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# COMBATING GLOBAL CLIMATE CHANGE BY COMBATING LAND DEGRADATION

*Proceedings of a Workshop held in*
*Nairobi, Kenya, 4-8 September 1995*

*Sponsored by*

***United Nations Environment Programme (UNEP)***

With support from

**The University of Adelaide, Adelaide, Australia**
**The University of Arizona, Tucson, USA**

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***Victor R. Squires, Edward P. Glenn and Ali T. Ayoub***

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## Acknowledgments

We acknowledge the input from all authors who contributed papers for this Workshop and this volume. We are indebted to the leaders of the Working Groups who toiled so diligently to weigh the evidence and produce their reports. We acknowledge the support of the team at UNEP Headquarters who did so much to make the Workshop possible, especially Ms Sarah Kariuki.

With respect to funding organisations we thank UNEP for its sponsorship of the key speakers and to AusAid for sponsoring Dr Mark Stafford Smith.

The support of the University of Arizona and the University of Adelaide was especially appreciated.

## UNEP Opening Statement

It is my great pleasure to welcome you on behalf of our Executive Director Mrs Elizabeth Dowdeswell to UN Gigiri complex and to UNEP Headquarters and to convey to you all her best wishes for a successful workshop.

The greenhouse effect is not an evil! Without it, life on Earth as we know it could not exist; it keeps the surface of the Earth some 30ºC warmer than it would otherwise be. The concern about the greenhouse effect is of course that the increasing emissions of GHG e.g., carbon dioxide, methane, nitrous oxide and CFCs resulting from human activities may cause a rise in global-mean surface air temperature or induce "global warming".

Both natural (e.g., solar variability, volcanic eruptions) and human activities (e.g., combustion of fossil fuels, agricultural practices and deforestation) can affect climate change by modifying the emissions of greenhouse gases, aerosols and their precursors. The rates of increase in the atmospheric concentrations of carbon dioxide and nitrous oxide have continued to grow or remain steady while those of methane and some halocarbon compounds have slowed.

The potential *impacts* of climate change and global warming are numerous and I do not need to mention them here. Drylands occupy a large part of the Earth's soils, approximately 5.2 billion ha, and support an average net primary productivity of 1200-1500 kg/ha/yr. This represents a large potential sink for carbon, estimated to be about 1.5 - 2 Giga (billion) tonnes of carbon annually.

Furthermore, the potential for increasing the carbon storage in the soils of the drylands is large because of the low organic carbon in these soils. Out of the total carbon stock in tropical grasslands, savannas and woodlands, 80% is in the soil, and 20% is in woody plants. These soils have, therefore, a large potential for either sequestering or releasing of carbon, depending on land management practices.

The United Nations Framework Convention on Climate Change commits all parties to promote sustainable management, and promote and cooperate in the conservation and enhancement, as appropriate, of sinks and reservoirs of all greenhouse gases not controlled by the Montreal Protocol, including biomass, forests and oceans as well as other terrestrial, coastal and marine ecosystems. The recently adopted global Convention to Combat Desertification and Drought asks governments to *link* desertification problems with (preventing) climate change and loss of biodiversity.

Scientific data, including the results of a UNEP Project coordinated by King's College, London, suggest that tropical grasslands are far more productive than previously estimated in the early 1970s, and that the carbon fixed by this increased productivity probably plays a significant role in the global carbon cycle. Recent work on modelling of grasslands and savannas worldwide showed that tropical grasslands (as opposed to temperate or high-latitude grasslands) have the greatest potential for carbon storage under a climate change and elevated $CO_2$ scenario for the next 50-100 years.

Recent studies indicate reported that the deep-rooted grasses introduced to Latin America from African savannas appear to be fixing significant amounts of organic carbon at up to 1 metre soil depth. If this is projected over these large areas sown to introduced pastures in Latin America notably Brazil, it could amount to a significant carbon sink perhaps half a billion tonnes per year, and hence account for a substantial part of the "missing sink" in the estimates of the global carbon dioxide balance.

