on-going experiment in a Vertisol in NW NSW. The experimental treatments were: cotton monoculture sown either after conventional tillage or on permanent beds, and a cotton-wheat rotation on permanent beds where the wheat stubble was retained as *in-situ* mulch into which the following cotton crop was sown. In 2005, a split-plot design was superimposed on the existing experiment such that the main plot treatments were irrigation frequency ("frequent", 7-14 day irrigation interval; "infrequent", 14-21 day irrigation interval), and sub-plot treatments were the historical tillage system/crop rotation combinations. In comparison with cotton monoculture sown either after conventional tillage or on permanent beds, soil water storage, particularly during the early part of growing season when rainfall provided the major proportion of crop water requirements, and drainage were greatest when a cotton-wheat rotation was sown on permanent beds. Drainage was higher when irrigation frequency was higher. Drainage water losses in a furrow-irrigated Vertisol may be reduced and rainfall conservation improved by sowing a cotton-wheat rotation with *in situ* stubble retention under less frequent irrigation.

Key Words

Cotton, minimum tillage, farming systems, rotation, Vertisol

Introduction

Land preparation methods used in irrigated cotton (*Gossypium hirsutum* L.) production in Vertisols range from intensive to minimum tillage or permanent beds (McKenzie *et al.*, 2003). By definition, a permanent bed implies that the bed stays in place for several seasons in comparison with being ploughed down and reconstructed every year as with more intensive tillage systems (McKenzie *et al.*, 2003; (Hulugalle and Daniells, 2005). Long-term use of permanent beds can lead to significant improvement in soil physical, chemical and biological properties (Hulugalle and Daniells, 2005).

Comparative studies of drainage and leaching under tillage systems in irrigated tropical and sub-tropical Vertisols are sparse. Research has, however, been conducted in rainfed Vertisols. In tropical central Queensland, Tolmie and Radford (2004) observed that, compared with conventional tillage, deep drainage estimated using the chloride mass balance approach was greater with zero tillage and suggested that this was due to a higher numbers of drainage pores (McGarry *et al.*, 2000). Dalal (1989) inferred from soil chloride profiles that that drainage and leaching were greater under zero-tillage in southern Queensland. Drainage and leaching in fine-textured, swelling soils from temperate regions have also been reported to be greater under zero- or minimum-tillage than under intensive tillage (Addiscott and Thomas, 2000; Catt *et al.*, 2000; Shipitalo *et al.*, 2000). These authors suggested that the major pathways of nutrient and water movement under dry conditions were soil cracks and fractures, whereas under wet conditions, particularly in zero- and minimum-tilled soil, macropores created by earthworms and plant roots dominated. In summary, although direct measurements of drainage and leaching in Vertisols under varying tillage systems are sparse, inferences made from salt distribution in the soil profile suggests that drainage and leaching are likely to decrease with increasing tillage intensity. Drainage pathways in Vertisols may also differ with varying tillage intensity. The objective of this study was to quantify drainage under cotton monoculture sown after conventional tillage or on permanent beds, and under cotton in a cotton-wheat rotation sown on permanent beds.

Materials and methods

The experimental site was located at the Australian Cotton Research Institute, near Narrabri (149°47'E, 30°13'S) in New South Wales, Australia. Narrabri has a sub-tropical, semi-arid climate and experiences four

distinct seasons with a mild winter and a hot summer. The hottest month is January and July the coldest. Mean annual rainfall is 593 mm. The soil at the experimental site is an alkaline, self-mulching, grey clay, classified as a fine, thermic, smectitic, Typic Haplustert (Soil Survey Staff, 2006). Mean particle size distribution in the 0-1 m depth was: 64 g/100g clay, 11 g/100g silt and 25 g/100g sand. ESP values were of the order of 10 in the 0.6-1.2 m depth but were < 6 in the shallower depths.

Drainage was measured during the 2002-03, 2004-05, 2006-07 and 2008-09 cotton seasons in a long-term experiment, est. 1985 (Constable *et al.*, 1992) The experimental treatments were: cotton monoculture sown either after conventional tillage (slashing of cotton plants after harvest, followed by disc-ploughing and incorporation of cotton stalks to 0.2 m, chisel ploughing to 0.3 m followed by bed construction) or on permanent beds (slashing of cotton plants after harvest, followed by root cutting, incorporation of cotton stalks into beds, and bed renovation with a disc-hiller), and a cotton-wheat rotation on permanent beds where the wheat stubble was retained as *in-situ* mulch into which the following cotton crop was sown laid out in a randomized complete block design with four replications. The rows (beds) were spaced at 1-m intervals with vehicular traffic being restricted to the furrows. Cotton was sown during October and picked during May, and wheat was sown during May or June. Wheat was sown on beds and in furrows. Cotton and wheat rotation crops were furrow irrigated at an average rate of 1 ML/ha (100 mm) subject to water availability, rainfall and soil water content at intervals of approximately 7-14 days. In 2005, a split-plot design with two replications was superimposed on the existing experiment such that the main plot treatments were irrigation frequency (“frequent”, 7-14 day irrigation interval; “infrequent”, 14-21 day irrigation interval), and sub-plot treatments were the historical tillage system/crop rotation combinations. Quality of irrigation water is given in Table 1. Individual plots were 190 m long and 36-44 rows wide.

Table 1. Seasonal average of irrigation water quality during 2002-03, 2004-05, 2006-07 and 2008-09 cotton seasons. EC_w, electrical conductivity; SAR, sodium adsorption ratio.

| Season | pH | EC _w (dS/m) | Cl | Ca | Mg | K | Na | NO ₃ -N | SAR |
|---------|-----|---------------------------|--------|------|------|-----|-------|--------------------|-----|
| | | | (mg/L) | | | | | | |
| 2002-03 | 8.3 | 0.26 | 12.0 | 14.0 | 13.4 | 2.7 | 19.4 | 21.8 | 0.9 |
| 2004-05 | 8.1 | 0.30 | 18.2 | 23.0 | 14.9 | 3.8 | 38.0 | 5.2 | 1.5 |
| 2006-07 | 7.8 | 0.39 | 24.1 | 23.6 | 16.1 | 3.6 | 111.4 | 14.4 | 4.3 |
| 2008-09 | 8.1 | 0.38 | 24.3 | 31.8 | 15.3 | 6.0 | 47.4 | 24.4 | 1.7 |

Soil chloride concentration was evaluated in samples taken after cotton sowing and after cotton picking. Six 50-mm diameter soil cores were extracted from each plot with a tractor-mounted soil corer from the 0-0.30 m, 0.30-0.60 m, 0.60-0.90 m and 0.90-

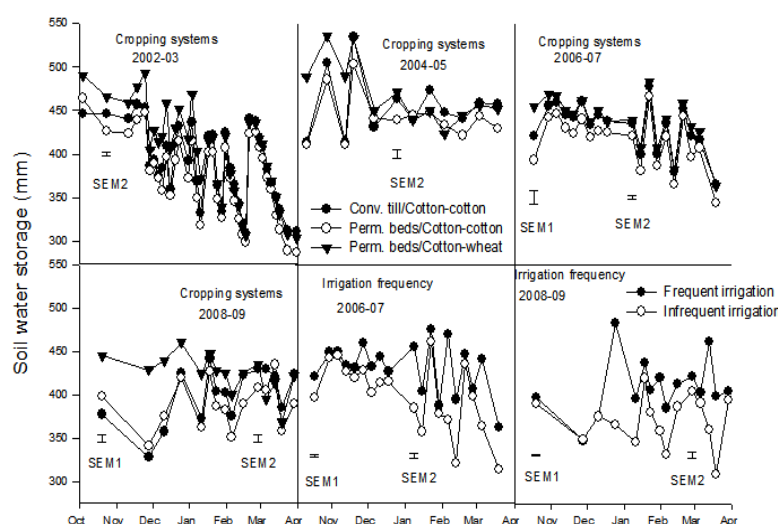


Figure 1. Effects of cropping system and irrigation frequency on soil water storage in the 0-1.2 m depth. SEM1, for first data point; SEM2, all other data points.

0.15 m, 0.15-0.30 m, 0.30-0.45 m, 0.45-0.60 m and 0.60-1.20 m depths subsequently. Air-dried soil was passed through a 2 mm-sieve and chloride concentration determined by AgNO₃ titration (Loveday, 1974). Soil water content in the 0.20-1.20 m depth interval was measured at 7-10 day intervals during the cotton season with a neutron moisture meter which had been calibrated *in-situ*. Soil water content in the soil surface was measured gravimetrically. Drainage was estimated with the chloride mass balance method assuming either steady state or transient state conditions (Weaver *et al.*, 2005). Results were analysed using analysis of variance, and means and standard errors of the means calculated.

Results and discussion

Soil water storage

Among cropping systems, soil water storage was generally highest under the cotton-wheat rotation sown on permanent beds (Figure 1). During growing seasons when rainfall was the major source of early season water for the cotton crop (2004-05, 2008-09) soil water storage was greatest under the cotton-wheat rotation sown on permanent beds (Figure 1). When a major proportion of early season water requirements were supplied by irrigation (2002-03, 2006-07), however, differences in soil water storage among cropping systems were smaller. The higher early season soil water storage under the cotton-wheat rotation when rainfall was adequate reflects the greater rainfall infiltration (Silburn and Glanville, 2002) and lower evaporation resulting from the *in situ* wheat stubble, and better soil water storage capacity due to its greater subsoil porosity (Hulugalle and Daniells, 2005; Hulugalle *et al.*, 2005). As the season progressed, and water used by the cotton increased (Tennakoon and Hulugalle, 2006), with much of it coming from irrigation, the magnitude of the differences among treatments decreased or disappeared.

In summary, the rainfall harvesting and storage capability of a cotton-wheat rotation where cotton is sown into *in situ* wheat stubble is superior to cotton monoculture sown either on permanent beds or after conventional tillage. Reduction in water availability due to a combination of drought and legislation has become a major constraint in irrigated farming systems in many Australian states during the past decade (DECC, 2009; eWater CRC, 2009). Consequently, management systems which conserve all rainfall received *in situ*, thus reducing the requirements for irrigation water, can contribute greatly to the sustainability of irrigated farming systems.

Drainage

Drainage, particularly in the deeper depths, was highest ($P < 0.01$) during the cotton seasons of 2004-05,

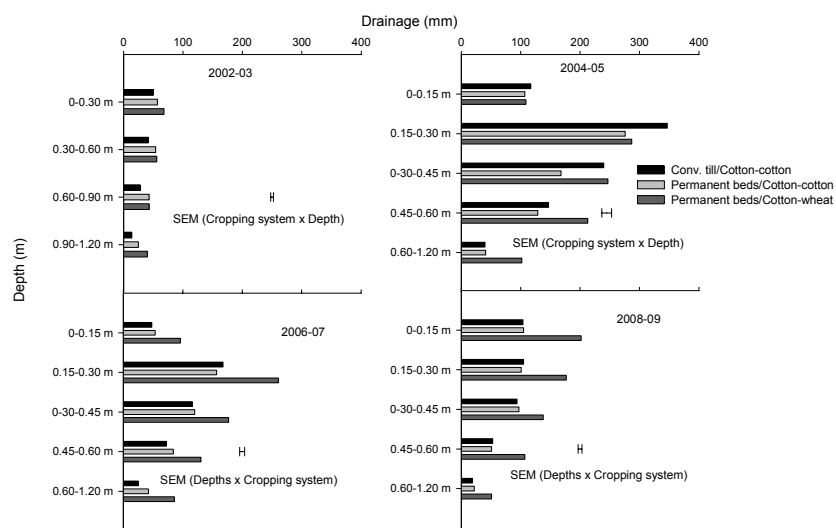


Figure 2. Effect of cropping system on drainage. Horizontal bar is SEM.

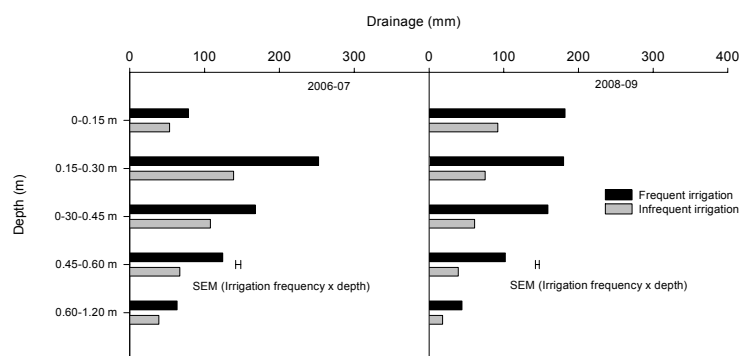


Figure 3. Effect of irrigation frequency on drainage, 2006-07 and 2008-09 seasons. Horizontal bar is SEM.

2006-07 and 2008-09 under the cotton-wheat rotation on permanent beds (Figure 2). This reflects the distribution of drainage pores in these treatments (Hulugalle *et al.*, 2005). During the 2002-03 cotton season, however, subsoil drainage did not differ significantly among treatments, although there was a trend towards higher values under permanent beds. Drainage also varied significantly ($P < 0.001$) with depth, with that out of the 1.2 m being lowest. Drainage was least during the 2002-03 season, presumably because water inputs were relatively low.

Drainage in all depths was greater ($P < 0.05$) with a higher irrigation frequency than with an “infrequent” irrigation frequency (Figure 3) during the 2006-07 and 2008-09 cotton seasons. This is not unexpected, as the proportions of water (as a fraction of seasonal rainfall + irrigation) which drained out of the root zone were similar with both irrigation frequencies. However, the pathways of drainage may have

differed between the two treatments. The drier soil profile in the infrequently irrigated treatments would have resulted in more cracking, and consequently drainage via the soil cracks would have been the dominant pathway (Favre *et al.*, 1997; Novak *et al.*, 2000) whereas with frequent irrigation and wetter soil, more water would have drained by matric flow.

Conclusions

In comparison with cotton monoculture, soil water storage, particularly during the early part of growing season when rainfall provided the major proportion of crop water requirements, and drainage were greatest when a cotton-wheat rotation was sown on permanent beds. Drainage was higher when irrigation frequency was higher, *viz.* 7-14 day irrigation interval. Drainage water losses in a furrow-irrigated Vertisol may be reduced and rainfall conservation improved by sowing a cotton-wheat rotation with *in situ* stubble retention under less frequent irrigation.

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Effect of elevated carbon dioxide on ¹⁵N-fertilizer recovery under wheat in Australia

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Abstract

The effect of elevated [CO₂] on the recovery of ¹⁵N-labeled urea by wheat was investigated under ambient (380 μmol/mol) and elevated (550 μmol/mol) [CO₂] at the Free-Air Carbon dioxide Enrichment (FACE) experiment in western Victoria, Australia. ¹⁵N-enriched (c. 10% atom excess) granular urea was applied to PVC microplots at 50 kg N/ha at rainfed and irrigated condition and two times of sowing. At physiological maturity, total biomass was increased by 23% under elevated [CO₂], as a result of a 25% and 22% increase in stem and root biomass, respectively, but not grain yield. Total N uptake of wheat was increased by 17% under elevated [CO₂], irrespective of irrigation and sowing time. Elevated [CO₂] increased grain C/N ratio by 11% only under irrigation, but not for stem and root. Elevated [CO₂] had no significant effect on percentage of N derived from fertilizer (%Ndff) for grain, stem and root but irrigation increased %Ndff for all wheat parts only in late sowing. Elevated [CO₂] did not alter the percentage of ¹⁵N recovered in grain, stem and root, but marginally increased the total recovery by 30% at late sowing time under irrigation. There were no significant effects of elevated [CO₂] on ¹⁵N recoveries in soil and total fertilizer N losses, indicating potentially similar availability of fertilizer N residues to subsequent crops in the longer term.

Key Words

Free-air carbon dioxide enrichment, climate change, nitrogen fertilizer recovery, irrigation, sowing time, wheat.

Introduction

Meeting crop nutrient demand is crucial to the sustainability of any crop production system (Torbert *et al.* 2004). Nitrogen (N) fertilizers are commonly applied in agricultural fields to optimize crop yield. However, the N-use efficiency of N-fertilizers by crop plants worldwide remains low, rarely more than 40% (Chen *et al.* 2008) while up to 20-50% of annual applications can remain unaccounted for at harvest (Azam *et al.* 1990; Pilbeam 1996). As atmospheric carbon dioxide (CO₂) levels rise due to the burning of fossil fuels, elevated [CO₂] generally results in higher crop biomass and yield but tissue N content is lower (Kimball *et al.* 2002). It is not clear whether higher fertilizer-N input is required or capable of meeting the increased demand of N under elevated [CO₂]. For example, elevated [CO₂] had no significant effect on recovery of fertilizer N in both the soil and grain sorghum (Torbert *et al.* 2004), but increased the recovery in spring barley (Martín-Olmedo *et al.* 2002) and rice (Weerakoon *et al.* 2005). The effect of elevated [CO₂] on N fertilizer recovery in crop, especially by wheat grown on low fertility sites, is unclear. This information is important to not only improving crop productivity, but also addressing Progressive N Limitation (PNL) (Luo *et al.* 2004) under elevated [CO₂] in the long term. The objective of this study is to investigate the interactions of elevated [CO₂], irrigation and sowing time on the recovery of ¹⁵N-labelled urea on a wheat field in Victoria, Australia using Free-Air Carbon dioxide Enrichment (FACE) facility.

Methods

Experimental site and design

Field experiments were conducted from early June to mid December in 2008 at Horsham, Victoria, Australia (36°45'S, 142°07'E) on a Vertosol used for a range of crops. The climate is temperate with an average rainfall and maximum temperature of 316 mm and 17.5°C during wheat growing season. The trial was a factorial combinations of two [CO₂]s, two irrigation scenarios and two times of sowing with four replicates in a randomized complete block design.

Carbon dioxide elevation

The elevation of atmospheric [CO₂] was achieved using a FACE system, consisting of 16 12 m diameter experimental areas, eight ambient and eight elevated. The two target CO₂ concentrations were 380 (ambient) and 550 μmol/mol (elevated). Carbon dioxide exposure commenced at sowing and terminated at harvest.

Wheat cultivation, fertilization and irrigation

Spring wheat (*Triticum aestivum* L. cv. Yitpi) was sown on 3 June (early sowing) & 6 August (late sowing) in 2008. The ¹⁵N labelled granular urea was topdressed at 50 kg N/ha at 3-5 leaf stage for both sowing times. The two irrigation scenarios were decile 5 (225 mm April to November) and decile 7 (275 mm April to November) rainfall conditions.

¹⁵N labelling

At each site, one circular PVC microplot (internal diameter of 24 cm and height 25 cm) was inserted to 20 cm depth. ¹⁵N-enriched granular urea with an abundance of 10.22 atom % was applied at the same rate (50 kg N/ha) as non-labelled urea was applied to the larger plots.

Sample collection

At harvest on 27 November and 16 December for early and late sowing time, respectively, plants were cut at ground level from within each microplot and separated into grain and aboveground biomass. In each microplot soil was sampled from 0-10, 10-20 and 20-40 cm depths. For the 0-10 and 10-20 cm depths, all the soil within the microplot was removed and a representative subsample was taken after the thorough mixing. For the 20-40 cm depth, one soil core was collected using a 5 cm diameter auger in each microplot. Major wheat roots were collected by digging out the top (0-10 cm) soil. Reference plant and soil samples were taken around 1 m away from the microplot to determine the background enrichments.

Chemical analysis

The plant and soil samples were dried at 60°C and 40°C, respectively, for 48 h, weighed, finely-ground and analysed for total C, total N and ¹⁵N enrichment by isotope ratio mass spectrometry following combustion. The recovery of ¹⁵N applied and percentage of plant N derived from fertilizer (%Ndff) were calculated by the equations described by Malhi *et al.* (2004):

$$\text{Recovery of applied N in plant (\%)} = \frac{[(\%N \text{ plant}) \times (\text{kg dry yield/ha}) \times (\text{atom}\%^{15}\text{N excess of total N in plant})]}{[(\text{Rate of applied N in kg N/ha}) \times (\text{atom}\%^{15}\text{N excess of total N in fertilizer})]} \quad (1)$$

$$\text{Recovery of applied N in soil (\%)} = \frac{[(\%N \text{ soil}) \times (\text{kg dry soil/ha}) \times (\text{atom}\%^{15}\text{N excess of total N in soil})]}{[(\text{Rate of applied N in kg N/ha}) \times (\text{atom}\%^{15}\text{N excess of total N in fertilizer})]} \quad (2)$$

$$\%Ndff = (\text{atom}\%^{15}\text{N excess of total N in plant} / \text{atom}\%^{15}\text{N excess of total N in fertilizer}) \times 100 \quad (3)$$

Statistical analysis

All data were analysed using the MINITAB 14 statistical package using a factorial model analysis of variance with main effects as [CO₂], irrigation and time of sowing.