*The developed countries need low-cost carbon offsets* - *in which* investments in projects that have the effect of sequestering carbon, balances their carbon emissions from fossil fuel burning. Offsets so far have been sought in forested areas rather than drylands. Arid zone developing nations, on the other hand, face problems of soil erosion and land degradation which, if arrested, could result in significant carbon storage in restored ecosystems as well as qualify as carbon offsets. The beneficiaries will thus be both developed countries with their excess fossil fuel emissions and the dryland developing countries with land degradation problems that require outside funding for the restoration and sustainable use of the lands.

This workshop is intended to 1. explore such issues for the benefit of policy-makers of both developed and developing nations, and 2. to arouse the interest of the IPCC experts in the drylands as potential to sequester significant amounts of carbon.

You are invited, therefore, distinguished scientists, to elaborate on the following issues:

1. How much useful soil carbon sequestering could be achieved through implementation of programmes aimed at restoring degraded lands, and which type of programmes are most compatible with carbon storage?
2. To what extent are social goals of land restoration programmes compatible with strategies to store carbon in dryland soils? Are the goals complementary or conflicting in specific cases?
3. How can a direct economic linkage be made between carbon storage in dryland soils and land restoration programmes?
4. Can the policy of "carbon offsets" sought by developed countries become a funding source for programmes aimed at restoring degraded lands needed in developing dryland zone nations? What policy instruments would be needed to create this economic linkage?

A primary goal of the workshop should, therefore, be to reach an expert consensus as to whether or not dryland soils and the ecosystems they support could play a significant role in mitigating global warming, and thereby to *interest modelers working with IPCC in quantifying these potentials.*

I wish you all success in these endeavors, and a pleasant stay in Kenya. Please do not hesitate to make the full benefit of your presence at the UN Gigiri complex and visit our library, GRID, Dryland Ecosystems and Desertification Control, OCA/PAC and other facilities of interest to you.

Dr. Harvey Croze
Assistant Executive Director
Division of Environment Assessment, UNEP Nairobi

## Preface

The UNEP World Atlas of Desertification shows that drylands occupy 6150 million ha (41 % of the world's land area). This is a very large landbase exceeding the area of closed forest (4400 million ha). About 69% of the drylands have been damaged to a greater or lesser degree by human use. Of the 900 million inhabitants of the drylands, 100 million are already impacted by desertification and the remainder are considered at risk. Many are in Africa.

Land degradation is caused by too intensive use of land resources. This abuse of natural resources is not necessarily due to carelessness or ignorance but may represent survival mechanisms under harsh conditions. UNEP has developed strategies for arresting desertification and restoring productivity to these lands, but external funding is required. To the extent that anti-desertification measures result in net, long term carbon sequestration, or reduction in carbon emissions, they may qualify for funding as "carbon offsets" to mitigate environmental effects of fossil fuel burning.

An International Workshop was held at UNEP headquarters in Nairobi to consider land degradation in drylands and its management, particularly as it relates to carbon management as a means to combat global warming.

Twenty-eight specialists from eleven countries met in Nairobi for 5 days in September 1995. The papers presented at the Workshop, and published here, represent a state-of-the-science assessment of dryland degradation, the processes of carbon storage, the feasibility of mitigation and the problems of implementation.

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## EXECUTIVE SUMMARY

An International Workshop was held to consider land degradation in drylands and its management, particularly as it relates to carbon management as a means to combat global warming. Twenty specialists from nine countries met in Nairobi for 5 days in September 1995 to discuss:

- Dryland resources - their extent and condition
- Management options for drylands
- Linking drylands and their management in global climate particularly as related to carbon emissions and storage in drylands; and
- the question of whether burners of fossil fuel can be interested in dryland management/anti-desertification measures would be interested in implementing biotic offset programmes.

### Summary of papers presented

The UNEP World Atlas of Desertification shows that drylands occupy 6150 million ha (41 % of the world's land area). This is a very large landbase exceeding the area of closed forest (4400 million ha). About 69% of the drylands have been damaged to a greater or lesser degree by human use. Of the 900 million inhabitants of the drylands, 100 million are already impacted by desertification and the remainder are considered at risk. Many are in Africa.