Results

Crop biomass, total N uptake and C/N ratio

When averaged across all treatments, total biomass was 23% ($p < 0.01$) higher from elevated than from ambient [CO₂] microplots, regardless of irrigation and sowing time, which was associated with a 25% ($p < 0.01$) and 22% ($p < 0.01$) increase in stem and root biomass, respectively, but not grain yield ($p > 0.05$) (Table 1). There was a trend for the interaction between irrigation and sowing time to be significant ($p = 0.066$), with higher (82%) biomass observed under irrigation treatment in late sowing, but not in early sowing, irrespective of [CO₂]. Total N uptake of wheat was increased by 17% ($p < 0.05$) under elevated [CO₂], irrespective of irrigation and sowing time. Irrigation increased ($p < 0.05$) total N uptake by 86% only in late sowing, but not in early sowing. Elevated [CO₂] increased grain C/N ratio by 11% only under irrigation, owing to a slight decrease in N concentration, rather than a change in C concentration; no change in C/N ratio was observed for stem and root.

Table 1. Dry weight and total N uptake of crops grown from microplots under ambient and elevated [CO₂]

| [CO ₂] (μmol/mol) | Biomass (g/m ²) | | | Total N uptake (g/m ²) |
|-------------------------------|-----------------------------|-------|-------|------------------------------------|
| | Grain | Stem | Root | |
| 380 | 264.1 | 571.8 | 50.2 | 886.1 |
| 550 | 314.2 | 712.8 | 61.3 | 1088.3 |
| % change | +19 ns | +25** | +22** | +23** |

ns, no significant difference, * $p < 0.05$, ** $p < 0.01$

Percentage of plant N derived from fertilizer

Elevated [CO₂] had no significant effect on %Ndff for grain, stem and root, regardless of irrigation regime and sowing time. Irrigation increased %Ndff by 260% ($p < 0.001$), 313% ($p < 0.001$) and 66% ($p < 0.05$) for grain, stem and root, respectively, but only in late sowing (Figure 1).

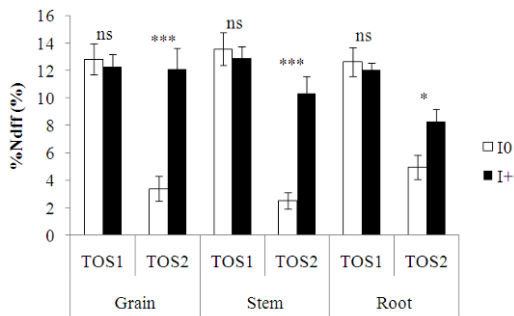


Figure 1. The effect of irrigation and sowing time on %Ndff of grain, stem and root of wheat crops. Bars indicate standard errors. ns, no significant difference, * $p < 0.05$, *** $p < 0.001$. I0: rainfed; I+ irrigated; TOS1: early sowing; TOS2: late sowing

Recovery of fertilizer in the crop and soil

The percentage of ^{15}N recovered in the crops averaged 42-48% and 4-31% for early and late sowing, respectively (Figure 2). Elevated $[\text{CO}_2]$ did not alter the percentage of ^{15}N recovered in grain, stem and root irrespective of irrigation regime and sowing time, but increased the total recovery by 30% ($p = 0.066$) at late sowing time under irrigation (Figure 2). The percentage of ^{15}N recovered in the soil averaged 24-28% and 52-80% for early and late sowing, respectively. The percentage recovered was not significantly different between ambient and elevated $[\text{CO}_2]$ for soil depths of 0-10 cm and 10-20 cm except less (46%, $p < 0.01$) ^{15}N was recovered in the lower soil depth (20-40 cm) under elevated $[\text{CO}_2]$ at early sowing. When averaged across soil depths, elevated $[\text{CO}_2]$ had no significant effect on soil ^{15}N recovery (Figure 2).

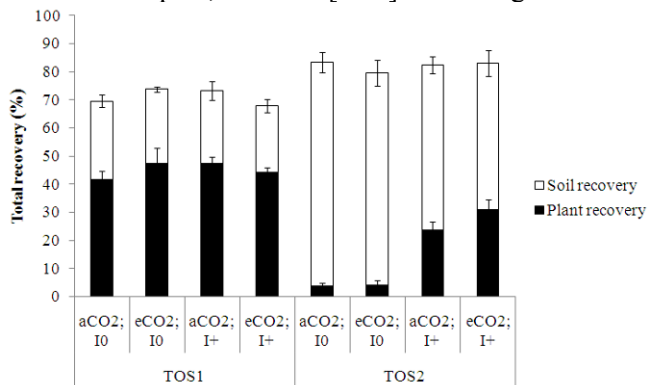


Figure 2. The effect of elevated $[\text{CO}_2]$, irrigation and sowing time on ^{15}N fertilizer recovery of plant (■) and soil (□). Bars indicate standard errors. aCO₂: ambient $[\text{CO}_2]$; eCO₂: elevated $[\text{CO}_2]$ I0: rainfed; I+ irrigated; TOS1: early sowing; TOS2: late sowing

Fertilizer unaccounted for

At harvest, the percentage of the ^{15}N -labelled fertilizer unaccounted for was reduced ($p < 0.001$) from 29% in early sowing time to 18% in late sowing, but was not affected by either elevated $[\text{CO}_2]$ or irrigation.

Discussion

Effect of $[\text{CO}_2]$, irrigation and sowing time on N demand

Elevated $[\text{CO}_2]$ significantly increased total biomass by 23%, which is consistent with the 20% stimulation of biomass at maturity reported in the main FACE experiment (Norton *et al.* 2008), with total N uptake by the crop 17% higher under elevated than ambient $[\text{CO}_2]$ microplots. These results indicate that N demand was increased, and PNL is possible in this wheat field if additional N input (or reduction in N losses) is not made to compensate for the greater demand in the longer term (Luo *et al.* 2004). There was no change in C/N ratio (other than for grain) in contrast to other studies where elevated $[\text{CO}_2]$ increased the C/N ratio of plant residues (Kimball *et al.* 2002). Nonetheless, the present study suggests that PNL may eventually occur as a result of increased biomass and residue production, rather than changes in C/N ratio of crop tissues, as grain is generally removed from the field. Irrigation was critical for the greater increase in crop biomass and N uptake, and therefore demands for N, in drier and warmer climate.

Effect of $[\text{CO}_2]$, irrigation and sowing time on fertilizer uptake

Recovery of ^{15}N -labelled fertilizer in the early sown crop (42-48%) was higher than late sowing (4-31%), and was slightly higher than the range of 22-40% recovery of ^{15}N fertilizer observed by Carranca *et al.*

(1999) for wheat, but within the range (22-59%) reviewed by Chen *et al.* (2008). Despite the increase in N demand under elevated [CO₂], the contribution of fertilizer N to the uptake of N in plant was not significantly different from the ambient, as evidenced by the values of %Ndff. Moreover, no significant effect of elevated [CO₂] was observed on the percentage of fertilizer recovery by the crop itself, which agrees with Torbert *et al.* (2004)'s study on ¹⁵N fertilizer recovery by sorghum. Furthermore, the recovery of the fertilizer N remaining in the soil at crop maturity did not differ between [CO₂] treatments. In the short term, PNL is unlikely because of relatively high soil N supply at this site. Nevertheless, the marginally significant increase in total plant recovery N percentage under elevated [CO₂] and irrigation in late sowing suggests water availability plays profound role when the climate becomes drier and warmer under future atmospheric [CO₂]. Early sowing resulted in higher recovery of fertilizer N in the crop than that remaining in the soil in this semi-arid environment, but the opposite was observed in late sowing with a drier climate. This is consistent with a meta-analysis of the recovery of ¹⁵N-labelled fertilizer applied to wheat (Pilbeam 1996), which shows greater recovery of fertilizer N by crops in humid environments, but the opposite happened in dry environments. This has major implications on fertilizer recovery of crops grown in regions which are predicted to be drier in the future. A higher percentage of unrecovered ¹⁵N was observed under early sowing than late sowing, which may be attributed to greater denitrification in early sowing when soil moisture was higher (Bolan *et al.* 2004) which may lead to high N₂O flux. N loss via leaching was unlikely as evidenced by the small percentage of ¹⁵N recovered in soil depth of 20-40 cm (data not shown). Ammonia volatilization was likely increased with air temperature (Bolan *et al.* 2004) (late sowing in the present study), yet the greater N loss in early sowing suggests denitrification contributed more than ammonia volatilization in the percentage ¹⁵N unaccounted for.

Conclusions

Wheat total biomass, total N uptake and grain C/N ratio were increased under elevated [CO₂]. Progressive N Limitation, if any, is likely a result of greater biomass and residue production, rather than change in C/N ratio of crop residue. Elevated [CO₂] had no effect on the relative contribution of fertilizer N to the uptake of N in plant, indicating soil N was not limiting in the present study, and higher fertilizer-N application is not necessary. There was an indication however of higher fertilizer recovery in the potentially drier and warmer climates predicted under future [CO₂], if irrigation is applied. Higher inputs of N may be required to satisfy greater N demand and to address PNL, if it happens. Irrigation was also important to crop biomass, total N uptake and %Ndff in late sowings. Longer term experiments, in combination with process based plant-soil modeling, are required to assess the likelihood of PNL occurring in this system.

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Effects of elevated CO₂ and O₃ on soil amino sugar from wheat straw decomposition in a meadow brown soil of Northeast China

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Abstract

Amino sugars, being predominantly of microbial origin, can help elucidate the role of microbes in decomposition of crop residues in soil. A 12-month CO₂ and O₃ enrichment field experiment was conducted in open-top chambers in Shenyang suburb of Northeast China to study the dynamic changes of soil amino sugar during the decomposition of spring wheat straw. Compared with an ambient treatment, a significantly higher amount of soil glucosamine (GluN), galactosamine (GalN), and muramic acid (MurA) was observed in treatment elevated O₃ across the 12 months and in treatment elevated CO₂ in the first 4 months, which illustrated the stimulation effects of elevated O₃ and CO₂ on the proliferation of soil microbes. The GluN/MurA ratio under elevated O₃ increased and decreased under elevated CO₂, suggesting that elevated O₃ favoured the dominance of fungi decomposition of spring wheat, while elevated CO₂ stimulated the bacterial population and its decomposition.

Key Words

Elevated CO₂ and O₃; amino sugar; wheat residue decomposition; meadow brown soil.

Introduction

Amino sugar is a major constituent of microbial cell wall (Turrión *et al.* 2002; Roberts *et al.* 2007). Its amount in soil can be used to estimate soil microbial mass (Amelung 2001; He *et al.* 2006). Soil glucosamine (GluN) is the dominant component of soil amino sugar, and exists in fungal cell wall. Muramic acid (MurA) uniquely originates from terrestrial bacteria (Amelung 2001), and galactosamine (GalN) has dubious microbial origins. The summation of these three amino sugars represents soil microbial mass, and the relative abundance of MurA and GluN has been successfully used to assess the relative contribution of soil bacteria and fungi to the turnover of organic matter in many soils (Zhang *et al.* 1998; Amelung *et al.* 2001; Dai *et al.* 2002; Glaser *et al.* 2004; 2006; Liang *et al.*, *et al.* 2007, 2008). Elevated CO₂ and O₃ have definite effects on soil microbial community structure (Hu *et al.* 2006; Shi *et al.* 2006; Yue *et al.* 2007; Van Groenigen *et al.* 2007; Kanerva *et al.* 2008). Glaser *et al.* (2006) investigated the effects of elevated pCO₂ on the bacterial and fungal-derived C in *Lolium perenne* pasture soil by using compound-specific isotope analysis ($\delta^{13}\text{C}$) of soil amino sugar. Van Groenigen *et al.* (2007) studied the effects of elevated CO₂ on the fungal decomposition pathway by quantifying the contents of soil GluN, MurA, and Gal N in three terrestrial ecosystems, but few studies were made on the effects of elevated O₃ on soil microbial community by soil amino sugar analysis. Returning wheat straw to farmland is an important agricultural practice in China. To understand the effects of elevated CO₂ and O₃ on the relative contribution of soil bacteria and fungi to the turnover of amended wheat straw, a 12-month CO₂ and O₃ enrichment experiment with meadow brown soil, an important agricultural soil in Northeast China, was conducted, using the summation of GluN, MurA and GalN to represent soil microbial mass, and the GluN/MurA ratio to characterize the relative contribution of soil fungi and bacteria to the decomposition of amended wheat straw.

Methods

Study site and treatments

The enrichment experiment was conducted in open-top chambers (OTCs, 3 m in diameter and 2.8 m in height) at the National Field Observation and Research Station of Agro-ecosystems in Shenyang suburb of Northeast China (41°31' N, 123°24' E). The annual mean air temperature is 7-8 °C, annual mean precipitation is 700 mm, and frost-free period is 147-164 d. Meadow brown soil is the main soil type for agricultural production. Three treatments were installed, i.e., control (ambient, ~342 $\mu\text{mol CO}_2/\text{mol}$ and ~40 nmol O₃/mol), elevated CO₂ (550 $\mu\text{mol CO}_2/\text{mol}$), and elevated O₃ (80 nmol O₃/mol). They were tri-replicated, and arranged in a randomized complete block design. CO₂ was provided 24 h/d, and O₃ was provided 8 h at day time (08:00-12:00 and 14:00-18:00) from 27 April to 27 June. The CO₂ and O₃ concentrations in OTCs

were measured by ES-D infrared CO₂ sensor and S-900 O₃ analyser, with the variance being ± 4 and $\pm 9\%$, respectively (Ruan *et al.* 2006). Chinese spring wheat (*Triticum aestivum* L. cv. Liaochun 17) was sown in pots (23 cm in diameter and 20 cm in height) on 2 April, and harvested on 30 June 2006. 20 plants at three-leaf stage were established in each pot. The pots were watered periodically to prevent water deficit. Ammonium-based N fertilizer and P fertilizer were applied as basal at 150 kg N hm⁻² and 10 kg P hm⁻², respectively. Each OTC contained 25 pots.

Sample preparation

Soil samples were taken from the pots after spring wheat harvested, and air-dried and sieved to < 2 mm, with their properties listed in Table 1. The harvested aboveground part of spring wheat was crushed into ~1 mm pieces, and mixed with 150 g soil at a rate of 3% of dry soil. The soil-wheat mixture was adjusted to 60% water-holding capacity, and placed into 300-mesh sieve nylon bags. A total of 144 bags were buried at 10 cm depth in 9 OTCs on 7 July 2006. In 2007, the bags were exposed to elevated CO₂ and O₃ during spring wheat growth period, and two bags per OTC were sampled at the 0, 1st, 2nd, 3rd, 4th, 9th, 10th, 11th and 12th month (i.e. 7 July, 11 August, 11 September, 8 October and 8 November 2006, and 9 April, 5 May, 8 June and 9 July 2007, the 5th- 8th month was frozen period) for amino sugar analysis, respectively.

2.3 Amino sugar extraction and derivatization. Soil amino sugar was extracted and purified according to Zhang and Amelung (1996). Briefly, 100 µg myoinositol was added as an internal standard to the samples containing about 0.3 mg N. The samples were hydrolyzed with 10 mL 6 M HCl at 105°C for 8 h, and the released amino sugar was separated from impurities by neutralization (pH 6.6-6.8) with 0.4 M KOH. Before derivatization, 100 µg N-methyl-glucamine was added as a recovery standard.

The aldonitrile derivatives of amino sugar were prepared according to Guerrant and Moss (1984). Samples were dissolved in 0.3 mL derivatization reagent, and heated at 75-80°C for 30 min. After acetylation with 1 mL acetic anhydride at 75-80°C for 20 min, dichloromethane was added, and excess derivatization reagent was removed by washing with 1 mL 1 M HCl for four times, air-dried at ambient temperature, and finally dissolved in 0.3 mL ethyl acetate-hexane (1:1 v/v). The aldonitrile derivatives were analysed on an Agilent 6890 gas chromatograph equipped with an Agilent DB-5MS (30 m × 0.25 m × 0.25 µm) column and a flame ionization detector. The optimized carrier gas flow and oven temperature programs for successful separation of aldonitrile derivatives on the DB-5MS chromatograph column were as follows: 0.8 mL/min constant gas flow, 1.0 µL injection volume at 1:10 split ratio, 250 and 120°C injector and oven temperatures, respectively, at injection time. For further details on temperature program, see Zhang and Amelung (1996).

Data analysis

All data were analysed by repeated measures ANOVA. The metadata were analysed by using SPSS 13.0 and Excel 2003 software.

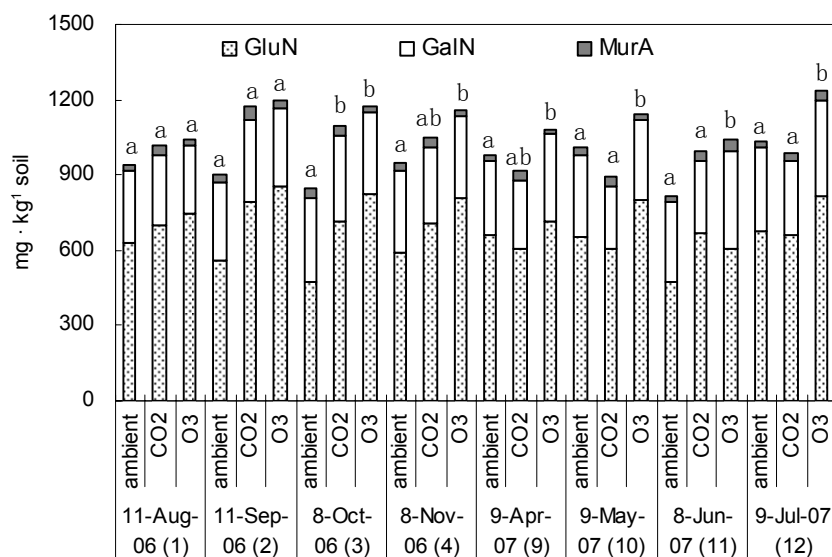
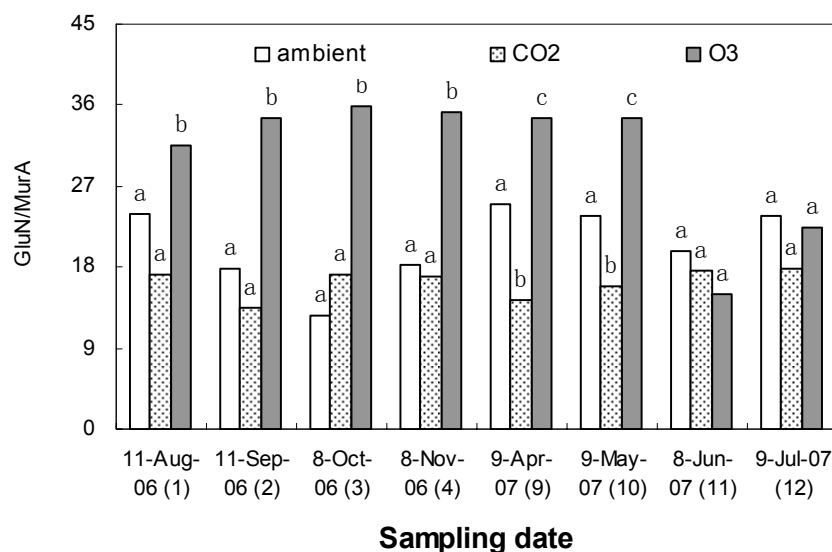
Results

During 12 months decomposition of spring wheat straw, the average summation of soil GluN, GalN and MurA across all treatments was 813-1234 mg/kg soil. A significantly higher summation of soil GluN, GalN, and MurA was observed in treatment elevated O₃ from the 2nd month to the end of the experiment, and in treatment elevated CO₂ from the 2nd to the 4th month, compared with that in treatment ambient ($p < 0.05$, Figure 1). GluN was the dominant amino sugar and generally mirrored the total pattern of soil amino sugar (Figure 1). For example, GluN, GalN, and MurA contributed 68.0%, 29.7%, and 2.3% in treatment elevated O₃, 67.0%, 28.9%, and 4.2% in treatment elevated CO₂, and 62.8%, 34.0% and 3.2% in treatment ambient, respectively, which were similar to the findings of other studies (Glaser *et al.* 2006; Van Groenigen *et al.* 2007).

The average amount of GluN in treatments elevated CO₂ and O₃ was significantly higher than that in treatment ambient ($p < 0.05$). Elevated O₃ significantly increased the average amount of GalN and but decreased that of MurN, compared with elevated CO₂ ($p < 0.05$). The ratio of GluN to MurA has been successfully used to track the relative contributions of fungi and bacteria to the decomposition of spring wheat straw. In present study, the GluN/MurA ratio increased by 47.9% under elevated O₃, but decreased by 20.7% under elevated CO₂, compared with that under ambient (Figure 2), suggesting that elevated O₃ favoured the dominance of fungi decomposition of spring wheat, while elevated CO₂ stimulated bacterial population and its decomposition.