Land degradation is caused by too intensive use of land resources. This abuse of natural resources is not necessarily due to carelessness or ignorance but may represent survival mechanisms under harsh conditions. UNEP has developed strategies for arresting desertification and restoring productivity to these lands, but external funding is required. To the extent that anti-desertification measures result in net, long term carbon sequestration, or reduction in carbon emissions, they may qualify for funding as "carbon offsets" to mitigate environmental effects of fossil fuel burning.

World dryland soils store 241 Gt of organic carbon, 50 times more than is added to the atmosphere annually through fossil fuel burning. Small unit area changes in the rate at which carbon is emitted or sequestered in these soils can have relatively large impacts on the atmospheric carbon budget given the large area of drylands. Vigorous efforts to control land degradation in the drylands can result in a net sequestration of up to 1.0 Gt of carbon per year from the atmosphere (see Table in Report on p. 336).

If anti-desertification measures to restore drylands were funded at levels recommended by UNEP(1991) and resulted in say 0.5 Gt of carbon sequestration per year, the costs would be $10-18/tonne of carbon stored.

One option for carbon storage is to manage grazing on the arid rangelands. Storage rates of 4-6 t/ha of C can be achieved over 25 years in 300-600 mm/yr rainfall zones by relaxing grazing intensity from 50% to 10% above ground removal rates. Introducing legume trees into grass pastures can also result in carbon storage at the same time enhancing soil nutrients.

These options are difficult to implement because they reduce short term returns to landholders. Yet they increase long term stability. Examples of dryland systems which have shown increased carbon storage after implementation of anti-desertification measures are found in sub-humid Africa, the Sahel and Australia. Other areas that have great potential in terms of available land area include India, the Russian steppes and the "fertile crescent" of the Near East.

Achieving carbon storage will require careful planning however as not all rehabilitation options increase soil carbon storage. Some desert rangelands may have a finite storage capacity for carbon which cannot be increased by boosting productivity alone. It may also be difficult to boost carbon storage in cultivated dryland soils by no-till farming since straw and stubble rapidly decompose. However, all sustainable yield options positively involve carbon sequestration.

Achieving carbon storage will also require careful consideration of the local economic conditions and must be compatible with the needs and aspirations of local inhabitants. In Africa, for example, opportunities for carbon storage through tree plantings can be complimentary to the need for improved fuel wood supplies.

In Australia by contrast, improvement in rangeland through control of woody weeds may conflict with the goal of carbon storage.

The funding mechanisms for carbon offsets in the drylands and elsewhere are only starting to be developed. In principle, funding can come through government programmes, but it is anticipated that private funding will be the main support for carbon offsets. Major fossil fuel C02 sources are currently supporting carbon offset programmes. For example, the United States electric power industry has embarked on a voluntary programme of managing forest carbon as part of the U.S. plan to return net greenhouse emissions to 1990 levels in the year 2000.

Although the drylands have potential for storing carbon through anti-desertification measures, individual projects must be designed with care to be of interest to private funders or agencies such as the Global Environment Facility (GEF). Carbon storage must be verifiable and stable. Carbon offsets at $5-10/t are available from forested areas and dryland carbon offsets must compete with these to be successful. Nevertheless, there does appear to be a significant potential economic link between problems of desertification and the need to enhance biosphere carbon storage to ameliorate global warming (see Report of Working Groups, this volume).

## Carbon sequestration in the drylands - the rationale

In the world's drylands the major problem is desertification (as defined by UNEP 1992) which is caused by human activities and climatic marginality.

Carbon sequestration, the process of carbon stock protection and aggradation, is being looked at as a viable option by a world increasingly worried about the potential impact of greenhouse gases.

Drylands can help society manage a significant amount of CO2 creatively and relatively expensively. In this context carbon sequestration, is compatible with most techniques adopted to mitigate the effects of desertification through the land cover and improved soil organic matter storage.

In the management of carbon the major options are in the more efficient production and use of energy and better land management.

In the latter instance, sequestration and reduced emissions can achieve significant reductions in the level of atmospheric carbon dioxide, Even though the land areas are large the specific rates of retention of carbon in biomass and soils may be modest.

Programmes of desertification control that provide sustainable land management lead, for example, to increased biodiversity, higher food production, increased woodland products, increased livestock feeds and hazard (floods,soil erosion) reduction. Virtually all of these programmes, if properly managed, can lead to carbon mitigation.

Almost all sustainable projects in drylands, including areas where desertification is not being experienced, will involve reductions in carbon losses to the atmosphere or its subsequent sequestration

There are still areas of uncertainty in the character and extent of the areas, the impact of future climate changes, the dynamics of the processes, and in the spatial and temporal variability of carbon production, the appropriate technology and management structures.

Many of the proposed anti-desertification measures, would increase carbon storage, but also help alleviate the plight of the local inhabitants some of whom are among "the poorest of the poor". Furthermore, existing revegetation projects play a role in carbon storage.

Major conclusions

- The principal conclusion is that drylands have the potential to reach an annual carbon sequestration rate of 1.0 Gt/yr. This is the figure cited as the minimum value to be considered as relevant in the efforts to mitigate the build-up of atmospheric carbon.

- It has been verified that drylands have a significant carbon sequestration and reduction potential but there are high risks. The risks arise because of the small differentials between degraded and rehabilitated land and because of the vagaries of the climate. Carbon is easily gained but just as easily lost.

- Active programmes of carbon sequestration are feasible. "Business as usual" scenarios will not achieve the expected net sequestration of 1.0 Gt/yr.

- Carbon offsets, properly documented and monitored, should be a component of any regime for mitigating carbon emissions.

- The land area required to make a significant difference to the global carbon balance is quite large. If terrestrial carbon storage is attempted, in the order of 2-5 x billion ha of land will be required to absorb just 25% of the carbon dioxide emissions.

- Validation of carbon sequestration will be difficult in drylands because of the large areas involved and the remote locations.

- The risks associated with carbon sequestration projects will increase along the aridity gradient. Carbon stores are less secure in drier regions because drylands are characterised by vagaries of climate and because of the complex land tenure and community structures. Factors are less favourable to long term management of most arid lands. There are some exceptions, for example, areas of inland Australia where there are no livestock.

- There is a synergy between global environmental problems and local aspirations. "Bottom-up" efforts are the key to successful implementation of any anti-desertification and/or carbon offsets programmes.

- In forging a link between global climate change, fossil fuel burning and anti-desertification measures it is should be remembered that anti-desertification measures benefit carbon sequestration and that carbon sequestration per se benefits anti-desertification efforts.

- There is still much to be done yet to formalise carbon offset agreements, Mechanisms for payments, auditing carbon stores, prevention of carbon fraud, accounting for carbon credits etc, need to be worked out.

Recommendations from the Workshop

- Examine the results of past efforts at reversing land degradation, in different regions of the world, with a view to assessing outcomes (successes and failures). Look for lessons learned from these projects. A team effort involving a resource economist and a physical scientist would be most appropriate for this desk top study. It is estimated that each region would take 3 weeks work. There is need to look at 4 or 5 sample areas. The purpose would be to develop a set of Guidelines on how to implement a carbon sequestration project.

- Establish a network of people and agencies concerned with global climate change, especially the UNEP/WMO working group concerned with Climate and Desertification.

- Involve the GCTE programme and its established network of scientists involved in the global change and terrestrial ecosystems. This is particularly relevant in that the 1km x 1km land cover data is almost finalised. Access to this database would assist the delineation of dryland areas which have most potential for carbon sequestration.

- Develop two projects, one in the near term and one which would take 3 years. Box 1 sets out the project in outline. The objectives would be to a) improve the quantification of carbon sequestration and b) undertake a global synthesis of dryland ecosystem land use change.