Table 1. Chemical properties of test soil.

| Treatments | TOC (g/kg) | Total N (g/kg) | C/N | GluN (mg/kg) | GalN (mg/kg) | MurA (mg/kg) | Total amino sugar (mg/kg) |
|-----------------|---------------|-------------------|-----|-----------------|-----------------|-----------------|------------------------------|
| Ambient | 10.01 | 1.21 | 8.3 | 586.0 | 276.6 | 31.8 | 894.4 |
| CO ₂ | 10.18 | 1.18 | 8.6 | 586.1 | 272.6 | 35.3 | 894.0 |
| O ₃ | 9.94 | 1.27 | 7.8 | 629.0 | 271.8 | 24.2 | 925.0 |

**Figure 1. Amount changes of soil GluN, GalN, and MurA over time. Data are the means ± standard error of three replicates. Within a given date, the means with the same letter were not significantly different at P > 0.05.****Figure 2. Ratio changes of GluN/MurA over time. Data are the means ± standard error of three replicates. Within a given date, the means with the same letter were not significantly different at P > 0.05.**

Conclusion

The increased summation of soil GluN, GalN and MurA in treatments elevated O₃ and CO₂ illustrated the stimulation effects of elevated O₃ and CO₂ on the proliferation of soil microbes. The larger GluN/MurA ratio under elevated O₃ than under elevated CO₂ suggested that elevated O₃ favoured the dominance of fungi over bacteria, while elevated CO₂ stimulated bacterial population and its decomposition.

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Emissions of CO₂ and N₂O from volcanic soil under different crop management using closed non-fixed chambers

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Abstract

Agriculture soils may be a potential source of greenhouse gases, however it is not directly estimated yet in countries like Chile, which has special soil and climate conditions for a competitive agricultural activity and particular nutrient cycling patterns and processes in volcanic soils. The present work shows the first data of CO₂ and N₂O fluxes estimated from direct gas samples of passive chambers on volcanic soils in a crop system of Southern Chile. Gas samples were obtained from a oat-wheat rotation in a volcanic soil surface in Chile (36° S, 72° W) under two treatments of N fertilization (150 kg N/ha NH₄⁺-and NO₃⁻-N, respectively) and two lime dose treatments (0,5 and 1 Ton lime/ha, respectively), from closed non-fixed chambers and fluxes of CO₂ and N₂O were estimated from a gas chromatograph concentration in function of time during gas accumulation in the chambers headspace in a period of 1 year. Soil variables, including temperature, water content and mineral N content were registered in order to check correlations with gas fluxes. Results show low emissions of GHG from volcanic soils of Chile and no important influences of some agriculture management and environmental variables.

Key Words

Greenhouse gases, agriculture, nitrogen, global change, air pollution.

Introduction

Agriculture is either a potential source or sink of Greenhouse Gases (GHG), mainly carbon dioxide (CO₂) and nitrous oxide (N₂O), which may contribute to the temperature increase of the atmosphere and climate change. The patterns of GHG emissions from Chilean soils and the related processes are of great interest for the different agricultura; scenarios in order to place Chile in the global agreements of mitigation policies. Pristine ecosystems of southern Chile still reflect efficient nutrient cycling, only comparable to pre-industrial ages (Huygens *et al.* 2008), thus, the biogeochemical patterns of volcanic soils in terrestrial ecosystems of southern Chile may promote nutrient retention and limit GHG emissions. However, ecosystem disturbances such as agricultural activities, disrupt the retention patterns in soil and favor nutrient losses, which may occur as GHG (Van Cleemput and Boeckx 2005). Agricultural activities are increasing at Southern Chile and will certainly change the environmental patterns and processes due to an intensive and inappropriate soil uses that should increase GHG emissions. Research on gas fluxes directly from the soil surface is seriously suggested in order to improve knowledge on GHG mitigation and future propositions of sustainable agriculture systems. A Chilean GHG inventory for agriculture activities has been made with empirical information from the International Panel for Climate Change (IPCC) (Novoa *et al.* 2000; Geng 2003; DICTUC 2004) and estimate a flux of 10.5 Tg eq.-CO₂/y, being the methane (CH₄) and N₂O, the main gases from this fountain, while CO₂ is generated mainly from plant respiration processes and is compensated during cycle of year. Such inventory is a proposed method related to the first uncertainty level (Tier 1) and allowed to have the baseline preliminary data. However, it is urgently necessary to quantify GHG emissions at different agroecosystems due to the variability of geographic location, climate, productive levels and management involved on, which will decrease the level of uncertainty (Tier 2) (IPCC 2007). Chile is one of the countries that have the conditions to be a leader in agricultural products with an environmental seal of low GHG emissions, because of its open economy, low country-risk and the international commercial agreements that make easier the transactions. A first methodological approach regarded to passive chambers to estimate GHG fluxes from soils (Livingston and Hutchinson, 1995; Hutchinson and Livingston, 2002), has been widely used in agriculture soils and contributed to the recent knowledge of *in situ* GHG emissions (Livingston *et al.*, 2006), and may be easily complemented with gas chromatography (Van Cleemput and Boeckx 2002). The GHG sampling *in situ* may be done from several agricultural systems under different soil and climate conditions including a non-equilibrium pattern like soil erosion events which have a major impact to atmospheric CO₂ (Follett *et al.* 2001). Soil respiration from crop systems represents 10 to 15 fold

greater CO₂ emissions than from fossil fuels (Raich and Schlesinger 1992). Emissions of GHG has been estimated *in situ* from several crop systems using similar methodology (Beauchamp 1997), as well as from N fertilized pastures (Groffman *et al.* 1993; Abbasi and Adams 2000), and related to environmental variables like soil temperature and water content (Choudhary *et al.* 2002; Ponce-Mendoza *et al.* 2006). The biological processes of nutrient cycling could be correlated to GHG emissions of soil through the chambers quantification (Smith *et al.* 2003; Roser *et al.* 2006). The present work presents the first values of GHG fluxes from crop soils in Chile using passive chambers and gas chromatography analyses.

Methods

Gas sampling was done in a long-term (12 years) field experiment of crop rotation (oat-wheat) in Southern Chile (36° S, 72°W), on a volcanic (Typic Fulvudand) soil. Complete randomized blocks (n = 3) were set to assess the effect of N amendments and lime: nitrate-N fertilization; (T1); ammonium-N fertilization (T2); ammonium-N + 0.5 Ton lime/ha (T3); and ammonium-N + 1 Ton lime/ha (T4). The rate of fertilizer was 150 kg N/ha for every treatment. Gas sampling was done from March of 2008 to December January of 2009, every month and after important events of precipitations and crop fertilization. Passive closed non-fixed chambers (4 L) were set in soil surface and gas samples were collected each 15 minutes starting at time 0 from the chamber headspace during 45 minutes of gas concentration (Hutchinson and Livingston 2002; IAEA 1992). The concentrations of gases (µg/L) were estimated with a Perkin Elmer Clarus 600 Gas Chromatograph with a Porapak Q column and a Flame Ionized Detector plus methanizer for CO₂ and an Electronic Capture Detector for N₂O (Van Cleemput and Boeckx 2002). Fluxes of GHG were estimated for each sampling date with the linear relation of gas concentration in function of time and reported as Mg CO₂/ha/y and kg N₂O/ha/y. At each gas sampling date, different soil variables were registered in order to find correlation to GHG fluxes. Soil and air temperature (°C) was registered with a digital thermometer, gravimetric soil moisture was estimated with a TDR di-electric detector and reported as percent of Water-Filled Pore Space (WFPS%) (Linn and Doran, 1991) and mg NH₄⁺- and NO₃⁻-N kg soil were reported from estimations in FIA Star Foss mineral N analyzer. One-way ANOVA and Honest Tukey analyses of data were used to detect effects (p<0.05) of field treatments on GHG emissions and linear correlations to soil variables.

Results

Fluxes of CO₂ ranged from about 20 to 50 Mg/ha/y, mainly before the beginning of winter and during spring, being at much lower values in the other months (Figure 1). No clear differences were reported for CO₂ among different crop management and the seasonal variation seems to be the main factor of influence to CO₂ fluxes in volcanic soils of Chile. Fluxes of higher values of CO₂ fluxes corresponded to soil temperature about 13 °C and 80% WFPS (not shown), which must be the best condition of soil respiration at this agriculture zone. Fertilization and lime amendment to soil occurred during October, coinciding with favorable conditions of soil temperature and moisture. This fact masks crop management effects on gas fluxes. Fluxes of N₂O reported very negligible values never higher than 5 kg N₂O/h/y during the period of study and the environmental variables had no influence in these fluxes (data not shown)

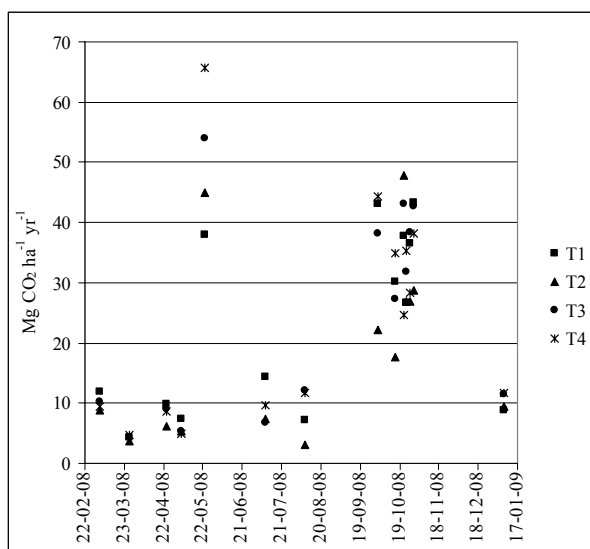


Figure 1. Fluxes of CO₂ (Mg/ha/y) from passive chambers at different crop soil management of N fertilization and lime amendments in a volcanic soil of Southern Chile during the study period.

Results of GHG fluxes in Chilean volcanic soils with crop managements are initially indicating a conservative pattern that seems to have less influence of management or environmental variables. Compared to previous research (Choudhary 2002; Ponce-Mendoza 2006), the fluxes of gases in volcanic soils of Chile may support some agriculture activities without major impact.

Conclusion

The fluxes of N₂O were clearly negligible for every crop management (field treatments) and seem to have no influence of N amendments and lime. However, CO₂ fluxes responded to a season pattern being affected by soil temperature and moisture during the year, while a less clear effect of crop management also occurred with CO₂ emissions for this soil. Mineral N variation is apparently not related to GHG fluxes. The present data show initially that a volcanic soil of Southern Chile has a low potential of GHG emission under agricultural activities.

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Heavy metal contamination of water bodies, soils and vegetables in peri urban areas of Bangalore city of India

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Abstract

A study was conducted in peri-urban Bangalore where city wastewaters from four water bodies viz, Bellandur, Varthur, Byramangala and Nagavara tanks were used for cultivation of vegetable crops to assess heavy metal contamination of water, soil and vegetables. Analyses revealed high concentrations of Cd and Cr in waters of all the tanks, exceeding the recommended levels of 0.01 and 0.1 mg/L respectively. Concentration of Cd was highest in waters of Bellandur (0.039mg/L) and concentration of Cr was highest in waters of Byramangala tank (0.311mg/L). Among all the tanks, Bellandur and Varthur were found to be highly contaminated with Cd, Pb and Ni. The concentration of heavy metals (mg/kg) in soils receiving sewage waters from the four tanks ranged from 1.92 -2.90 for Cd, 47.04-68.12 for Pb, 35.08-92.78 for Cr and 48.2-57.3 for Ni. The Cd and Pb contents were highest in the soils near Varthur and Bellandur tanks, while Cr was highest in soils near Byramangala. A similar trend was observed with respect to heavy metal content of vegetables. Among all the vegetables, Amaranthus and palak, accumulated higher concentrations of heavy metals followed by carrot and radish. The Cd concentration of all the vegetables grown near Varthur and Bellandur tanks exceeded the PFA safe limit. Pb and Ni concentrations exceeded the safe limits in all the vegetables in all the tank areas.

Key words

Cadmium, lead, chromium, nickel, soil, vegetables.

Introduction

Contamination of environment with toxic heavy metals has become one of the major causes of concern for human kind. Heavy metals in surface water bodies, ground water and soils can be either from natural or anthropogenic sources. Currently, anthropogenic inputs of metals exceed natural inputs due to increased urbanization and industrialization. Industrial wastes, atmospheric deposition from crowded cities and other domestic wastes are among the major sources of heavy metals in the urban sewage (Sorme and Lagervist 2002). Bellandur tank, Varthur tank, Byramangala tank and Nagavara tank are important water bodies of Bangalore. These tanks are part of the city drainage system that drain untreated and partially treated domestic sewerage and industrial effluents from a number of small scale units like garment factories, electroplating industries, distilleries, etc. The farmers in peri urban areas use water from these tanks for cultivation of vegetables. Soils receiving these waters accumulate heavy metals to varying degrees depending on their concentration in water and the frequency of irrigation. The heavy metals are absorbed by crops along with other essential plant nutrients. Contamination of soils and crops with these metals may have adverse effects on soil, plants, animals and human beings. The present study was aimed at finding out the levels of contamination of four toxic heavy metals viz. cadmium (Cd), lead (Pb), chromium (Cr), and Nickel (Ni) in four tanks, in the surrounding soils receiving water of these tanks and the vegetables grown in those soils.

Methods

Surveys have been conducted from 2005-2008 in the villages surrounding Bellandur, Varthur, Byramangala and Nagavara tanks where farmers were using water from the tanks for cultivation of vegetables. A farm which is away from these tanks and where bore well water was used for growing vegetables was included for study as uncontaminated control site. The water samples from all four tanks and bore well of uncontaminated site were collected in polyethylene bottles for analysis. Soil samples and samples of six vegetables viz. amaranthus, palak, carrot, radish, tomato and beans from the farmers fields were also collected. Soil and vegetable samples from uncontaminated field were also collected in similar fashion. Soil samples were dried at room temperature and ground to fine powder. Vegetable samples were dried in oven at 80° C, powdered and passed through a 2mm sieve. The soil and vegetable samples were digested with triacid

mixture (HNO_3 , HClO_4 and H_2SO_4 in 5:1:1 ratio) (Allen *et al.* 1986). The total heavy metal contents in water samples, digested soil and vegetable samples were estimated using Perkin Elmer Flame Atomic Absorption Spectrophotometer. The standard deviation of each result was estimated.

Results

Heavy metals in tank waters

Mean heavy metal concentration (mg/L) of water from the four water bodies is given in Table 1. Metal concentrations ranged from 0.014-0.039 for Cd, 0.039-0.075 for Pb, 0.120-0.291 for Cr and 0.027-0.042 mg/L for Ni. In comparison with the standard guidelines of irrigation water (Pescod 1992), it was found that mean Cd and Cr contents of all the tank waters exceeded the recommended levels of 0.01 and 0.1 mg/L respectively, while contents of Pb and Ni were within safe limits. The levels of all the four heavy metals were within safe limits in bore well waters of uncontaminated site. Among all the four tanks, Bellandur and Varthur tanks were highly contaminated with Cd, Pb and Ni. This may be due to rapid industrialization and urbanization around these tanks, increased number of IT industries, electroplating industries and a number of small scale industrial units and releasing of waste waters and other solid wastes from these units into these tanks through storm water drains. The higher levels of Cr in Byramangala tank can be attributed to waste waters and effluents released from the chromium electroplating industries in the surrounding areas. The mean Cd, Cr and Ni contents in waters of four tanks were about 20-25 times higher than the bore well waters of uncontaminated site.

Heavy metals in soils

The concentration of heavy metals (mg/kg) in agricultural soils receiving sewage waters from the four tanks ranged from 1.92 -2.90 for Cd, 47.04-68.12 for Pb, 35.08-92.78 for Cr and 48.2-57.3 for Ni (Table 2). The mean Cd, Pb, Cr and Ni contents of uncontaminated site were 0.90, 39.6, 34.2 and 34.9 mg/kg respectively. The Cd content was highest (2.90 mg/kg) in the soils near Varthur tank followed by the soils near Bellandur tank (2.38 mg/kg). These elevated concentrations may be due to long term use of tank waters for irrigation. The mean concentration of Pb was also highest in soils near Varthur and Bellandur tanks (68.12 and 64.9 mg/kg respectively). This can be attributed to nearness to highway, increased traffic, atmospheric deposition and prolonged use of tank waters. The mean concentration of Cr was highest in soils near Byramangala (92.78 mg/kg). This might be due to long term use of tank water which contained the waste water effluents discharged from chromium electroplating industries. Ni content was highest in soils near Vathur (57.3 mg/kg). This may be due to effluents discharged from electroplating industries around the tank.

Heavy metals in vegetables

Figure 1 shows highest content of Cd in almost all the vegetables grown near Bellandur and Varthur tanks exceeding PFA safe limits of 1.5 mg/kg. This is likely due to the high concentration of Cd in the tank waters, long term use of tank waters for vegetable cultivation and high content of Cd in the soils. Leafy vegetables, amaranthus and palak, accumulated maximum levels of Cd followed by root vegetables carrot and radish. This may be due to genotypic variations in different species to absorb or translocate toxic metals (Patterson 1977). The Cd content of vegetables in Varthur and Bellandur was 6-16 and 6-20 folds higher than that of Cd content of vegetables in the uncontaminated site. The mean Pb concentration of all the vegetables was above PFA safe limit of 2.5ppm irrespective of the sites from where they were collected though vegetables grown near Bellandur tank and Varthur tank accumulated higher concentrations of Pb compared to vegetables grown near the other two tanks (Figure 2). The elevated levels of Pb in vegetables near Varthur and Bellandur may be attributed to long term use of tank waters, high levels of Pb in these tanks, nearness of the fields to highways and atmospheric deposition. The highest concentration of Cr was found in vegetables grown near Byramangala tank. Except tomato and beans all the leafy and root vegetables contained Cr levels exceeding PFA safe limit of 20 mg/kg (Figure 3).

The effluents discharged into the Byramangala tank from the surrounding industrial units and long term use of the tank waters for vegetable cultivation might be the cause of higher levels of Cr in vegetables near Byramangala tank. Cr concentration of vegetables near this tank was about 5 folds higher than that of uncontaminated site. The mean Ni concentration of all the vegetables grown near all the tanks exceeded PFA safe limit of 1.5mg/kg for human consumption though the levels in vegetables near Bellandur tank were higher compared to that of other tank areas (Figure 4). The mean concentration was higher in leafy vegetables amaranthus and palak followed by root vegetables radish and carrot. The mean Ni concentration in vegetables near Bellandur tank was about 10 folds higher than that of vegetables of uncontaminated site.

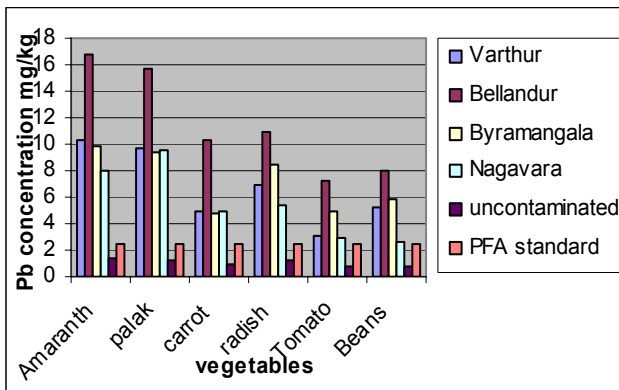


Figure 1. Cd content of vegetables.

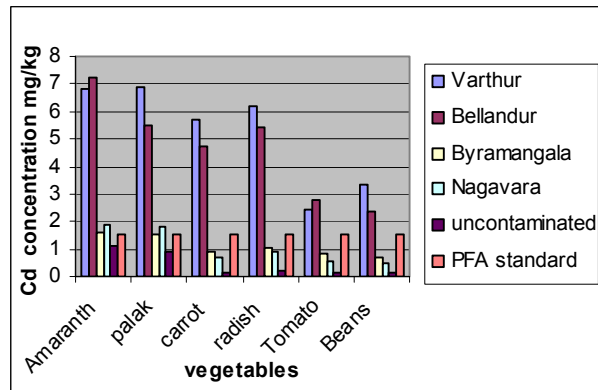


Figure 2. Pb content of vegetables.