## PROPOSED PROJECT 1: NEAR-TERM IMPROVED REGIONAL QUANTIFICATION OF CARBON SEQUESTRATION OPPORTUNITIES AND PROJECT DESIGN

Given that drylands seem to represent a sufficient potential opportunity for carbon sequestration that further analysis is worthwhile, a 3 month desk-top study should be undertaken to:

1. consistently and critically assess what management changes would preserve or capture carbon stocks for the major landuse/ecosystems of a small number of well-defined regions, incorporating at least one iteration through biophysical, socio-economic and cultural/political constraints for each change;

2. design a number of potential projects in specific regions, and rigorously assess them comprehensive against the credibility criteria defined by Trexler.

Methodologies should be based on approaches defined at this workshop. Regions could include Mahgreb, and other areas of Africa, but might also include some data-rich regions such as the US, Mediterranean or Australia.

## PROPOSED PROJECT 2: GLOBAL SYNTHESIS OF DRYLAND ECOSYSTEM LAND USE CHANGE

A three-year project with the objective of quantifying changes in carbon storage and radiative forcing in dryland regions of the world through a systematic geographic analysis of opportunities. The project will reduce uncertainties in the following factors:

1. land area considered
2. rate of land cover/ecosystem/land use change
3. potential and actual carbon storage densities
4. rates and mechanisms for the recovery of carbon storage potentials
5. economic cost
6. probability of success

The procedure would be to combine literature search and review, ground-based data from existing studies, remote sensing and photogrammetric analyses, improved global climate surfaces, and model analyses. The intention must be to incorporate economic analyses so that the uncertainties in costs are genuinely reduced, and to assess the opportunities and risks arising from community participation.

Products would include:

1. within 6 months, a critical review of mechanisms and magnitudes based on existing information;
2. after 3 years, a detailed analysis based on new work.

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## TECHNICAL PAPERS

# Land degradation control measures in the drylands - the human dimensions

Elizabeth Migongo-Bake

## Synopsis

This review examines the concept of desertification and outlines the causes, both natural and anthropogenic. The area of land affected and the socio-economic implications are reviewed. New approaches towards more sustainable land degradation control and improvement in carbon storage and conservation in drylands are outlined. These emphasise the human dimension of desertification and the need for the adoption of a bottom-up approach if any change in land use practice is to be implemented. Coping strategies of dryland inhabitants are examined and their relevance to anti-desertification measures explained.

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Key points

1. There is a great deal of discussion in environment and development circles about the importance of involving rural communities in activities aimed at controlling land degradation in drylands, promoting good land management and achieving sustainable development. However, experience has shown that these objectives are extremely difficult to achieve in practice, especially since, in the past, the approach to development has largely been a top-bottom rather than the currently advocated bottom-up one.

2. Intensifying farming systems to produce more food for subsistence for sale has become extremely difficult in degraded land without excessive external support. More intensive use of local natural resources often accelerates land degradation. With the current problems of high rate of population increase in the drylands, both local and as a result of cross-border civil war migrants, improved approaches towards carbon storage and conservation are crucial, through the creation of alternate livelihoods and a more sustainable use of the natural resource base.

3. Coping strategies frequently imply accelerated land degradation associated with over-use of diminishing resources in a fragile environment. This abuse of natural resource base is seldom due to carelessness or ignorance, but one resulting from survival mechanisms under harsh conditions. Droughts often stimulate sequences of actions and reactions leading to long-term land degradation. Drought may also trigger local food shortages, speculation, hoarding, forced liquidation of livestock at depressed prices, social conflicts and many other disasters associated with famines that may catastrophically affect numerous groups and strata of local populations. In some instances, however, droughts may contribute to the emergence of social strategies that enhance sustainable land productivity while protecting local livelihoods.

4. The main problem that persists in promoting the bottom-up approach is the bottle-neck that exists in the vertical communications between local communities and the state powers that formulate policy, planning and financing that affect local people's ability to make land management decisions. Developing effective lines of communication remains a challenge for the future. Decentralization of these three areas controlled by state and empowering the community in decision-making in development and management issues that affect them directly and indirectly. Local communities need to be encouraged to play a leading role in forging the direction for their own future development.

Key words: Coping strategies, land degradation, farming options, biodiversity, carbon storage, sustainability, land tenure, bottom-up approach, drought

## The concept of land degradation and desertification

Desertification is most frequently, and wrongly, imagined as the encroachment of desert sands on once fertile and productive lands. This ambiguity is a major contributor to the lack of consensus on what the term desertification means. However, most definitions do agree that the term implies land degradation in dryland ecosystems.

UNEP has defined desertification as a "complex process of land degradation in arid, semi-arid and sub-humid areas resulting mainly from adverse human impact". UNCED broadened this definition to "land degradation in arid and semi-arid and sub-humid areas resulting from various factors, including climate variations and human activities".

Land degradation is a social concept, as is desertification. Different land user communities will have different perceptions of land degradation. For example, pastoral and peasant agricultural communities will perceive the processes of land degradation differently from commercial farmers and other land managers in industrial societies. Also, within the same society, perceptions may vary greatly according to the observers experience, class position, social status, gender, and many other factors. Definitions and measurements of land degradation are, therefore, to a large extent arbitrary.

### The causes of land degradation and desertification - a natural-social interaction

The causes of land degradation are both direct and indirect, and interactive, at both the natural and anthropogenic levels. Natural and social processes both play important roles in bringing about land degradation. The two are usually inextricably intertwined, but in different mixes in each local situation.

It is not a single disaster that creates land degradation, but a succession of these and which has a great damaging effect, especially when on-going social and economic conditions are such as to expose the production system and land to abnormal harm from such events. The main direct causes of soil degradation in dry regions may largely be attributed to human activities. However, natural causes such as past and recent climatic changes, pests and diseases, plagues and other natural disasters, have an indirect interaction that indirectly and directly ties up with the human coping strategies that cause soil degradation.

Human activities that are directly related to land degradation include deforestation, over-cultivation, overgrazing and mismanagement of irrigated lands. Indirect factors include land tenure, trade policies, monoculture, especially in the cash crop sector, poverty and under development, among others. The direct causes do not occur by accident, but are greatly influenced by effects of growing populations, economic development and conscious policy decisions by government and aid agencies.

Droughts often stimulate sequences of actions and reactions leading to long-term land degradation. Drought may also trigger local food shortages, speculation, hoarding, forced liquidation of livestock at depressed prices, social conflicts and many other disasters associated with famines that may catastrophically affect numerous groups and strata of local populations. In some instances, however, droughts may contribute to the emergence of social strategies that enhance sustainable land productivity while protecting local livelihoods.

## The extent and area of land degradation

There are considerable differences in opinion on the extent, seriousness and trend of land degradation in the drylands. Estimates from 1980s suggest that over 30 million square kilometers suffered from at least moderate desertification (Dregne, 1976; Grainger, 1990). This amounts to about one fourth of the earth's land area and over two thirds its dry areas, excluding hyper-arid deserts. UNCED's Agenda 21 estimates that world-wide there are some 3.6 billion hectares of drylands have been degraded. The proportion of drylands at risk that were already suffering at least moderate desertification were believed to range from about 70 to 80 per cent in Africa, Asia, and South America, to less than half in North America and Australia.(Dregne, 1976, Grainger, 1990). UNEP's World Atlas of Desertification (UNEP, 1992), showing the extent of dryland regions and the proportions that have suffered various degrees of soil degradation, appears to be the most authoritative to date. In terms of severity, the Atlas states that by far the worst affected are North America and Africa with 76 and 73 per cent of their drylands degraded. Dregne (1991) estimates that of the world's drylands, 69.5 per cent are degraded, 19.5 per cent of which is human induced (Table 1)

*Table 1. Status of Desertification in the World*

| | Millions of Hectares | % of total drylands |
|---|---|---|
| 1 Degraded irrigated lands | 43 | 0.8 |
| 2 Degraded rainfed croplands | 216 | 4.1 |
| 3 Degraded rangelands (soil and vegetation degradation) | 777 | 14.6 |
| 4 Drylands with human-induced soil degradation (1+2+3) | 1036 | 19.5 |
| 5 Degraded rangelands (vegetation degraded without recorded soil degradation | 2556 | 50.0 |
| 6 Total degraded drylands(4+5) | 3592 | 69.5 |
| 7 Non-degraded drylands | 1580 | 30.5 |
| Total area of drylands excluding hyper-arid deserts | 5172 | 100 |

Source: Dregne, H., Kassas, M. and Rozanov, B. 1991.