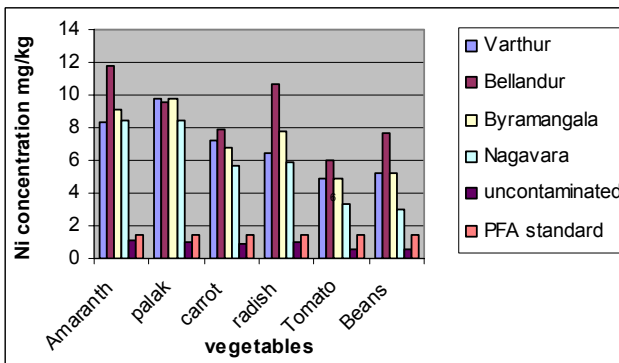


Figure 3. Cr content of vegetables.

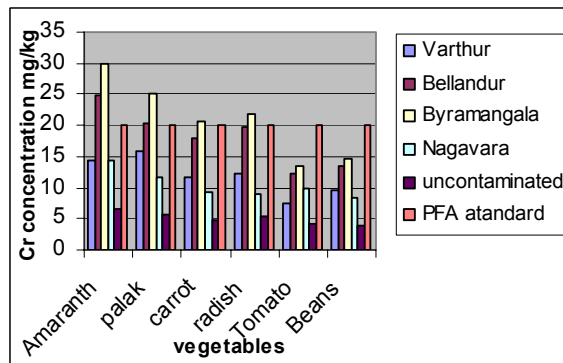


Figure 4. Ni content of vegetables.

Table 1. Heavy Metal Concentrations (mg/L) In Different Water Bodies of Bangalore.

| Location | | Cd | Pb | Cr | Ni |
|---------------------------------|----------|--------|-------|-------|--------|
| Varthur tank | Mean | 0.033 | 0.075 | 0.289 | 0.039 |
| | Std. dev | 0.009 | 0.039 | 0.189 | 0.036 |
| Bellandur tank | Mean | 0.039 | 0.065 | 0.291 | 0.042 |
| | Std. dev | 0.008 | 0.025 | 0.198 | 0.033 |
| Byramangala tank | Mean | 0.022 | 0.059 | 0.311 | 0.04 |
| | Std. dev | 0.011 | 0.024 | 0.215 | 0.03 |
| Nagavara tank | Mean | 0.014 | 0.039 | 0.12 | 0.027 |
| | Std. dev | 0.002 | 0.016 | 0.067 | 0.018 |
| Borewell of Uncontaminated site | Mean | 0.002 | BDL | 0.015 | 0.0016 |
| | Std. dev | 0.0008 | - | 0.007 | 0.0008 |
| Safe limit* | | 0.01 | 0.5 | 0.1 | 0.2 |

*Source: Pescod, 1992. n: number of samples

Table 2. Heavy Metal Concentrations (mg/kg) In Soils Receiving Sewage Water From Different Water Bodies in Bangalore.

| Location | | Cd | Pb | Cr | Ni |
|-----------------------|----------|---------|--------|---------|-------|
| Near Varthur tank | Mean | 2.9 | 68.12 | 56.5 | 57.3 |
| | Std. dev | 0.7 | 18.7 | 14.64 | 19.9 |
| Near Bellandur tank | Mean | 2.38 | 64.9 | 51.8 | 45.7 |
| | Std. dev | 0.67 | 12.5 | 13.4 | 14.5 |
| Near Byramangala tank | Mean | 2.06 | 55.02 | 92.78 | 46.1 |
| | Std. dev | 0.71 | 7.67 | 14.05 | 79 |
| Near Nagavara tank | Mean | 1.92 | 47.04 | 35.08 | 48.2 |
| | Std. dev | 0.25 | 7.67 | 7.83 | 9.09 |
| Uncontaminated field | Mean | 0.9 | 39.6 | 34.2 | 34.9 |
| | Std. dev | 0.22 | 7.47 | 6.31 | 8 |
| Safe limit* | | 1.6-3.0 | 90-300 | 100-120 | 48-75 |

*Source; Kabata and Pendias (1984) n: number of samples

Conclusion

It can be concluded from the above study that the waters of 4 major water bodies of Bangalore were contaminated with heavy metals especially Cd and Cr. Though the levels of heavy metals in the soils under study were within safe limits as per the standards, the levels in vegetables exceeded official Indian standards (Awasthi 2000) by many folds. Leafy vegetables accumulated highest concentrations followed by root and fruit vegetables. Vegetables grown with waters of Varthur and Bellandur tank accumulated higher concentrations of Cd, Pb and Ni whereas vegetables grown with waters of Byramangala tank accumulated very high levels of Cr. The high levels of heavy metals in tank waters, soils and vegetables can be attributed to discharge of municipal and industrial waste waters into the water bodies.

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Hydrological and land use control on N export sensitivity to climate in three adjacent watersheds

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Abstract

Although fluctuation in air temperature can cause changes in snowpack depth in winter and stream discharge in summer, the response of N export to the climate is poorly understood. We investigated the response of N concentrations to hydrological process and climatic conditions in three adjacent watersheds with different land use. Hydrological events contributed to 58-64% of the annual discharge and 62-78% of the annual TN loading in three headwater streams, and wet years tended to trigger larger discharge and N export. Stepwise multiple regression revealed that stream discharge was the best predictor of N concentrations, but variability in stream N concentrations also corresponded to changes in air temperature and snowpack depth throughout the winter, and were sensitive to precipitation in the summer. The agricultural watershed contributed more N loadings and higher concentrations than the forested watershed, but the mole ratios of Si to TN were much lower in the agricultural watershed and most of them were below the threshold value (2.7) for eutrophication during hydrological events, posing a high threat to coastal water.

Key Words

Nitrogen export; stream discharge; climate; land use; watershed.

Introduction

Nitrogen (N) export from a watershed at temporal and spatial scales has been widely reported and an understanding of N export is improved, for example, the well-known hydrological flushing of N, especially NO_3^- -N, during the snowmelt season and storm events (Creed and Band 1998; Zhang *et al.* 2007). Long-term studies on Adirondack forested watershed showed that temperature fluctuation in winter played a key role in interannual variation in the export of NO_3^- -N (Park *et al.* 2003) and DOC (Park *et al.* 2005). Therefore, an interaction between temperature and streamflow might be an important regulator for the export of N. However, the impact of climatic conditions responsible for affecting the snowpack depth and streamflow is usually neglected and poorly understood.

The Shibetsu area in northern Japan has a hemi-boreal climate, characterized by warm summers and cold winters with a substantial portion of snow-covered period. Our previous studies found that there was a significant positive correlation between NO_3^- -N concentrations and the proportion of upland (Hayakawa *et al.* 2006) and that N export was controlled by hydrologic processes during storm events (Jiang *et al.* 2009, under review). Therefore, we investigated the response of N concentrations to climatic variation in three watersheds with different land use to further understanding the coupling impact of climatic variation and hydrological processes on N export at a spatial scale.

Methods

Watershed description

Three headwater stream watersheds adjacent to each other (agriculture-dominated watershed, AW; forest-dominated watershed, FW; and mixed agriculture-forested watershed, AFW) were selected for this study, which are located in the Shibetsu watershed in eastern of Hokkaido, Japan. The characteristics of watersheds are given in Table 1.

Watershed monitoring, sampling, and analysis

Base flow water samples were grabbed from the stream outlet in 1 L polypropylene bottles once a month from March to November during 2003-2005. Automated water samplers (ISCOTM 3700) were installed at the outlet of watersheds, and water samples were collected during storm events and snowmelt season. Daily stream water level at every 15 min was recorded using a water sensor and a logger. Meteorological data was obtained from Japan Meteorological Agency (<http://www.jma.go.jp>). Chemical analysis including total N

(TN), total dissolved N (TDN), NO_3^- -N, NH_4^+ -N, DOC, SO_4^{2-} , Si, and basic cations were analyzed at the laboratory following the standard methods.

Table 1. Location, watershed characteristics, and discharge statistics.

| Watershed | Site coordinates | | Area km ² | Land use % | | Discharge (2003-2005) m ³ /s | | |
|-----------|------------------|----------|----------------------|-------------|--------|---|---------|---------|
| | | | | Agriculture | Forest | Mean | Maximum | Minimum |
| AW | 43.544N | 144.865E | 9.3 | 90.3 | 9.3 | 0.24 | 2.58 | 0.11 |
| AFW | 43.525N | 144.833E | 28.3 | 52.2 | 45.9 | 0.99 | 50.40 | 0.11 |
| FW | 43.540N | 144.762E | 76.0 | 14.7 | 85.1 | 2.63 | 23.34 | 0.84 |

Results

Climatic and hydrologic conditions

Stream discharge of the three watersheds all peaked sharply during storm events, and quickly responded to the increase in mean daily air temperature at the beginning of the snowmelt season (Figure is not shown). All stream discharge values showed a strong positive correlation with precipitation, and an inverse correlation with snowfall and snowpack depth in a whole period.

Nitrogen concentration

A significant difference in discharge and concentrations of N species was found among years and watersheds. The largest discharge and N concentrations were observed in wet year (2003). The forested watershed FW had the largest discharge with the lowest N concentration. By contrast, the agricultural watershed AW had the lowest discharge with the largest N concentration. NO_3^- -N was the dominant N species for all the watersheds.

Response of N concentrations to climatic condition and discharge

N concentrations in headwater streams generally showed a synchronized response to runoff and precipitation during the study period. In winter months, N concentrations displayed a similar trend with the fluctuations of temperature, increasing significantly on the several consecutive days with above-freezing temperatures and increasing runoff at the beginning of snowmelt events (Figure 1, only watershed AW is shown, the other two are similar). To better understand the relationship, we separated the results for winter and summer. A positive correlation of N concentrations with runoff and temperature, and a negative correlation with snowpack depth were found in each watershed during the winter (table is not shown). Stepwise multiple regression selected discharge, temperature, and snowpack depth as significant predictors of N concentrations in each watershed (Table 2). In the summer, N concentrations showed a positive correlation with discharge in all watersheds, and a positive correlation with precipitation was found in watershed AW and AFW, but positive correlation with temperature for watershed FW (table is not shown). Overall, discharge was found to be the best predictor of N concentrations (Table 2).

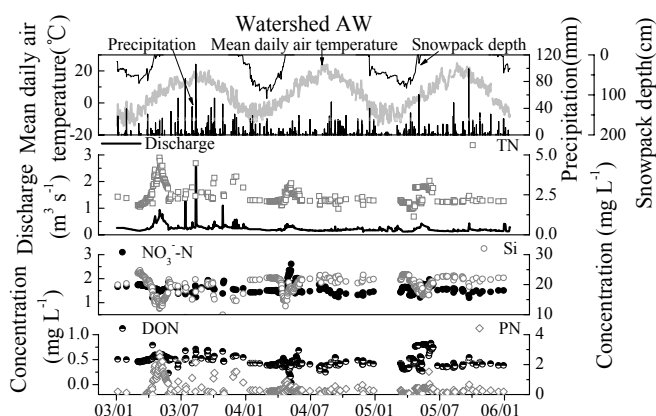


Figure 1. Response of N species concentrations to discharge and climatic conditions.

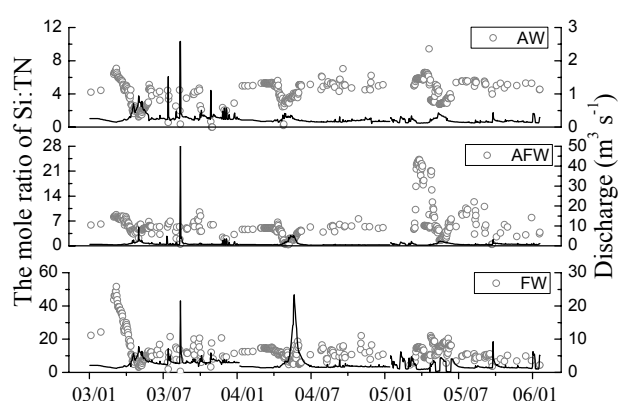


Figure 2. The mole ratio of Si to TN in three headwater stream watersheds.

Table 2. Results of stepwise multiple regression between N concentrations and climatic and hydrological variables. Asterisks indicate significant difference (*p<0.001; **p<0.01; *p<0.05). Q: discharge; SD: snowpack depth; T: mean daily air temperature; P: precipitation; SLD: sunlight duration.**

| | | Winter | Summer |
|-----|---------------------------------|---------------------------------------|---|
| AW | TN | Y=1.650+3.23Q***-0.002SD***+0.007T*** | Y=1.636+2.906Q*** |
| | TDN | Y=2.197-0.003SD***+0.003T*** | Y=1.984+0.363Q***-0.004P***-0.004SLD*-0.002T* |
| | NO ₃ ⁻ -N | Y=1.786-0.003SD***-0.364Q*** | Y=1.541-0.003P***+0.11Q*** |
| | NH ₄ ⁺ -N | Y=0.027+0.001T***-0.037Q*** | Y=0.01-0.00005T* |
| | DON | Y=0.417+0.319Q***-0.0008SD*** | Y=0.417+0.23Q***-0.003T*** |
| | PN | Y=-0.58932+3.384Q***+0.001SD** | Y=-0.317+2.691Q*** |
| AFW | TN | Y=1.314+1.498Q***-0.010SD* | Y=1.439+0.041P***+0.053Q***+0.028SLD* |
| | TDN | Y=1.210+0.193Q***-0.012** | Y=1.289+0.016P***+0.013SLD*-0.022Q* |
| | NO ₃ ⁻ -N | Y=0.982+0.075Q***-0.008T*** | Y=1.079-0.018Q*** |
| | NH ₄ ⁺ -N | Y=0.025+0.001T*** | - |
| | DON | Y=0.196+0.124Q***-0.006T* | Y=0.219+0.015P***+0.01SLD* |
| | PN | Y=0.093+1.327Q***-0.011SD* | Y=0.151+0.025P***+0.075Q***+0.015SLD* |
| FW | TN | Y=0.165+0.129Q***+0.004SD***-0.006T* | Y=0.23+0.144Q***+0.01T*** |
| | TDN | Y=0.212+0.055Q***+0.003SD***-0.004T* | Y=0.364+0.022Q***+0.004T*** |
| | NO ₃ ⁻ -N | Y=0.084+0.002SD+0.045Q-0.006T | Y=0.332-0.02Q***+0.004T*** |
| | NH ₄ ⁺ -N | Y=0.023+0.001T***-0.001Q*** | Y=0.019-0.002Q*** |
| | DON | Y=0.125+0.008Q** | Y=0.014+0.044Q*** |
| | PN | Y=-0.044+0.072Q***+0.001SD** | Y=-0.134+0.123Q***+0.006T** |

Nitrogen loading

The annual discharge and N loading were much larger in 2003 than that in 2004 and 2005 for all watersheds (Figure is not shown). The forested watershed FW showed the lowest N export. Hydrological events contributed the most of annual loading, of which TN accounted for 62, 78, and 62% of annual loading with 58, 64, and 62% of annual discharge in watersheds AW, AFW, and FW, respectively.

Implication of N export to surface water

A strong positive correlation between NO₃⁻-N and SO₄²⁻ concentrations was found for all watersheds and all periods (table is not shown), indicating that higher N export during hydrologic events also led to higher SO₄²⁻ export, which might enhance the potential for acidification and pose a high threat to coastal water. The mole ratio of Si:TN in river water is a useful predictor for eutrophication, and the value below 2.7 is considered as the threshold. Our results showed that the mole ratios of Si:TN were below 2.7 during the snowmelt season and rain events in watersheds with high agricultural proportion (AW and AFW), posing a potential hazard of eutrophication to water bodies (Figure 2).

Conclusion

Hydrological events had control over N export in the three headwater streams, and wet year tended to trigger larger discharge and N export. Discharge was the best predictor of N concentrations; however, stream N concentrations were also sensitive to temperature, precipitation, and snowpack depth. Agricultural watershed exhibited remarkably high N loadings to streams and higher concentrations than that of forested watershed, and the mole ratio of Si to TN was below the threshold value for eutrophication during hydrological events, posing a high threat to coastal water.

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Indices of the status of freshwater resources for impact analyses

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Abstract

The sustainability of fresh-water use is increasingly becoming a topic of global concern. Adequate clean water is a fundamental necessity for human and ecological health. Indicators can be used to evaluate the vulnerability of freshwater systems. This paper reviews three freshwater indicators and their advantages and limitations. Simple indicators like the Falkenmark Index and the Water Scarcity Index are easy to use, but fail to delineate accurately a comprehensive picture of freshwater availability. In contrast, the Water Poverty Index covers wider aspects of freshwater availability and use. However, its complexity limits discrimination in international comparisons. The utility of indices depend on how well they assess the water demand and supply of a country. It is understood that it is important to account for the water transferred due to traded goods. None of the indices considered here take this into account. The water footprint concept is based on consumption of water use which helps accounts for water traded in virtual form. The sustainability of water use depends upon the impact of water use and this varies spatially and temporally. The water footprint can be referenced to water resource indices for a better understanding of the impact of water use. For international comparisons to be valid they need to take into account the localized impact of water use. According to considered indicators of water availability, New Zealand is well endowed with freshwater resources compare to Australia, USA and Sri Lanka, but its high levels of water use reveal the need for vigilance to protect the country's valuable and renewable water resources.

Key Words

Freshwater, indicators, impact, water footprint.

Introduction

Globally, issues related to the quantity and quality of freshwater are becoming as important as those associated with greenhouse gas emissions (Clothier *et al.* 2009). Water is a complex resource: it cycles in a dynamic fashion between rainfall and irrigation, runoff and drainage, storage in the soil profile plus transpiration and evaporation, all with enormous temporal and spatial variation. In addition, variation in water quantity and quality governs its value to people and ecosystems (Rijsberman 2005). Across the planet, water supplies are unevenly distributed among and within countries. A range of water related indices have been developed to measure, track and evaluate the state and vulnerability of water systems, and to assess the impact of water use. It is important to understand the actual water use in a country and its impact on the sustainability of fresh- water resources. The utility of an indicator depends on how well it captures the use and availability of water in a country. The objective of this paper is to review three freshwater related indicators which vary in complexity and content, and to discuss New Zealand's fresh water status in relation to those indicators.

The Falkenmark Index

The Falkenmark Index is defined as the amount of water available in a country per capita (Gleick *et al.* 2002). Falkenmark *et al.* (1989), as cited by Rijsberman (2005) a threshold value of 17,000 m³ renewable water resources per capita per year is proposed. Countries whose renewable water supplies cannot sustain this figure are said to experience water stress. When supply falls below 1000 m³ per capita per year, a country is said to experience 'water scarcity', and below 500 m³ per capita per year, 'absolute scarcity'. The Falkenmark Index has been widely used (Gleick *et al.* 2002) as it is very straight forward to calculate and easy to interpret. However, this indicator has a number of shortcomings. It assumes that water availability is constant over time and so does not account for the fluctuations in water availability on a seasonal basis. This index also assumes that water availability is evenly distributed within a country, thereby hiding inequalities in regional availability. Finally, simple threshold values do not reflect important variations in the demand for water in different countries due to, for instance, climate, lifestyle, and economic factors.

The Water Scarcity Index

The Water Scarcity Index for a country is calculated as the annual water use expressed as a percentage of the available water (Hoekstra and Hung 2002). In this index, the available water is given by the precipitation falling within the country's borders. According to Hoekstra and Hung (2002), total water use should ideally refer to the sum of 'blue' water (the amount of water withdrawn from ground or surface water) and 'green water' (the amount of water evaporated and transpired from plants that comes from rain water). But for the water scarcity values presented in Table 1, only blue water use has been accounted for. This index is simple and easy to understand, but it also fails to accurately delineate a comprehensive picture of water scarcity. Water withdrawals do not take into account how much of it is 'consumptively' used. Ecological water requirements are not considered; and neither is the quality of the water.

The Water Poverty Index (WPI)

The Water Poverty Index is an attempt to develop a comprehensive index of water well-being (Lawrence *et al.* 2002). Gleick *et al.* (2002) showed various approaches which can be used in calculating the WPI. Lawrence *et al.* (2002) pointed out the link between "water poverty" and "income poverty" and used a holistic approach to calculate WPI. This index clusters around five sub-indices: resources; access; capacity; use; and environmental aspects. The 'resources' component includes internal water resources and external water inflows. The 'access' sub-index includes the population with access to safe water and sanitation, irrigation water for crops, and water for non-agricultural use. 'Capacity' covers socio-economic aspects which includes income to allow purchase of treated water, plus education and health. The 'use' component accounts for domestic, agricultural and non-agricultural water use (Lawrence *et al.* 2002). As stated above, the WPI indicator has the advantage of being comprehensive. But it is complex to calculate and is not intuitive. This is nonetheless a common problem with comprehensive indicators. Sullivan (2002) presented ways in which an interdisciplinary approach can be taken to produce an integrated assessment of water stress and scarcity. It illustrates the importance of including ecological water requirements plus geo-referencing the various WPI variables in the calculation (Sullivan 2002).