## People affected

In the world's drylands, excluding hyper-arid deserts, more than one hundred countries are affected by the consequences of land degradation and desertification and as many as 900 million people living in these areas are at risk from the effects of the loss in productivity. The people most directly affected by land severe land degradation in the drylands are among the poorest and least educated with limited access to power.

The majority of the dryland human populations struggle daily with persistent and almost universal poverty in their struggle to scrape a living from a harsh environment where with droughts being a common phenomenon and soils are of low fertility, productivity is very low. In addition, traditional technologies have not kept up with the present rate of population growth and increased demands for food, fuel and shelter. The end direct results are poverty, hunger and malnutrition.

Unable to survive with scarce land and water resources, these poor populations are often forced to become environmental refugees that migrate to neighboring lands and urban centers in search of relief, employment and refuge.

## Social and economic impacts of land degradation

The human aspects are related to both population pressure and land-use technologies that are not sustainable as they have not developed alongside the rapid population growth that is being witnessed in the third world but whose negative effects hit the drylands most. The best known of these land-use technologies is the fallow system that in earlier times involved the resting of exhausted land long enough to allow fertility recovery through secondary revegetation. This original time span is almost non-existent now as a result of land pressure, especially in the African drylands.

As much as the inherent ecological fragility of the drylands coupled with recurrent droughts increase the degree of susceptibility of human-related land degradation processes, so do the latter affect the impact of drought through the weakening of the resilience of the system and the ability to return to equilibrium.

Land degradation through loss of vegetation and soil cover contributes to global climate change by increasing land surface albedo, increasing the potential and decreasing the actual evapotranspiration rate, changing the ground surface energy budget and adjoining air temperature, and adding dust and carbon dioxide to the atmosphere.

Impacts of land degradation on the natural resource base with direct effect on affected human populations include:

- reduction of perennial and annual livestock forage in rangelands;
- reduction of available fuelwood material;
- reduced biodiversity
- reduced water availability due to a drop in water table;
- sand encroachment on productive land, human settlements and infrastructure;
- increased flooding as a result of sedimentation of water bodies; and
- reduction of yield or crop failure in irrigated or rainfed farmland;

All these factors may ultimately lead to disruption, at various degrees, of human life due to deteriorating life-support systems that are expressed by:

- increase in the spread of poverty and hunger due to loss of land resources and consequent inability to provide sufficient food and shelter to growing populations, leading to a reduction in the nutritional and health status of the affected populations, especially the young and the elderly;
- migration in search of relief and refuge as a result of economic and political stress as populations struggle to survive on the diminished water and land resources; and
- an influx of environmental refugees that puts enormous pressure on the physical environment, economy and stability of societies in the immediate neighborhood, often exacerbating political differences and in some cases civil strife.

## New approaches towards sustainable land degradation control and improved carbon storage

The social issues to land degradation are always to some extent locality specific. However, the principal socio-economic processes and institutions stimulating land degradation do not originate locally (Swallow, this volume). Remedial initiatives have to be national, international as well as local.

The main areas that have recently been identified as prerequisites to sustainable land use and natural resource management include: participatory approaches to research, project planning and implementation; indigenous knowledge; biodiversity; land tenure and common property resources and government policy, among others.

*Community participation - a bottom-up approach*

A prerequisite for any kind of really sustainable development is a popularly-based development strategy. Trade, price and credit, investment and social policies should be a part of a broader strategy aimed at meeting the basic needs of vulnerable groups and creating opportunities for them to improve their livelihoods both qualitatively in terms of food security and standard of living. Such a strategy, however, implies popular participation. This also means the participation of local communities in formulating the strategy's objectives and in its execution when it directly affects their livelihoods. It means participation at all levels of development processes from problem identification to programme design, implementation and monitoring progress and evaluation of the end results.