Any method used to assess the water demand and availability of a country needs to consider the comprehensiveness, complexity and the utility of the various indices. Most of the water availability indicators use total water withdrawal within a country as the water demand. This ignores the water that enters and leaves the country in the virtual form due to the traded goods and services between nations. Virtual water is defined as the volume of water required to produce a commodity or service (Hoekstra and Chapagain 2007). Virtual water content covers three components: blue, green and grey water use. Water footprinting looks at the overall water required to produce a given product or meet a specific demand, whether national or personal. The water footprint of a nation is defined as the total volume of freshwater that is used to produce the goods and services consumed by the people of the nation (Hoekstra and Chapagain 2007), and it will be an inevitable part of a metric of water sustainability.

However, the quantification of water use and trade by water volume alone is only a partial guide to environmental impact and to the sustainability of water use. The economic and the environmental impacts of water use vary from region to region, and season to season as do water supply and demand. Therefore, indices need to take into account the localized impact of water use for the international comparisons to be valid.

Status of fresh water resources in New Zealand

New Zealand is a country with abundant national water resources and a relatively small population density. This is made evident by comparing the values for the Falkenmark and WSI Indices for New Zealand with Australia, USA and Sri Lanka (Table 1).

When considering the water poverty index (WPI), it is evident that there is a loss in discrimination between countries, because of countervailing values of the sub-indices. For example, due to very high domestic consumption of water, New Zealand has a very low value (of just above zero) in the 'use' sub-index of the WPI (Lawrence *et al.* 2002). According to Lawrence *et al.* (2002), New Zealanders used 653 liters of water a day per capita for domestic purposes. This rates as "excess use" and compromises the overall New Zealand value for the WPI.

Table1. New Zealand's water indices compared to some selected countries.

| Indicator | Country and Value | Reference |
|----------------------|---------------------|-------------------------------|
| Falkenmark Index | (Thousands of | (Lawrence <i>et al.</i> 2002) |
| | m ³) | |
| | New Zealand | |
| | Australia | |
| | Sri Lanka | |
| USA | | |
| Water Scarcity Index | New Zealand | (Hoekstra and Hung 2002) |
| | Australia | |
| | Sri Lanka | |
| | USA | |
| | Water Poverty Index | |
| Australia | | |
| Sri Lanka | | |
| USA | | |

Why New Zealand needs to pay attention on water use and productivity?

Given the values for the indicators reported in Table 1, one could argue that there are few risks to the quantity of water resources in New Zealand. But we demonstrate the importance of increasing water productivity especially, agriculture in New Zealand. New Zealand is in the top three in the world list of net virtual water exporters with respect to the livestock products. The net export volume for the period of 1995-1999 was 20.4 Gm³/yr (Chapagain and Hoekstra 2003). The virtual water content of some of New Zealand's livestock products is very high. For example, it is claimed that the milk produced in NZ has a virtual water content of 1909 m³/ton which is more than double that of the world average value of 820 m³/ton (Chapagain and Hoekstra 2003).

International sustainability standards for water are currently being developed. New Zealand, as a major virtual water exporter, will increasingly need to show that its use of water is sustainable. In addition, the labels for promoting sustainable water use and communicating this to consumers are being developed. The stated sustainability of water use might well be reflected in the premium price of the final product. When international crop-trade related virtual water flow is considered, New Zealand shows a 30% virtual water import dependency (Hoekstra and Hung 2002). The major water use in New Zealand is by agriculture and irrigation. Intensification has led to a rise in the amount of irrigation, with a near doubling in the past twenty years (Davie 2009). It is predicted that it could double again in the next 20 years. As fresh water resources are finite, increasing demand will eventually be limited by supply unless significant efficiencies can be found.

Conclusion

Indicators related to water resource availability vary in content and complexity. Simple indicators like the Falkenmark Index and the Water Scarcity Index are easy to use, but they do not provide a clear picture of the fresh-water status of nations. On the other hand, the Water Poverty Index covers wider aspects of freshwater status, including environmental aspects which most of the other indicators fail to consider. However, its complexity limits discrimination in international comparisons. The utility of indices depends on how well they assess water demand and water supply of a country. The water footprint concept brings in a consumption-based indication of water use as it accounts for water traded in virtual form. The sustainability of water use depends upon the impact of water use. For the international comparisons to be valid it needs to take into account the localized impact of water use. According to key indicators, New Zealand is well endowed with fresh water resources. But our high levels of water use show that there is a need for vigilance to protect our valuable and renewable water resources.

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Influence of the pig manure-based liquid fertilizers on the water quality properties in an agricultural catchment with different land uses

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Abstract

A wide diversity of liquid fertilizers and composts produced by the livestock manure in Korea are commonly applied to agricultural lands as alternative chemical fertilizers. However, their effects on the crop production and environmental impacts are unknown. The current study was conducted to understand the effects of the pig manure-based liquid fertilizer on water quality. Cultivated paddy rice and upland plots located in Gyeong-gi province, Korea were treated with two liquid fertilizers, SP (Liquid fertilizer with storage process) and SCB (Liquid fertilizer with slurry composting and bio-filtration process). Plots with no fertilizer (control A) and chemical fertilizer (control B) were also prepared for comparison. Nutrient concentrations in streams were monitored from May through June, which is the dry and early cultivation period in Korea. During this period, there were a lot of nutrient inputs to agricultural lands and this caused the higher concentrations of nutrients in May and June. The nutrient concentrations were dramatically changed during the rainy season, July through August, which resulted from the major problem in streams, such as eutrophication. The losses of N and P to the surface drainage water from paddy rice plots treated with SP and SCB were higher than the ones from the control plots (A and B). In addition, the losses of N and P through the runoff water from upland plots with SP and SCB treatment were higher than for control plots (A and B). The nutrient outflow from paddy rice fields and uplands with application of liquid pig manure was lower than for control plots (A and B). Particularly, the outflow from uplands may directly affect water quality in neighbouring streams. Therefore, it is necessary to establish proper management practices to prevent nutrient losses from agricultural fields and pollution of water environments. The long-term effect of the continuous treatment with manure-based liquid fertilizer will be determined in a successive study.

Key Words

Compost, liquid fertilizer, livestock manure, water quality, SCB.

Introduction

Traditionally, the livestock manure-based composts have been used for agricultural purposes because the composts contain a wide range of nutrients and ameliorate soil properties. In 2008, 42 million tons of livestock manures were generated in Korea, and 84% of the total livestock manure was used for compost and liquid fertilizer productions. In spite of ever-increased usage of livestock manure in the agricultural areas, its environmental impact on neighbouring streams is not well known. Therefore, the present study was conducted to provide understanding of effect of the liquid fertilizer treatment on water quality.

Methods

Experimental site

The experimental plots in paddy rice fields and cultivated uplands were located in GyeongGi province, Korea. Soils for paddy rice field and upland were all sandy loam, and soil pH ranged between 6.1~6.7 and 6.2~6.7, respectively. The organic matter contents were within the range of 34~42 and 24~30 g/kg, respectively.

Treatment

Two pig manure-based liquid fertilizers were applied in this study, which were from a bio-filtration process (SCB) and storage process (SP). They were applied to the separate experimental plots (20x36 m in a paddy rice field and 2.7x10 m in upland). The amount of N required was pre-calculated for the crop needs. For comparison, the experiment plots also included no fertilization plots (control A) and chemical fertilization plots (control B). Following the treatment, paddy rice (*Oriza Sativa* L.) was transplanted into each experimental plot at the spacing 15x30 cm and grown for five months, and corn (*Zea mays* L.) was sown in each experimental plot at the spacing of 20x60 cm and grown for three months.

Sampling and analyses

Water sampling was carried out from paddy rice fields and uplands to monitor the changes of water quality analysis during the cultivation period. All samples were refrigerated at 0 to 4 °C soon after collection until analysis. Water samples were shaken to obtain homogeneous aliquots for N and P analysis. Nitrogen and phosphorus concentrations in the drainage and runoff waters were analysed. The NH₄-N and major anions were analysed using the indophenol-blue method (Bremner and Mulvaney 1982) and an ion chromatography (DX-320, Ion Chromatograph, Dionex Corp.), respectively. Total N was analysed by the standard methods according to the FWPCA manual (USDI 1971), and total P determination was made by the isobutanol extraction method described by Golterman and Glymo (1969).

Results

Table 1 shows the concentrations of N and P in surface drainage waters from paddy rice fields during the cultivation period. The soluble salt (SS) concentrations in water samples were ranged from 0.01 to 0.07 mg/L for all treatments. The concentrations of T-N and T-P in drainage water samples were 1.85~3.88 and 0.18~0.44 mg/L, respectively. Losses of N and P including EC to the drainage waters were higher from SP and SCB treated plots than the ones from controls, such as control A (no fertilization) and control B (chemical fertilization).

Table 1. Water quality properties in the drainage waters from paddy rice fields.

| Treatment | SS (g/L) | EC (dS/m) | NH ₄ -N (mg/L) | NO ₃ -N (mg/L) | T-N (mg/L) | PO ₄ -P (mg/L) | T-P (mg/L) |
|------------------|-------------|--------------|------------------------------|------------------------------|---------------|------------------------------|---------------|
| No fertilization | 0.01±0.01 | 0.07±0.09 | 0.07±0.13 | 1.22±1.01 | 1.85±0.95 | 0.01±0.02 | 0.18±0.07 |
| CF | 0.01±0.02 | 0.09±0.11 | 0.09±0.12 | 2.14±1.75 | 2.28±0.64 | 0.03±0.02 | 0.39±0.03 |
| SP | 0.07±0.05 | 0.16±0.16 | 0.17±0.13 | 3.45±2.20 | 3.88±2.65 | 0.05±0.03 | 0.44±0.20 |
| SCB | 0.03±0.04 | 0.08±0.12 | 0.16±0.15 | 1.64±1.10 | 2.63±2.06 | 0.01±0.01 | 0.21±0.18 |

CF; Chemical fertilization, SP; Liquid fertilizer with storage process, SCB; Liquid fertilizer with slurry composting and bio-filtration process

Table 2 shows the observed concentrations of N and P in runoff waters from uplands during the experiment period. The SS concentrations in water samples ranged from 1.0 to 1.7 mg/L for all treatment. The T-N and T-P concentrations in runoff waters were 6.07~8.45 and 1.39~3.02 mg/L during the rainfall events. Losses of N and P including EC to the runoff waters were higher from SP and SCB treated plots than for the control plots (A and B). The losses of N and P in runoff waters from SP and SCB treated plots were larger than for the control plots (A and B). Particularly, the loss of nitrate-nitrogen was larger than for ammonium-nitrogen, and this can be explained by the nitrification process after liquid manure was applied.

Table 2. Water quality properties in run-off water with application of liquid pig manure at the study upland.

| Treatment | SS (g/L) | EC (dS/m) | NH ₄ -N (mg/L) | NO ₃ -N (mg/L) | T-N (mg/L) | PO ₄ -P (mg/L) | T-P (mg/L) |
|------------------|-------------|--------------|------------------------------|------------------------------|---------------|------------------------------|---------------|
| No fertilization | 1.7±2.0 | 0.03±0.02 | 0.41±0.02 | 1.45±0.52 | 6.07±2.23 | 0.56±0.22 | 1.39±1.36 |
| CF | 1.0±0.7 | 0.04±0.03 | 0.47±0.16 | 1.66±1.11 | 6.15±0.89 | 0.61±0.26 | 2.58±1.25 |
| SP | 1.6±1.0 | 0.10±0.06 | 0.43±0.29 | 2.47±2.21 | 8.45±4.16 | 1.15±0.50 | 3.02±3.06 |
| SCB | 1.0±0.9 | 0.04±0.02 | 0.47±0.32 | 1.84±0.68 | 8.09±3.80 | 0.68±0.32 | 2.82±0.51 |

CF; Chemical fertilization, SP; Liquid fertilizer with storage process, SCB; Liquid fertilizer with slurry composting and bio-filtration process

Figure 1 shows the change of nutrient concentrations in stream waters near the experimental sites. A lot of nutrient inputs were applied to agricultural land, particularly in May through June, which caused the higher concentration of nutrients in the drainage waters. Therefore, the concentrations of COD, T-N and T-P in May through June were higher than the ones in July through August, which is the rainy season. Nutrient concentrations showed the dramatic changes during the rainy season and this caused the major eutrophication problem in neighbouring stream waters.

Conclusion

Losses of N, P and EC to the drainage and runoff water samples from paddy rice fields and uplands treated with SP and SCB were higher than for no fertilizer and chemical fertilizer. The nutrient outflow from paddy rice fields and uplands was lower than for control plots (A and B). The outflow from uplands could directly affect the stream water quality near agricultural fields. The nutrient concentrations in stream waters near the

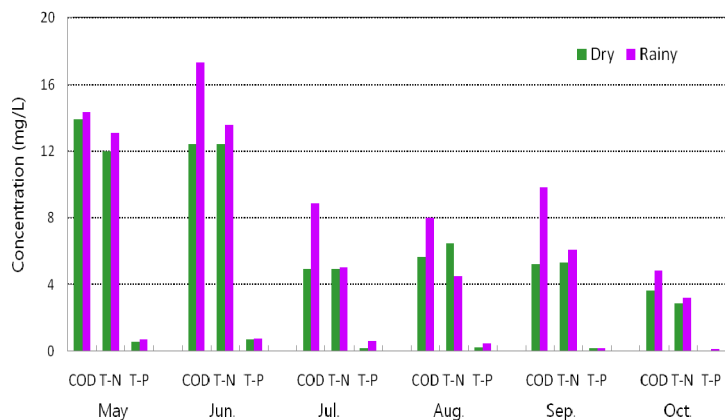


Figure 1. Concentrations of COD, T-N and T-P concentrations in stream waters nearby paddy rice fields and uplands during the crop cultivation period.

experimental sites in May and June (dry and early cultivation period) were higher than in July and August (rainy season). This is because there were a lot of nutrient inputs applied to the agricultural lands. Nutrient concentrations were changed dramatically during rainy season, and it caused the eutrophication problem in streams. Similar results were found in a previous study by Kim *et al.* (1999). Their study indicated that nitrogen and phosphorous concentrations in the drainage waters were relatively high depending on application type and amount of fertilizer during the cultivation period. Therefore, it is necessary to establish the proper management practices to prevent the loss of nutrient from agricultural fields and pollutants entering water. Moreover, it is essential to encourage the utilization of liquid fertilizer with an assurance of water quality. Long-term monitoring is recommended for sustainable water environments in association with continuous application to the agricultural soils.

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Modelling surface and shallow groundwater interactions in an ungauged subtropical coastal catchment using the SWAT model, Elimbah Creek, Southeast Queensland, Australia

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Abstract

The study presented here applies the highly parameterised semi-distributed U.S. Department of Agriculture Soil and Water Assessment Tool (SWAT) to an Australian subtropical catchment for the first time, to our knowledge. SWAT has been applied to numerous catchments worldwide and is considered to be a useful tool that is under ongoing development with contributions coming from different research groups in different parts of the world. In a preliminary run the SWAT model application for the Elimbah Creek catchment has estimated water yield for the catchment and has quantified the different sources. For the modelling period of April 1999 to September 2009 the results show that the main sources of water in Elimbah Creek are total surface runoff and lateral flow (65%). Base-flow contributes 36% to the total runoff. On a seasonal basis modelling results show a shift in the source of water contributing to Elimbah Creek from surface runoff and lateral flow during intense summer storms to base-flow conditions during dry months. Further calibration and validation of these results will confirm that SWAT provides an alternative to Australian water balance models.

Key Words

Water balance modelling, SWAT model, subtropical catchment, stream flow, hydrology, Australia.

Introduction

In Australia, the computation of the water balance has been widely applied to numerous catchments, to: (1) determine the water yield on a daily or monthly time step, (2) model the effects and severity of storm events, and (3) allocate and manage agricultural water resources. The majority of the water balance model codes used in Australia have been developed in Australia and for most of the models, all input data, calculations and outputs are in unit of length per time step, e.g. mm/day, thus representing one-dimensional models (Boughton 2005). This factor requires lumped data input and results in spatially uniform water balances with no distinction between different types of land use, soil, geological, and geomorphological features. The U.S. Department of Agriculture water balance model Soil and Water Assessment Tool (Arnold *et al.* 1998) has been widely used internationally. SWAT has been successfully applied to numerous catchments and climate zones on most continents; however, to our knowledge there are few studies reported in the international literature, which apply SWAT to an Australian catchment. Furthermore, most of the Australian water balance models are primarily applied to the Australian continent (Boughton 2005), which may impede the comparison of Australian catchment modelling with international studies. SWAT simulates a catchment water balance based on the interaction between different physical and climatic parameters; it also considers soil water and shallow groundwater, and may be applied to ungauged catchments. It is a semi-distributed model processing data on a continuous time-step that simulates water and nutrient transport (nitrogen and phosphorus) through a catchment and considers different land uses.

In this case study, we apply the Soil and Water Assessment Tool to a small, ungauged agricultural catchment in subtropical southeast Queensland, Australia to test its potential to simulate model runoff in local subtropical catchments, where no stream flow data is available. Only limited work has been reported for such applications in Australia and most of the Australian model codes did not successfully model runoff in ungauged catchments (Boughton, 2009). This study is part of a larger research project and in this paper we will present only preliminary findings and model outputs without calibration and sensitivity analysis.

Methods

Study Area

The Elimbah Creek catchment is located 80 km north of Brisbane, in southeast Queensland (**Figure 3**). Elimbah Creek and its two tributaries (Six Mile Creek and Beerburrum Creek) drain an area of 142 km² and

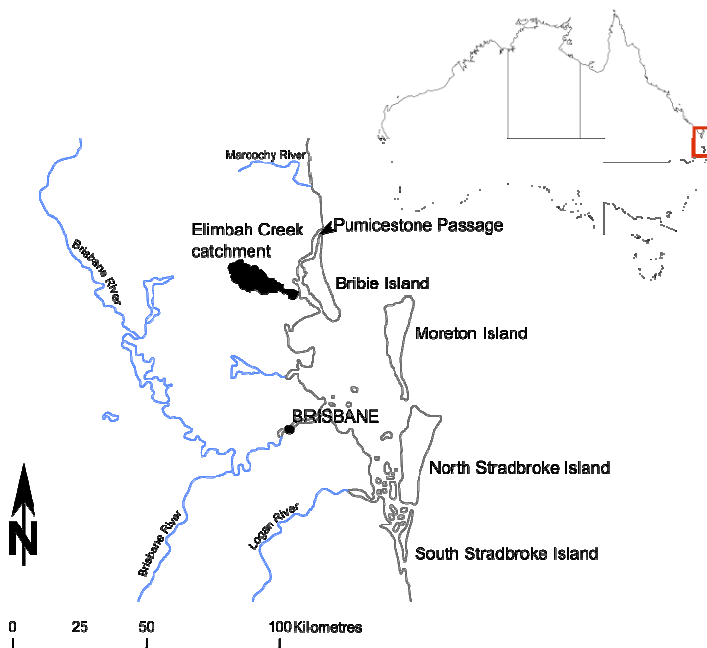


Figure 3. Location of the Elimbah Creek catchment.

form part of the Pumicestone catchment that discharges into Pumicestone Passage. Elimbah Creek is tidal in its lower section, which is common for these coastal catchments with low-gradient coastal plains. These are peri-urban areas with changing land use and the local Moreton Bay Regional Council is expecting almost a doubling in population from the current 139,000 to 235,000 in 2026 (personal communication MBRC, 2008). There are two small rural townships central to the area, Elimbah and Beerburrum, but new residential growth is occurring along waterways and the coastline.

The climate is subtropical, with typically hot, wet summers and cool, drier winters. Thunderstorms and heavy rainfall events are common during the wet season and are often associated with tropical cyclones. The mean minimum–maximum temperature range is 15–26°C. The average annual rainfall is 1450 mm, with the highest average monthly rainfall usually occurring in February (209 mm) and the lowest in September (44 mm), reflecting typical subtropical seasonal variations (recorded period 1958 - 2009). Ground elevations in the study area range from sea level to 5 m above sea level (ASL) along the estuarine sections, and topography in the remainder of the catchment is characterised by an undulating landscape with elevations of up to 50 m ASL. However, outcrops of Landsborough Sandstone bedrock and the volcanic plugs of the Glasshouse Mountains in the upper part of the catchment may reach elevations of 320 m ASL.

Yellow earths and yellow and sandy yellow duplex soils of varying depth cap the central part of the catchment, where plantation forestry and agricultural practices are most common. These earthy soils are highly porous. The dominant soils in the southeast of the study area are bleached sands. They are moderately deep to very deep and have uniform coarse-textured profiles. Detailed information on soil properties of the area is given by Stace *et al.* (1972).

Plantation forestry is by far the most dominant land use in the Elimbah Creek catchment and covers about 45% of the total area. Plantation forestry is concentrated around the north to northwestern part mainly in the Beerburrum subcatchment. Agriculture accounts for 14% with mainly pineapple and some strawberry farms in Six Mile Creek subcatchment; also turf production occurs in the southwest of this subcatchment. Even with this agricultural activity, there is no official information on fertiliser application for the study area (personal communication with MBRC 2009). Animal husbandry covers around 12% of the study area, and a current trend is increasing development of poultry farms especially in the South Eastern part of the catchment. Residential areas, water, and natural vegetation and national parks cover the remainder of the catchment with 4%, 3% and 22%, respectively.

SWAT principles

For this case study, the SWAT2005 model code was used. The Soil and Water Assessment Tool is a semi-

distributed, continuous time, highly parameterised hydrological water balance model (Arnold *et al.* 1998). The model was developed to calculate runoff, sediment, and nitrogen (N) and phosphorus (P) transformations and losses at catchment scale by incorporating the large-scale spatial variability of soil, land use, management practices, and climate. SWAT divides a catchment into subcatchments based on a digital elevation model (DEM) and a user-defined threshold. Each subcatchment may consist of at least one to several hydrologic response units (HRUs). An HRU refers to the total area in a subcatchment with a particular land use, management and soil type. The water balance, sediment, nutrient, and pesticides transformations and losses for each HRU are determined individually, which is different to the Australian model codes. The data is then routed to the associated subcatchment reach and further to the catchment outlet through the channel network. The hydrological processes in SWAT are based on the water balance equation in the soil profile and include precipitation, infiltration, surface runoff, evapotranspiration, lateral flow and percolation. Groundwater is partitioned into a shallow unconfined aquifer, which interacts with the channel system and a deep confined aquifer. Except where pumping occurs, the deeper bedrock aquifer is not connected with the surface hydrological system.

SWAT set-up

Watershed discretisation and data input were performed in ArcMAP (ESRI, 2008) using the ArcSWAT interface (Winchell *et al.* 2007). As Elimbah Creek is tidal in its lower section, the catchment outlet was set further inland to avoid tidal influences. Spatial data input comprised a 25m DEM, land use and soil distribution (Queensland Department of Environment and Resource Management 2007). The overlay of land use and soil resulted into 82 HRUs. Climate data, such as daily precipitation, minimum and maximum temperature, windspeed and solar radiation was obtained from the Beerburrum station (Australian Bureau of Meteorology 2009) for the period April 1999 to September 2009. In this study, seven new soil types were incorporated into the model, as SWAT provides detailed soil information based on American soils, only. Soil properties, such as saturated hydraulic conductivity, field capacity, and available water content are based on soil descriptions from Stace *et al.* (1972). Settings for land use were taken from the default SWAT classifications. **Table 2** presents major SWAT parameters, which were calculated based on described input data, and method used.

Table 2. Major SWAT parameters, which have to be modelled during SWAT run, and methods used.

| Parameter | Method |
|------------------------------|--|
| Surface runoff | USDA (1972) |
| Potential evapotranspiration | Hargreaves <i>et al.</i> (1985) |
| Actual evapotranspiration | Ritchie (1972) |
| Relative humidity | Sharpley and Williams (1990) |
| Sediment yield | Modified Universal Soil Loss Equation, Williams (1975) |

Results

The Soil and Water Assessment Tool calculated the water balance for Elimbah Creek catchment for the period April 1999 to September 2009. **Figure 4-A** shows the monthly estimated runoff at the outlet of the catchment, which drains an area of 115 km². Runoff correlates well with actual rainfall with a coefficient of determination (R^2) of 0.8 (**Figure 4-B**).

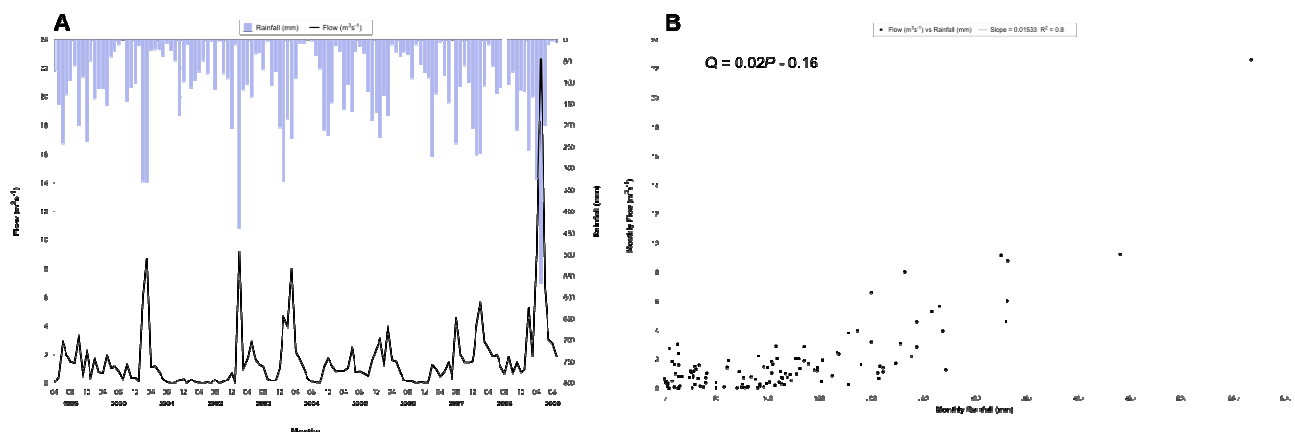


Figure 4. Data for Elimbah Creek catchment outlet, showing, (A) predicted flow (line) and actual precipitation (bars), and (B) corresponding scattergram (B) with R^2 .

For the modelling period, total surface runoff and lateral flow are contributing 65% to the water in Elimbah Creek. Only 36% of the water in the creek is contributed to total base-flow. Around 68% of the rainfall occurring in the catchment is lost through evapotranspiration. The seasonal pattern of the contribution of different water sources show that during the wetter months (January-June) most of the water in Elimbah Creek originates from surface runoff due to high intensity rainfalls. However, during the dry months (July-September) base-flow is the main source of water. Lateral flow contribution is negligible due to the undulating character of the landscape, with no significant steep hill slopes.

Conclusion

In this case study, we applied the Soil and Water Assessment Tool to an ungauged subtropical Australian catchment, which has never been attempted, to our knowledge. As this study is part of a larger research project, only preliminary results are presented here. They show that, with some modifications to its database, SWAT was able to provide a plausible estimation of the water yield in the catchment and to identify the different sources contributing to the total flow. In a similar study in Tunisia, Bouraoui *et al.* (2005) applied SWAT to an agricultural sub-humid catchment to link surface runoff and base-flow to total water yield. The calibrated model showed that SWAT predicted base-flow conditions during dry months and surface runoff into the drainage system during intense summer storms. Further calibration and validation of the Elimbah Creek model is required to confirm that SWAT is indeed a useful tool in catchment hydrological modelling in Australia. Furthermore, the ability to estimate nitrogen, phosphorus and sediment loadings under different management practices in small ungauged catchments makes SWAT a valuable instrument that should be further tested in Australia.

Acknowledgements

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Soil properties as a guide for specific adaptations to climate change

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Abstract

Climate change is impacting eastern Australia particularly in relation to the intensity and frequency of rainfall in some seasons. In a training program to assist adaptation by landowners it was found that some soil properties were critical in the selection of specific response strategies. In 46% of the 430 soils tested the rainfall infiltration rate was sub-optimal, reducing the penetration and increasing the run off from storm events. Specific responses to this limitation were guided by calcium/magnesium ratios, compaction and soil carbon. In 32% of locations sub-soil acidity was limiting root depth and the capacity of pastures to withstand dry periods. Exchangeable aluminium was an important factor in determining the appropriate remedies or adaptations. Soil Carbon was low in 28% of the soils and restricting the available moisture holding capacity. This problem had added importance in light of potential carbon trading and specific responses were influenced by infiltration, acidity, cation exchange capacity, and nutrient status. In all cases the final management recommendations were informed by a landscape key reflecting the potential vulnerability of each location to reduced soil moisture.

Introduction

In eastern Australia there is already evidence of an increasing incidence of hot days in spring and an increasing probability of inadequate rainfall in late winter and early spring. Models of climate change forecast that this trend will continue and that rainfall will be more intense and confined to fewer events (Pierce 2007). Adapting to this change requires that landowners attempt to optimise the amount of available soil moisture resulting from infrequent rain falling more heavily.

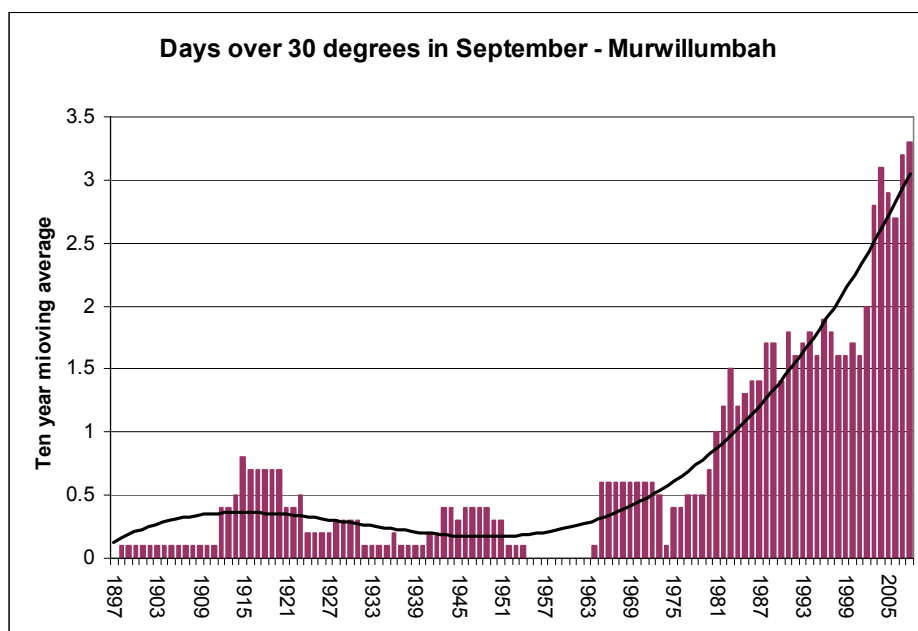


Figure 1. Incidence of hot days in early spring (September) at Murwillumbah in eastern Australia. Sourced from SILO patch point data provided by the Australian Bureau of Meteorology.

The soil moisture available to plants is limited by infiltration rate, root depth, soil texture and soil carbon (Rawls *et al.* 2003). Except for texture, each of these properties can be influenced by particular soil conditions such as compaction, dispersion and sub-soil constraints. These conditions can be diagnosed through related soil properties. Landscape is also influential to available soil moisture. Infiltration rate will be more limiting on steeper gradients where more of the rainfall is shed as run off. Root depth is more critical where the water table is low, for example in deep draining soils or elevated parts of the landscape.

Soil moisture is also affected by evaporation rate which in turn is influenced by aspect and exposure to dry wind quadrants. A training program was designed which used landscape and soil properties to identify limitations to available soil moisture and appropriate remedies or adaptations. The program was delivered to 330 landowners in locations across several climatic zones in the state of New South Wales, Australia.

Methodology

Landowners received training in the observed and projected impacts of climate change in their area. The landowners collected soil samples and performed simple field tests. Laboratory tests were performed on the samples and these results were read in association with the field results. Aerial plans from Google Earth or other mapping services were used to categorise the landscapes within each farm into a “Drying Order” reflecting relative vulnerability to reduced rainfall (Reid 2009). The landscape and soil properties were combined to identify priority limitations of available soil moisture and to recommend the most cost effective remedy or adaptation.

Results

Field infiltration was found to be less than 10mm/min in 46% of the soils tested. In a few cases this was due to temporary saturation of the soil and this was identified by the percentage moisture content. In 55% of cases, poor infiltration was associated with dispersion indicators (Calcium /Magnesium ratio less than 2:1 or Exchangeable Sodium Percentage above 4%). Pasture root development was found to be sub-optimal in 26% of cases. Of these instances 32% were linked with sub-soil acidity, 27% with sub-optimal infiltration and 17% with compaction, the remainder commonly attributable to overgrazing. Soil organic carbon was less than 2% in 28% of the soils tested. Of these cases 51% were associated with a pH (CaCl) of 4.7 or less and 38% of cases with a cation exchange capacity of less than 4 cmol/kg. Low organic carbon was accompanied by poor infiltration in 46% of cases. Through deficiencies in major nutrients were common (72% of tested soils) fertiliser amendments were not a first choice remedy in soils with limitations to available soil moisture (98% of soils tested). Recommendations were instead prioritised on a cost/benefit basis to measures which improved the potential moisture availability or which adapted to limited moisture availability. The participating landowners expressed considerable interest in the possibility of benefiting from soil carbon credits. The property planning process helped landowners to identify areas with the potential to improve soil organic carbon at the lowest cost (Jastrow *et al.* 2006). A survey at completion the training showed that a high proportion of the landowners were intending specific changes in management practices (Table 1).

Table 1. Summary of some workshop exit survey results from training workshops aimed at improving potential available soil moisture.

| Responses to : As a result of this workshop are you planning any of the following changes in your property management? | Percentage of respondents N=172 |
|--|------------------------------------|
| Soil aeration | 59% |
| Changes in paddock layout | 63% |
| Changes in pasture species | 72% |
| Management for Soil Organic Carbon | 88% |
| Windbreaks and shelter belts | 75% |
| Changes in grazing rotation | 61% |
| Management for deeper roots | 85% |
| Application of lime | 69% |

Discussion

Available soil moisture is a primary limitation to growth in Australian grazing systems (Moore *et al.* 1997). In locations where climate change is likely to reduce seasonal rainfall one avenue of adaptation is in measures aimed at optimising available soil moisture (White *et al.* 2003). Priorities in such measures can be guided by related soil properties and landscape factors. An example highlighted in this study was the use of lime to promote root development in acidic soils or to improve infiltration in soils prone to dispersion (Steed *et al.* 1987). Grazing management was another important adaptive response indicated by limitations to soil moisture (Lodge *et al.* 2003).

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Soil properties related to water repellency in volcanic soils at Tenerife (Canary Islands, Spain): relationships with vegetation and soil parent material

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Abstract

Soil water repellency (SWR) was studied in 64 surface soil samples from Tenerife (Canary Islands, Spain), distributed over three different plant communities (natural and afforested Canarian pine forest, and high mountain legume scrub) and three types of soil parent materials (basaltic and salic rocks and basaltic tephra). Soils were studied for physicochemical properties (soil pH and salinity, nutrient contents, organic C and total N, particle size fractions and parameters related to andic soil properties), as well as for common SWR tests (Water Drop Penetration Time and Molarity of the Ethanol Droplet). The results suggest that SWR is more intense in silty-textured soils, with relatively high contents in organic C and nutrients, as well as in Al-humus complexes, under afforested pine forest. Soil parent material was shown to be less decisive in this sense.

Key Words

Soil water repellency, volcanic soils, andic soil properties, pine forest, broom scrubland, soil parent material.

Introduction

Soil water repellency (SWR) has recently become a first-order research issue for soil scientists (Dekker *et al.* 2005). Concerns about this soil property are currently enhanced by the threat of global warming and the increasing impact of wildfires, very especially in areas under mediterranean-type climate. Studies on SWR comprise a wide variety of locations and soil types. However, soils derived from volcanic parent materials have received comparatively less attention in this sense (DeBano 2000). It is generally accepted that SWR relates basically to soil organic matter content and features and tends to be more intense in coarse-textured soils (Del Moral *et al.* 2005). Soils in volcanic areas are rich in short-range ordered minerals (allophane-like silicates) and organometallic complexes that accumulate in the upper soil layers, which stabilize soil humus, thus giving rise to dark, porous and well structured surface horizons. Both the structure and the accumulation of humus may influence water dynamics in these particular soils.

It is also known that SWR varies in space (Regalado and Ritter 2006) and in time, according to the climatic conditions, being usually highest along the dry season. In western Canary Islands, a volcanic archipelago, the main water supplies come from rainfall, especially along the forest belt lying between 800 and 1600 m in altitude, consisting mostly of moist (laurel, lower height) and xeric (pine, upper heights) forests. Therefore, SWR is particularly relevant in these ecosystems. Its knowledge could provide a basis for research efforts in similar ecosystems elsewhere.

In this work, we study the water repellency in a set of soil samples collected in the xeric, Canarian pine forest (natural and afforested) in Tenerife, the greater island in the archipelago. Additional samples have been collected in the high mountain scrubland (above 2200 m in height) for comparison purposes. Soil water repellency will be related to relevant soil physicochemical properties, including those related to the andic reactivity (i.e., active Al, Fe and Si contents, and P retention capacity).

Methods

Sampling tasks

Sampling areas were selected from a previous GIS study, taking into account the following factors: plant community (natural or newly forested Canarian pine forest, and broom scrubland), soil parent material (the most representative rock types are basaltic and salic -trachytic and phonolitic- rocks and basaltic tephra), accessibility (roads or pathways nearby) and absence of wildfires in the last fifteen years, as wildfires are known to increase SWR, either directly or indirectly, often for a long time (Shakesby and Doerr 2006). After this, 64 sampling locations were selected (Figure 1). We have distinguished between natural and afforested pine forest because the first one is open enough to allow the establishment of an understorey (scrubs + herbs) plant stratum, whereas in the second one the canopy is denser, thus limiting biodiversity, but also increasing litter supplies to the soil, and therefore increasing organic matter content in soil surface layers.

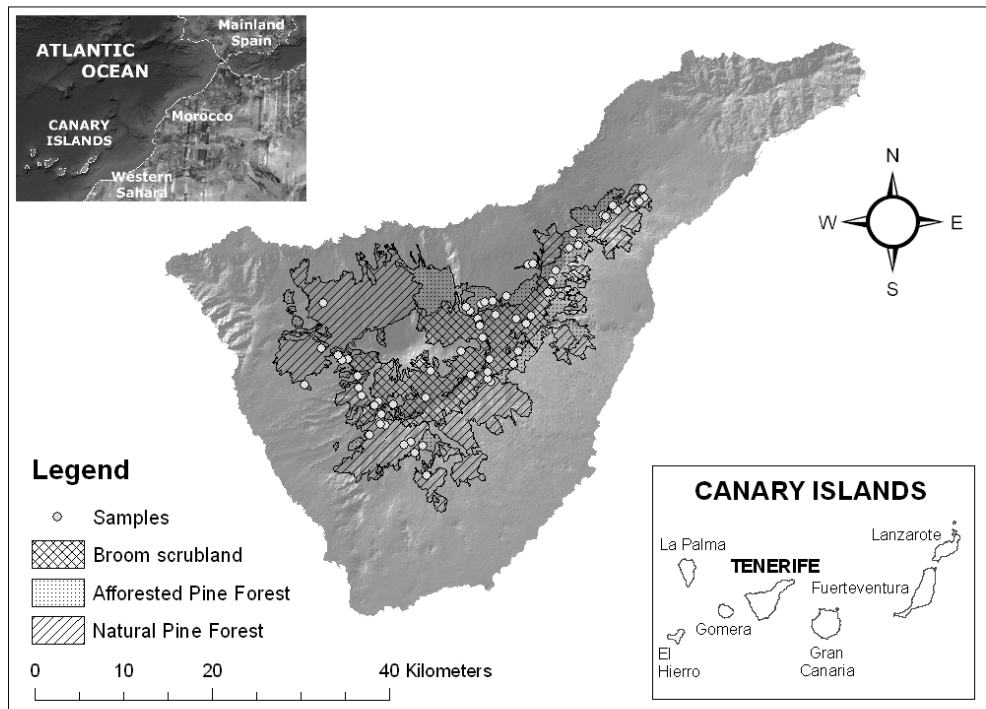


Figure 1. Geographical location of the Canary Islands (upper left and lower right), sampling areas and location Tenerife island.

Analytical procedures

In the field, sampling points were located with a Garmin eTrex GPS device. Surface (0-10 cm) soil samples were collected, air-dried and passed through a 2 mm sieve. In them, the following parameters were determined: pH and electrical conductivity (1:5 soil:water extract), available cations (neutral 1N ammonium acetate), and P (Olsen's method), total organic C (potassium dichromate wet oxidation), total N (Kjeldahl digestion), active Al, Fe, and Si (acid ammonium oxalate method), complexed Al and Fe (sodium pyrophosphate extraction), particle size fractions (Boyoucos densimeter), complexing organic C (sodium pyrophosphate), and P retention capacity (sorption from a 1000 ppm P solution). Determination techniques used included potentiometry (pH and E.C.), atomic absorption spectrophotometry (Ca, Mg, Al, Fe and Si), flame emission (Na and K), UV-VIS spectrophotometry (available P and P retention capacity), and titration (total and complexed organic C, total N). SWR was determined with the classical Water Drop Penetration Time (WDPT) and the Molarity of the Ethanol Droplet (MED) tests in soil samples previously heated at 105°C and cooled in a desiccator over silica gel (Jaramillo, 2004). The statistical analysis were carried out using the Statistical Package for Social Sciences (SPSSTM) v.17 software package.

Results

There was a great variability in the soil samples for the parameters studied (Table 1), save for pH values (coefficient of variation < 10%). With regard to the environmental factors considered (vegetation and parent material), plant community was associated with significant differences for the means (2-way ANOVA test) of soil nutrients (individual and total available cations), Fe_p , $(Al+0.5Fe)_{ox}$, total and pyrophosphate-extractable C, P retention capacity and particle size fractions. These results evidence that soils under high mountain scrubland are less developed than those under pine forest (either natural or afforested), which are richer in nutrients and organic C forms, and also have silty textures. In terms of soil parent material, only pH, Alox:Alp ratio and silt content were statistically different among sample groups.

An initial determination showed that 31 out of the 64 samples were hydrophobic (WDPT > 5 s). The results for WDPT and MED tests are shown in Figure 2.

Table 1. Descriptive statistics for the physicochemical parameters in the soil samples collected. S.D.: Standard Deviation. EC: electrical conductivity (\square S/cm). Cations and sum of bases in cmol/kg. Oxalate- and pyrophosphate- extractable elements, total C, N, particle size fractions and P retention capacity (PRC) as percentages. Available P as mg/kg. Alpo: Alp:Alox ratio. AlFeo: $\%(\text{Al}+0.5\text{Fe})_{\text{ox}}$.

| | pH | EC | Ca ⁺² | Mg ⁺² | Na ⁺ | K ⁺ | BS | Alo | Feo | Sio | Alp | Fep |
|-------------|------------------|-----|------------------|------------------|-------------------------------|----------------|-------------------|------|------|------|-------|------|
| Mean | 6.7 | 88 | 4.7 | 1.5 | 0.5 | 1.5 | 8.2 | 1.2 | 0.9 | 0.4 | 0.4 | 0.08 |
| S.D. | 0.32 | 45 | 3.8 | 1.6 | 0.4 | 0.9 | 5.9 | 1.1 | 0.6 | 0.4 | 0.3 | 0.06 |
| Max | 7.36 | 230 | 18.4 | 7.9 | 2.8 | 4.6 | 25.3 | 5.1 | 2.9 | 1.9 | 1.5 | 0.23 |
| Min | 5.93 | 30 | 0.2 | 0.1 | 0.03 | 0.8 | 0.8 | 0.1 | 0.2 | 0.02 | 0.0 | 0.01 |
| | Al _{po} | N | OC | Cp | C _p C _T | PRC | AlFe _o | Sand | Silt | Clay | P | |
| Mean | 0.55 | 0.4 | 6.6 | 2.03 | 0.31 | 44.4 | 1.64 | 56.3 | 35.3 | 9.6 | 26.98 | |
| S.D. | 0.59 | 0.3 | 5.4 | 1.86 | 0.11 | 20.9 | 1.26 | 15.1 | 12.4 | 5.1 | 18.09 | |
| Max. | 3.22 | 1.4 | 22.7 | 7.4 | 0.6 | 82.9 | 6.0 | 89.3 | 62.2 | 24.5 | 78.21 | |
| Min. | 0.01 | 0.0 | 0.1 | 0.0 | 0.0 | 5.9 | 0.2 | 21.1 | 8.7 | 1.4 | 1.02 | |

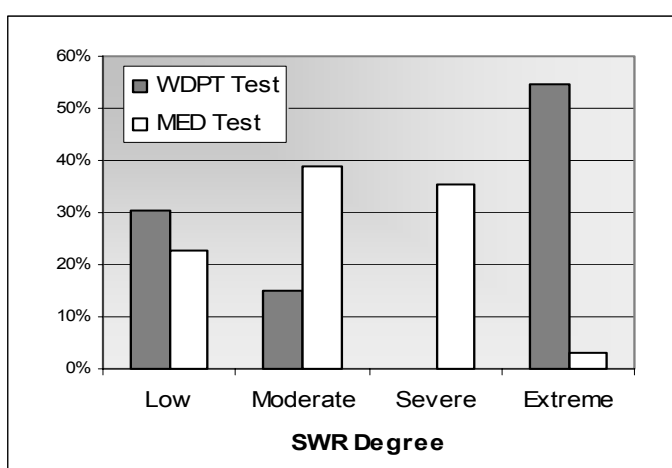


Figure 2. Frequency distribution for water-repellent soil samples. Categories are as follows: for WDPT, Low (5-60 s), Moderate (60-600 s), Severe (600-3600 s), Extreme (≥ 3600 s); for MED, Low (0.2-1M), Moderate (1.2-2M), Severe (2.2-4M), Extreme (≥ 4 M).

There was a tight relationship between both repellency measurements (Spearman correlation model, $R = 0.951$, $p < 0.001$). A comparison of hydrophobic vs. non-repellent samples (Student's t test) showed that repellent samples were richer in nutrients (higher E.C., basic cations and P levels), pyrophosphate-extractable C, Fe and Al, total organic C and N, and clay content.

MED test classes for hydrophobic samples were used to analyse these parameters (Kruskall-Wallis test, due to the small size of the resulting groups), so as to elucidate whether SWR intensity might be related to changes in these properties. To do this, the Severe and Extreme (1 sample) classes were joined into one. The results showed that only total organic C and pyrophosphate-extractable C were useful to discriminate different repellency intensities. A further pair-wise comparison for the three groups (Mann-Whitney's U Test) showed that both Total organic C and Cp were statistically different in samples with the lowest intensity in water repellency. However, no significant distinction could be made for samples with moderate and (severe + extreme) hydrophobicity.

A Principal Components Analysis (Figure 3) was also carried out for the standardized data matrix. Several approaches were necessary, rejecting those variables with low extraction (pH, E.C., Na⁺, %Fe_{ox}, Fe_p, C_p:C_T ratio and %Clay). From the plot of the remainder variables against the two main axes (accounting for 66.3% of the total variance), it can be seen that the first axis is positively associated with the organic C (total and Cp), total N, nutrient, silt and Al contents, and negatively with the sand content. The second axis is positively related to the total active Al and Si contents, as well as to the P retention capacity. Provided that SWR has shown to be related to most of the variables (save for the silt content) positively related to the first axis, it would follow that this factor could also be tightly related to SWR. It also seems reasonable to assume a connection between this axis and the age of the soil, represented by the opposite locations of the silt (more developed soils) and sand (vice versa) contents along it.

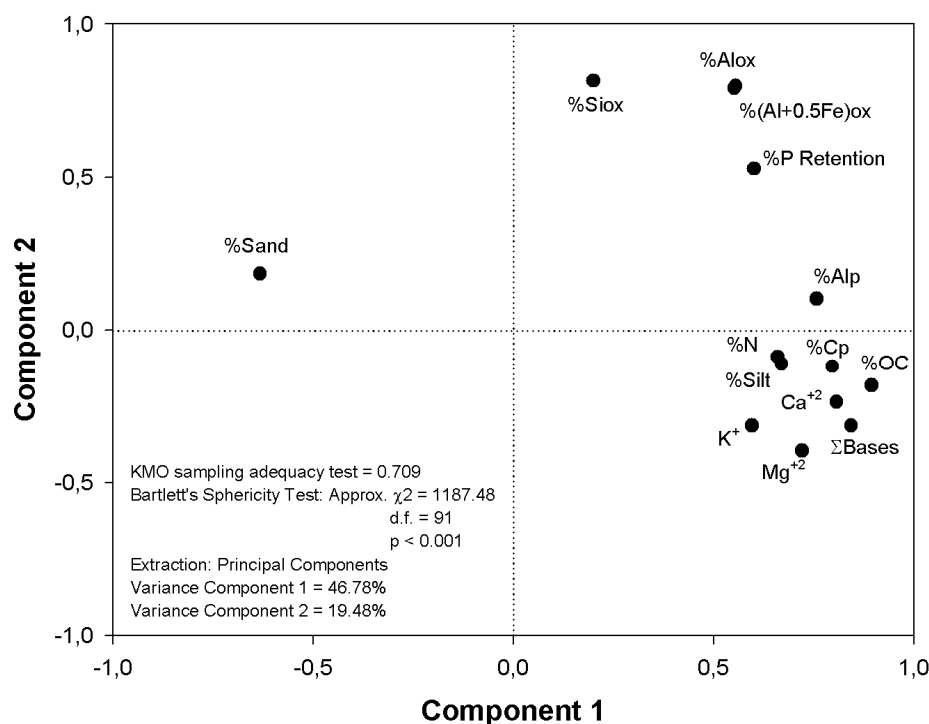


Figure 3. PCA plot for the variables studied, once disregarded those with low extraction (communality).

As far as the second axis is concerned, it seems to be clearly linked to the chemical reactivity of andic soils, whose relationship with SWR could come only from the levels of complexed Al (Al_p), which is part of the total active Al content (Al_{ox} and $[Al+0.5Fe]_{ox}$), and can also contribute to the P retention capacity. The active Si content keeps a low correlation with the first axis, but is also significantly related to the second one. Therefore, SWR can be expected to be more significant in Aluandic, rather than in Silandic Andosols.

Conclusions

Soil water repellency was shown to vary widely among the collected soil samples, although the results suggest that this soil property is more intense and persistent in relatively developed soils, which have high organic C and nutrient contents in the upper horizons which, in terms of vegetation, means: afforested pine forest \geq natural pine forest > broom scrubland. The relative significance of soil parent material has been lower than that of vegetation type. Andic soil properties only relate partially to soil water repellency, especially in soil rich in Al-humus complexes.

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Soils research and education linked to climate change in USDA's National/Agriculture and Food Research Initiative (NRI/AFRI) Programs

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Abstract

Federal funding for soil science research in the United States has changed over the last several decades, shifting from largely agriculture related work funded by the US Department of Agriculture to more interdisciplinary and environment oriented science funded by multiple federal agencies. Now that soils are recognized as a critical component of the global carbon cycle, much research is directed at understanding coupling between carbon cycle and climate change. When USDA's NRI first initiated a program-wide Strategic Issue to address aspects of global and climate change, the reaction from the scientific research community was enthusiastic, and nearly a third of the proposals submitted to the Soil Processes program were climate change related. Although the Strategic Issue concept was later dropped, the theme continued to be prominent in the proposals submitted to the Soil Processes program. In 2009, recognizing the decline in undergraduate soils majors and the increased interest in environmental studies at major universities, the USDA's AFRI prioritized increasing the relevance of soils curricula to climate change by promoting the incorporation of the many sub-disciplines of soil science into climate change related curricula. This paper presents significant highlights of the NRI and AFRI funded soils and global change research, and presents some of the key issues and uncertainties that will influence soil science research and teaching in the future.

Key Words

Soil carbon, climate change, research funding, education.

Introduction

Scientists now recognize the pervasive influence of human activities on all ecosystems and that these global changes will become more pressing in the future. Because soil processes exert a major influence on the terrestrial carbon cycle, as well as on CO₂, methane, nitrous oxide, and other greenhouse gas fluxes from terrestrial ecosystems, and on water availability and quality, they are central to many issues of managed and unmanaged ecosystem responses and feedbacks to climate change. Global climate change was one of three overarching Strategic Issues in the 2003 US Department of Agriculture's National Research Initiative's (NRI) request for applications, and in 2004 the first program was established to address this issue within the NRI as an interagency solicitation on carbon cycle science as it related to climate change. Since then there have been five additional interagency solicitations: in land use and land cover change, land use and invasive species, ecosystems services, and two follow-on carbon cycle calls for proposals. The National Aeronautics and Space Administration (NASA), Environmental Protection Agency (EPA), and Department of Energy (DOE) have been key partner agencies with USDA in these solicitations. Years after NRI and its successor program, the Agriculture and Food Research Initiative (AFRI) abandoned the Global Change program-wide Strategic Issue, applicants and awarded projects of the Soil Processes program continued to include many projects related to soils and climate change. Several of the recipients of research awards from the NRI and AFRI Global Change program were for projects involving soil organic carbon changes, greenhouse gas fluxes to and from soils, and other climate change issues involving soil processes. These small programs have produced some important advances, some of which will be described in this paper. Also presented are suggested soils research needs for informed soil management and adaptations to mitigate climate change and greenhouse gas emissions.

Key highlights of NRI and AFRI climate related soil research

Agricultural Ecosystems: Crop and animal production systems are often assumed to contribute to increases in atmospheric greenhouse gases whereas forests are believed to be carbon sinks. Yet the net effect of any ecosystem on greenhouse gases depends on ecosystem productivity and management as well as on type and intensity of disturbances such as forest fires, soil manipulations, pests and diseases, and these depend in part on soil factors. Stephen Ogle, and co-investigators at Colorado State University, funded by the NRI-NASA Global Change program, addressed uncertainties of carbon fluxes from agricultural lands in the U.S. Mid-

West. Working with the U.S. Natural Resources Conservation Service (NRCS), they developed sampling and analysis protocols for soil carbon assessments and decision support tools for land owners and carbon aggregators. The importance of this project is that it includes new refinements of carbon flux models, by coupling natural and crop vegetation and soil process models, thus increasing confidence in modeling ecosystem carbon changes using the integrated CASA-Century model (Ogle *et al.* 2010). This will allow farmers and carbon project aggregators to use the model in estimates of carbon changes as part of government regulatory programs and for future carbon trading. The integrated model may also be used for governmental reporting of carbon stock changes in agricultural lands to the United Nations Framework Convention on Climate Change. The sampling protocols developed will contribute to improving the NRCS's National Resources Inventory carbon stock change estimates across diverse climate regimes. The USDA soil monitoring network developed from the project's pilot sampling campaign will also be an asset for evaluating future carbon policies as greater demands are placed on soil resources for biofuel feedstocks, and to meet increasing demand for food, forage and fiber production with a growing world population.

Carbon cycling in forest ecosystems: Several recent Soil Processes program grants have led to a network of sites in eastern and western US forests as well as a site in Eastern Europe to study effects of forest litter and root inputs on soil carbon and nitrogen cycling (Crow *et al.* 2008). Another team has looked at soil processes under increased atmospheric CO₂ conditions (Cheng 2008). Both teams of researchers have shown the key role of nitrogen in controlling these processes. Another major finding was that increased carbon inputs as litter or roots, via management or through increased atmospheric CO₂, can have a priming effect on soil carbon decomposition, releasing greater amounts of CO₂, offsetting carbon fixed through photosynthesis or added via management practices. Although the net effect of this priming phenomenon over decadal time scales is uncertain, it is clear that this result and the mixed results regarding old growth forests as carbon sinks, has led to a rethinking of models and estimates for the impact of climate change and increasing CO₂ on organic matter in forest soils. These and other findings supported decisions to incorporate nitrogen as a factor in modeling the effects of increasing CO₂ on ecosystems and their feedbacks to climate change by the IPCC.

In the Western U.S., intense, stand-replacing fires are increasing due to warmer temperatures and longer growing seasons. Thomas Kolb and co-investigators from Northern Arizona University investigated effects of forest management and wildfires on CO₂ and methane fluxes from Ponderosa pine forests. They analyzed effects of prescribed burning and thinning on bark beetle attack, pest resistance, and tree mortality, providing forest managers with science-based information for management strategy decisions to optimize forest growth and carbon sequestration in soil and biomass, and reduce risk of stand-replacing fires that change these forest carbon sinks into carbon sources for over 10 years after the fire (Montes-Helu *et al.* 2009).

Soils and biofuel production: An important greenhouse gas mitigation strategy in agriculture and biofuels production is to reduce fertilizer use via increased efficiency and microbial nitrogen fixation in soils. Researchers at Cornell University showed that soil N-fixation rates by free-living (not symbiotic) microorganisms were correlated with the diversity of the diazotrophic community in soil. In long-term (>30 yr) experimental plots under continuous corn, diazotroph diversity and N-fixation rates were higher in fields where corn biomass was removed at harvest than where it was retained (Hsu and Buckley (2008). This has implications for crop selection and soil management for biofuel production. These findings demonstrate the need for long-term research trials and analysis since short term results of high C:N of biomass such as corn stover would be expected to stimulate N-fixation in soil.

Microbial ecology of soils: Cornell researchers funded by the NRI Soil Processes program recently developed methods for improving the technique of ¹³C and ¹⁵N DNA stable isotope probes (SIPs) (Buckley *et al.* 2008), overcoming previous constraints on the use of ¹⁵N labeled compounds in nucleic acid SIP, with far reaching implications for studying microbial processes in soils and identifying non-culturable microorganisms involved in major environmental processes. They used ¹⁵N₂-SIP of DNA to show that nonculturable free-living atmospheric nitrogen (N₂) fixers in soil can carry out nitrogen fixation *in situ*, and that ¹⁵N-DNA-SIP can be used to gain access to DNA specifically from these organisms. They identified three groups of free living (non-symbiotic) diazotrophs that are actively involved in N₂ fixation and provided evidence for N₂ fixation by previously unknown orders of microorganisms. They then examined their response to experimental manipulation *in situ*, beginning with carbon and energy sources thought to be major constraints on N fixation in soil, and were able to use ¹⁵N₂-DNA-SIP to explore carbon sources used by

specific populations of N₂ fixers under both aerobic and anaerobic atmospheres. Their results demonstrated, for the first time, nitrogen fixation by a specific group of methanotrophs, and that methane stimulates N₂ fixation in soils, suggesting the potential of using different carbon sources to manage this process. These data also explain observations of increased nitrogen concentrations in soils surrounding gas pipeline leaks. These findings link to climate change because methane and nitrous oxide are potent greenhouse gases, and conservation of soil nitrogen is critical to the maintenance and production of crops, pastures rangelands and forests to mitigate rising atmospheric CO₂ and greenhouse gases.

Soils and organic carbon dynamics: Stability and vulnerability of soil organic carbon fractions has become an important issue because soil carbon is the largest pool of terrestrial carbon, estimated to be about 1550 Pg C in the top 100 cm, and 2450 Pg C in the top 200 cm (Batjes 1996). This is three times the total carbon in the atmosphere, so the potential to influence CO₂ climate forcing via greater storage or the release from soils is high. Thus the program is supporting a network (National Soil Carbon Network) and searchable database for a community of researchers on soil carbon distribution and vulnerability. One potentially stable soil carbon form is biochar, a product of incomplete combustion of organic materials that occurs naturally in soils where fires are common. Recent advances in its detection have revealed that it constitutes a significant proportion of the total soil organic carbon in many soils. Biochar has come under scrutiny because it is a byproduct of a bioenergy production method, pyrolysis, and the recalcitrance and environmental effects of this form of biochar is unknown. Johannes Lehmann and coworkers received awards from the Carbon Cycle and Soil Processes programs to study biochar in soils. Early results from these projects suggest a mean residence time in soil of 1000-3000 years for forest fire biochar (Lehmann *et al.* 2008), but estimates vary in the literature and many questions remain about mechanisms for disappearance of this material from soils.

Soil carbon and permafrost: Another key climate change question is the potential impact of the melting of permafrost soils and subsequent release of this thermally stabilized store of carbon. Understanding soil processes controlling the fate of carbon in thawing permafrost is key to predicting impacts and feedbacks of global warming in boreal areas. Preliminary results from a recently funded project (E. Kasishke, University of Maryland, and A.D. McGuire, U.S. Geological Survey) indicate that these areas of the North American boreal forest are becoming weaker sinks or stronger sources primarily due to increased fire. The effects of fire and release of melting permafrost carbon are offset in part by increased plant growth, but questions remain about the balance over decadal timescales and the potential of a priming effect on this soil carbon. A recent article by McGuire and others (2009) reviews the sensitivity of Arctic carbon dynamics to climate change and calls for linking observations of carbon dynamics to the processes that control them, and soil behavior and processes with changing climate is central to that. The potential release of methane hydrates in and beneath permafrost soils is also of interest, as is the effect of temperature rise on methane and nitrous oxide release from wetlands. Soil microbial community capacities to reduce or exacerbate these and other greenhouse gases under a changing climate, and accompanying responses of humans and ecosystems are important and understudied questions that soil scientists are poised to address.

Soil science education: Despite growing interest in soils in the research community, and the relevance of soil science to environmental issues such as climate change, many university soil science departments in the U.S. are being lost, renamed or merged with other disciplines, resulting in less visibility. This is partly due to an overall decline in undergraduate student enrollment in soil science both nationally and internationally. Many professors note that graduate students in soil science often do not have undergraduate soils degrees and thus many lack fundamental coursework required in soils programs. At the same time, earth systems scientists now recognize the major role soils play many environmental issues, and major strides have been made in developing new methods and applying sophisticated and non-traditional methods to the study of soil science.

One way to address these issues is to improve the relevance of soil science courses and curricula to emerging and urgent environmental and socio-economic issues such as global change. Also, there is a particular need to incorporate fundamental soil science education into climate/global change analyses in order to understand the long-and short-term consequences of climate change and evaluate adaptation and mitigation strategies that include land use and soil management. The 2007 US National Research Council recommendations to the U.S. Global Change Research Program include addressing the need of the education community for a climate education framework, tools, and other resources for both formal and informal education. Whether students move on to careers in research, education, extension, business, policy or government, they will need a clear understanding of the links between climate change and soils in order to make responsible, environmentally

sound decisions in the context of changing conditions. In 2009, the AFRI Soil Processes program included a new priority and project type to support soil science education and its linkage to the issue of climate change, resulting in two funded projects: at the University of New Hampshire and Colorado State University. The program at New Hampshire will use a studio teaching approach with activity-based instruction including hands-on exercises, hypothesis driven experiments, and computer-based simulations, visualizations and data gathering exercises. The Summer Soils Institute at Colorado State University involves renowned faculty and focuses on hands-on experience with field and lab techniques, and emphasis on appreciation of critical issues in soil sustainability in the face of global change and societal pressures. This program is a small step, but it is hoped that this emphasis on soils education specifically linked to key societal issues such as climate change can continue and inspire students and faculty across the United States on the importance of soil science.

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The effects of two different biochars on earthworm survival and microbial carbon levels

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Abstract

Biochar is a material created from the thermoconversion of biomass through pyrolysis for the production of bio-energy. The use of biochar as a soil amendment has been proposed as a means to sequester carbon, thus offsetting the release of CO₂. Management strategies for the use of biochar as a soil amendment are still in development, and the effect of adding biochar to soil on soil organisms, in particular earthworms, is virtually unknown. We studied the effect of two different biochars, pine chip biochar and poultry litter biochar, on earthworm growth and survival in incubated mesocosms in two different field soils, as well as the effect of the two biochars and earthworms on soil microbial carbon biomass. The poultry litter char adversely affected earthworm survival, but resulted in higher levels of microbial carbon, especially at the higher rates of application. The pine char had a higher survival rate, and did not show any change in the microbial carbon levels.

Key Words

Eisenia fetida, poultry litter biochar, pine chip biochar, earthworm mortality.

Introduction

The organic matter the Terra Preta soils in the Amazon basin are very stable and have an estimated mean residence time of 250-3280 years (Stevenson 1994). The stable, dark component of this soil is biochar, which is created through pyrolysis, which is thermoconversion of biomass with the exclusion of oxygen which produces syngas, bio-oil, and biochar (Antal and Gronli 2003). This process provides a way to dispose of substances like human waste or poultry litter, in a way that could be beneficial to the environment, providing potential carbon sequestration and food for microbial systems. The potential benefits of biochars have generated an interest in creating industrially created chars for soil application outside of their endogenous environment. There are now studies emerging about the effects of artificial biochars on microbial populations. Steinbeiss *et al.* (2009) found that the fertilizer effect of biochar may be derived from an increased recycling of nutrients in biomass due to stimulation of soil microorganisms. Biochar provides a food source for microbes. This food source and the microbial respiration is all dependent on the material the char is made from. Das *et al.* (2008) found that adding a poultry litter derived char products in the soil accelerated the microbial respiration rates. Steinbeiss *et al.* (2009) tested two different biochars, a yeast derived char and a glucose derived char, and found that biochar type was the most important factor effecting the microbial community. Work is now emerging on the effects of biochar on microbial populations. Earthworms are biological specie that can impact the microbial cycle, yet there are few studies on biochar that measure the combined effects of microbial biomass carbon and earthworm activity on in the presence of chars.

Methods

We studied the effect of two different biochars, pine chip biochar and poultry litter biochar, on earthworm growth and survival in incubated mesocosms. The containers were controlled cylindrical containers with two different field soils, Cecil and Tifton. Both soils were collected in the field and dried. The pine chip biochar and poultry litter biochar were pyrolyzed with a maximum high temperature of 400°C, a holding time of 0.5 hr, and N₂ as a carrier gas. To establish a baseline of soil properties, a set of control samples were analyzed for mineral N, P, K, pH, total C and N, and microbial biomass. Amendment levels of 0 (control), 22.5, 45, 68, and 90 Mg/ha application were tested, respectively equivalent to 0%, 50%, 100%, 150% and 200% of a 45 Mg/ha application. Four replicates were run with each char at every percentage application rate above 0%; a single set of four replicates at 0% biochar was used as a control for both biochars. Mesocosms were covered in parafilm to maintain moisture and avoid earthworm escape, and incubated at 20°C for 28 days.

To each mesocosm, a total of 10 sub-adult *Eisenia fetida* earthworms were added, with the total earthworm weight per replicate recorded at the beginning and end of the experiment. Rather than total weight, average weights by number of surviving worms were used for comparison. Dead earthworms found at the surface of mesocosms during the incubation were removed. An earthworm was judged to be dead if it did not respond to stimulus with a blunt probe. As dead tissue decomposes rapidly in soil, earthworms not found were assumed to have died during the incubation period. Microbial carbon and nitrogen levels were measured using the fumigation-extraction method by Vance *et al* (1987), using 0.5 M K₂SO₄ as an extractant. Another sample is fumigated with Chloroform in a vacuum desiccator for three days. Samples were processed on the IL 550 TDC-TN. The levels of Carbon and Nitrogen in the initial non-fumigated soils were subtracted from the fumigated samples. Statistical analysis was performed for microbial biomass using a four way factorial ANOVA using the GLIMMIX procedure for SAS 9.1.

Results and discussion

Earthworm survival rates

Figure 5: Earthworm Survival Rates

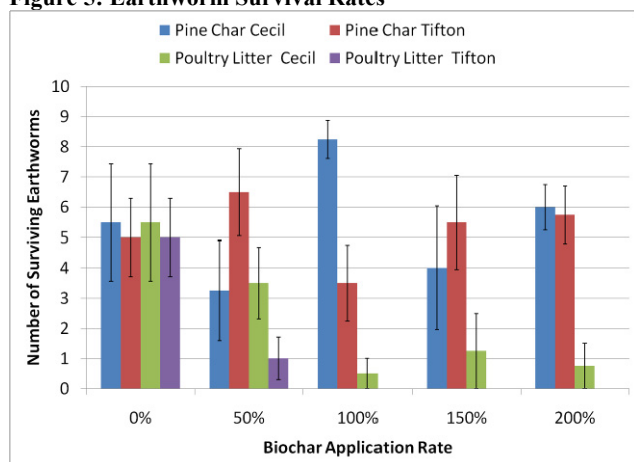


Figure 1. The Pine Chip Biochar survival rates are much higher than the Poultry Litter Biochar rates, and the Cecil soil has higher survival rate. Standard error bars are shown on the graph.

Earthworm survival was dependant on the type of soil and char present. In the Cecil soil, 52% of the total worms survived in the Pine Char, as opposed to only 13% that survived in the Poultry Litter Char. The Tifton Soil had a 47% survival rate for Pine Char, and only a 2% survival for Poultry Litter Char. The number of surviving earthworms is located in Figure 1. The control had a survival rate at 55% and 50% of worms for the Cecil and Tifton respectively. Char type and application rate had a significant interaction ($p = 0.014$). The pine char did not have any significant differences in earthworm survival between the high and low application rates. The Poultry Litter Char had a significant decrease in earthworm survival at all application rates compared to the pine char. At the 50, 100, 150 and 200% application levels, the Poultry litter char are not significantly different from each other, indicating a steep drop in earthworm survival at a low concentration (50%) that persisted in all treatments.

Microbial biomass carbon

There were several significant interactions between the microbial carbon biomass levels and the soil, char type, rate of application and the presence of worms (Table 1). The Cecil soil has a much higher overall microbial biomass than the Tifton soil. There is no significant difference in the microbial biomass in the pine char regardless of the rate of application or the soil. The Cecil soil actually exhibits a net decline in microbial biomass levels with increasing addition of biochar in the Pine Char (Figure 2). There is a significant decline in microbial carbon levels in the Pine Char plots between 0 and 200% and between 50 and 200%. The Tifton soils show no difference in microbial carbon levels for the Pine Char at all treatments.

The presence of poultry litter increased microbial carbon respiration, which agrees with the results of Das *et al* (2008). In the Cecil soil, there was initially a significant decrease in microbial carbon between 0 and 50%, but then there is a steady increase in microbial biomass carbon levels (Figure 2). The Tifton soil poultry litter treatment has a significantly higher carbon level than the pine char at all treatment levels. The microbial carbon increased four times between the 0% treatments to the 200% treatment. This indicates that

there is much more carbon available in the Tifton soils as a result of the addition of char, providing a potential nutrient source to plants and microbes. However, most of the earthworms in the poultry litter char died, which could have contributed to the increase in soil carbon levels.

| Effect | P Value |
|----------------|---------|
| Soil (S) | < .0001 |
| Char Type (CT) | < .0001 |
| S x CT | < .0001 |
| Rate (R) | < .0001 |
| S x R | < .0001 |
| CT x R | < .0001 |
| S x CT x R | 0.0089 |
| Worms (W) | 0.0211 |
| S x W | 0.0034 |
| CT x W | 0.6045 |
| S x CT x W | 0.0222 |
| R x W | 0.0046 |
| S x R x W | 0.0056 |
| CT x R x W | 0.0417 |
| CT x R x S x W | 0.0592 |

When you compare the interaction between soil type, char type, and worms, you find that there is no difference in microbial biomass carbon between the initial, with worm, and without worm treatments for either char. In the Cecil soil pine char treatment, the presence of earthworms was significantly higher than the treatments without earthworms or the initial treatments, which did not differ from each other. This indicates that earthworms were increasing the microbial carbon levels with their activity and respiration. For poultry litter, there is no difference between the presence of worms or not, but the initial measurement was statistically higher in microbial carbon. This could possibly be contributed to a lack of earthworm activity processing and moving the char around the containers.

Figure 6: Microbial Biomass in Two Chars by Rate

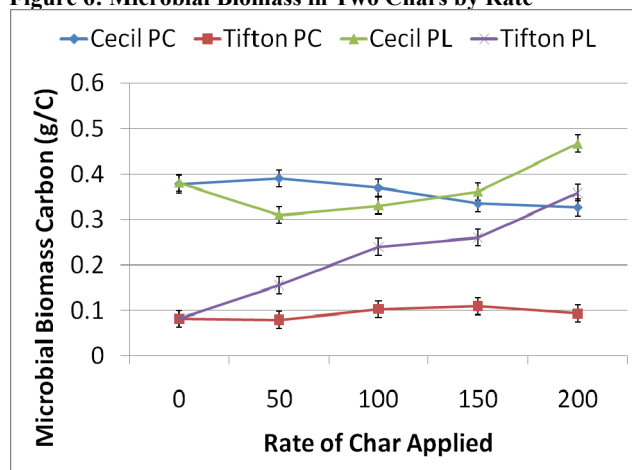


Figure 2. The microbial biomass carbon significantly improved in both Poultry Litter char soils, but remained steady in the Pine Char treatments. Bars are the 95% confidence intervals.

Conclusions

Poultry litter adversely affected the earthworm survival rates in comparison to the Pine Char. The pine char had a stable earthworm survival level that matched the control survival rate. The microbial biomass carbon levels under pine char were also steady in both the Tifton and the Cecil soils, regardless of the rates of char applied or the presence of earthworms. The microbial biomass levels increased in both poultry litter char soils. This may be caused by increased earthworm mortality, or it could provide a better food source for microbes.

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Trend analysis of soil surface temperature in several regions of Iran

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Abstract

The purpose of this study was to assess changes in soil surface temperature over the period 1976-2005. Monthly data of soil surface temperature of 7 synoptic stations of Iran were obtained from Iran Meteorological Organization (IRIMO). These stations represent different climates of the country based on Koppen climatic classification. All seasonal and annual series have been checked for normality with the Kolmogorov-Smirnov test. Time trends of the variable were analyzed using parametric and non-parametric techniques (Least square linear regression, Mann-Kendall, Pearson and rho-Spearman correlation coefficient). Regression analysis method showed no significant trend for all of the time series. The three other tests showed similar results (decreasing or no trend) in different seasonal series. In general, no increasing trend was observed in any of times series in study stations. In spring and annual series the highest percentage, i.e. number of stations with significant trend to total number of study stations, was observed using the Mann-Kendall and Pearson non parametric tests.

Key Words

Soil surface temperature, trend, Iran.

Introduction

The importance of soil temperatures can be seen on many levels. The temperature of the soil controls seed germination, as the soil temperatures must be at an optimum level for a specific crop to grow strong seedlings. Surface temperature, on the other hand controls plant emergence and growth. Soil temperature varies in response to exchange processes that take place primarily through the soil surface. Soil temperature varies from month to month as a function of incident solar radiation, rainfall, seasonal swings in overlying air temperature, local vegetation cover, type of soil, and depth in the soil. The temperature of the soil fluctuates both daily and annually and those changes are most evident at or near the surface where sunlight has the most influence. Several previous studies concerning long-term climatological trends have focused on air temperature or rainfall. For example, Lettenmaier *et al.* (1994) looked for evidence of long-term trends in rainfall, over the continental USA by adopting the Mann-Kendall test and an increase in rainfall during autumn was found in a quarter of the entire stations. Increasing rainfall trends were reported in Argentina (Viglizzo *et al.* 1995), Australia and New Zealand (Suppiah and Hennessy, 1998; Plummer *et al.* 1999). Decreasing rainfall trends were found in the Russian Federation (Gruza *et al.* 1999), Turkey (Türke 1996), and Africa (Hess *et al.*, 1995; Mason, 1996) and in China (Zhai *et al.* 1999). In 19 northern and central European weather stations, Heino *et al.* (1999) found no changes in precipitation extremes. The minimum temperature increased almost everywhere and the maximum and mean temperature increased in northern and central Europe, over the Russian Federation, Canada (Bootsma 1994) and in Australia and New Zealand (Plummer *et al.* 1999). These results support the suggestion of Smit *et al.* (1988) that mid-latitude regions such as the mid-western USA, southern Europe and Asia are becoming warmer and drier, whereas the lower latitudes are becoming warmer and wetter. Marengo and Camargo (2008) studied surface air temperature in southern Brazil. The comparison showed that the frequency of warmer days increased during both summer and winter, especially during the last two decades. Ghahraman (2006) studied the mean annual temperature in Iran and reported both decreasing and increasing trend but in general, most of the stations showed a positive trend. Little works have been done on soil temperature trend analysis in Iran; therefore the aim of this study was to assess the time trend of soil surface temperature in different regions of Iran using parametric and non-parametric tests.

Materials and methods

The weather stations

High-quality weather data for the period of 1976 to 2005 were obtained from the Islamic Republic of Iran Meteorological Organization (IRIMO). These stations represent some of different climates of Iran based on Koppen climatic classification. They were Tabriz (H = 1361m a.s.l.; latitude = 38°05'N; longitude =

46°17'E), Tehran (H = 1190.8m a.s.l.; latitude = 35°41'N; longitude = 51°19'E), and Zahedan (H = 1370m a.s.l.; latitude = 29°28'N; longitude = 60°53'E), Mashhad (H=992m a.s.l.; latitude = 36°16'N; longitude = 59°38'E), Kerman (1753m a.s.l.; latitude = 30°15'N; longitude = 56°58'E), Kermanshah (1318m a.s.l.; latitude = 34°21'N; longitude = 47°09'E) and Shiraz (H=1484m a.s.l.; latitude = 29°32'N; longitude = 52°36'E)



Figure 1. Spatial distribution of study stations

Data analysis

The collected data were carefully examined for missing data but no gap was revealed. Time trends of soil surface temperature were studied for annual and seasonal time series. All time series have been checked for normality with the Kolmogorov-Smirnov test. Least squares linear regression was used to test the increasing or decreasing trends in the study variable. For normally distributed data ($P \leq 0.05$), the statistical significance of the trends was indicated by the Pearson test. In all other cases the non-parametric Mann-Kendall and rho-Spearman test were applied instead.

Results and discussion

The results of trend analysis for different time series are presented in Tables 1 to 5. Also as an example, the results of time trend analysis for Mashhad station is shown in Figure 2.

Table 1. Results of the application of the parametric and non-parametric tests to annual series at 95% level of significance in study stations (1976-2005).

| Station | Method | | | |
|------------|---------|--------------|---------------------|--------------|
| | Pearson | rho Spearman | Regression analysis | Mann-Kendall |
| Shiraz | C | C | C | C |
| Kerman | B | B | C | B |
| Kermanshah | B | B | C | B |
| Mashhad | B | B | C | B |
| Tabriz | B | B | C | B |
| Tehran | C | C | C | C |
| Zahedan | B | B | C | B |

B: Decreasing trend and C: Non significant trend.

Table 2. Results of the application of the parametric and non-parametric tests to winter series at 95% level of significance in study stations (1976-2005).

| Station | Method | | | |
|------------|---------|----------|---------------------|--------------|
| | Pearson | Spearman | Regression analysis | Mann-Kendall |
| Shiraz | C | B | C | B |
| Kerman | C | B | C | B |
| Kermanshah | B | B | C | B |
| Mashhad | C | C | C | C |
| Tabriz | C | C | C | C |
| Tehran | C | C | C | C |
| Zahedan | C | C | C | C |

Table 3. Results of the application of the parametric and non-parametric test to spring series at 95% level of significance in study stations (1976-2005).

| Station | Method | | | |
|------------|---------|----------|---------------------|--------------|
| | Pearson | Spearman | Regression analysis | Mann-Kendall |
| Shiraz | C | C | C | B |
| Kerman | B | B | C | B |
| Kermanshah | B | B | C | B |
| Mashhad | B | B | C | B |
| Tabriz | B | B | C | B |
| Tehran | C | C | C | C |
| Zahedan | C | B | C | C |

Table 4. Results of the application of the parametric and non-parametric test to summer series at 95% level of significance in study stations (1976-2005).

| Station | Method | | | |
|------------|---------|----------|---------------------|--------------|
| | Pearson | Spearman | Regression analysis | Mann-Kendall |
| Shiraz | C | C | C | C |
| Kerman | C | C | C | C |
| Kermanshah | B | B | C | B |
| Mashhad | C | C | C | C |
| Tabriz | B | B | C | B |
| Tehran | C | C | C | C |
| Zahedan | B | B | C | B |

Table 5. Results of the application of the parametric and non-parametric test to autumn series at 95% level of significance in study stations(1976-2005).

| Station | Method | | | |
|------------|---------|----------|---------------------|--------------|
| | Pearson | Spearman | Regression analysis | Mann-Kendall |
| Shiraz | C | C | C | C |
| Kerman | B | C | C | C |
| Kermanshah | B | B | C | B |
| Mashhad | C | C | C | C |
| Tabriz | B | B | C | B |
| Tehran | C | C | C | C |
| Zahedan | B | B | C | B |

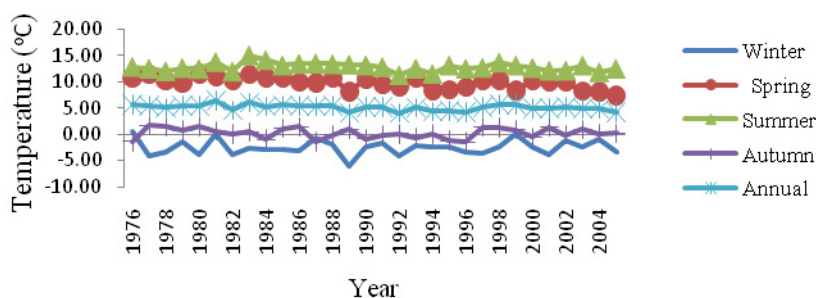


Figure 2. The soil surface temperature variations in Mashhad station.

For all of the time series in study stations parametric method of regression analysis did not show any significant trend. Whereas, the all other three methods showed a similar behavior in trend detection, which means regardless the type of the applied test, existence or non-existence of the trend was same in all stations.

Conclusion

In general, the soil surface temperature in study stations during the last 30 years has not changed significantly or has slightly decreased. This, to some extent, does not coincide with previous studies on air temperature across Iran which indicated a slightly increasing trend in most cases. Therefore, to come to valid conclusion further studies would be required especially on microclimate scales using more station data. None of these three tests showed a superiority to the others.

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