Two of the recommendations at the 1993 "Listening to the People" workshop called for the use of culturally appropriate technologies in all stages of development and for participatory research to be in line with the socio-economic, ecological and political contexts and management systems of the communities concerned.

To help make effective this new approach, donors, international development agencies and national governments will, in their policies and strategies concerning drylands, need to become more democratic and responsive to the needs of the poor in climatically marginalized lands.

*Indigenous knowledge, biodiversity and sustainable development*

In the area of biodiversity and the drylands it has been recognized that indigenous people's subsistence systems are often repositories of unique genetic variability (Peacock et al., this volume). Development activities at the local level therefore need to promote technologies that take this variability into consideration in their activity planning.

In consideration of the fragile nature of the drylands, governments, donor and development agencies should aim to support and enhance the adaptive capabilities of small-holder production systems on marginal agricultural and pastoral lands (i.e. silvi-pastoral and agroforestry with indigenous species, crop diversification or rotation, and irrigation technologies in farming systems), and to minimize the disruptive effect of new technologies and systems on the local capabilities and loss of biodiversity.

The international community and governments should support efforts by local communities to develop environmentally sustainable marketing system of renewable natural products from threatened dryland ecosystems to counter the prevalent exploitative activities that tend to degrade natural ecosystems.

*Land tenure and ownership*

Insecure land tenure and inadequate livelihoods induces the poor dryland communities to mine the natural resource base unsustainably. Land reform of some kind is essential in the drylands if land degradation is to be effectively controlled. The economics of managing low productivity drylands are such that efficient and equitable common property regimes that are self-administered would often be more preferable to the expensive top-down administrative costs implied by effective private or state-ownership. In many agricultural regions where customary tenure has already broken down, and especially where good land has been monopolized by a few large holders, redistributive reforms may be required by, dividing the land among individual families, or possibly small cooperatives-operatives.

It should be noted, however, that any attempt at future projections regarding tenure issues in the dryland ecosystems will have to proceed in stages, as solutions must be adapted to fit the mosaic of tenure situations found across the socio-cultural-economic mosaics in the various ecoregions. For example, in the Sahelian ecoregion, it is difficult to reconcile the application of complex, centralizing, cumbersome land tenure legislation with the diversity of customary practices which are based on different views of land and resources (Ayoub, this volume).

However, the political obstacles towards achieving this are always great and the international donor community will need to play a role in exerting influence, by establishing criteria of conditionality where national governments are unwilling to accord local communities appropriate land tenure that promotes sustainable land management.

### *Land tenure and policy*

To avert land degradation on a long-term basis, natural resource management in the drylands has to be a co-shared responsibility between the community and the state, where even though the community is the primary controller of natural resources, the state retains an essential role. This control role could be positively enhanced by, for example, by the state ensuring the provision of an enabling environment at the community level, in terms of policy, primarily, and an infrastructure conducive to long term sustainable natural resource management/use e.g. credit and market facilities, roads and services (seeds, implements, storage, pricing), etc. There can be no getting around the need to recognize the principle of co-management of resources by the state and the local people, whereby, while retaining the final control, the state may entrust its sustainable management to the user communities. In this regard, the state must also provide not only a national policy guidelines in the respect of tenure, but also design a legislative and macro-economic framework which fosters greater security of tenure, while respecting the extreme diversity of regional and local situations.

It is clear that decentralization processes for sustainable natural resource management will have to go hand in hand with security of tenure, especially in the light of the fact that the state has often had the ultimate power to expropriate land for public or private purposes, which continues to pose a real threat.

### Land-use policy and land degradation

Policies and programmes aimed at controlling land degradation in the drylands has includes activities such as : prohibition of burning on pastoral lands; forbidding grazing in wooded area; afforestation of eroded lands, often on a massive scale; planting of shelterbelts, contour ploughing, terracing, bunds and other soil conservation practices; sand dune stabilization by planting resistant grasses, trees and shrubs; drilling of bore-holes for livestock water and breed improvement programmes. Most of these activities, although relatively successful according to land conservation or social criteria for limited areas and for limited time periods, have rarely been successful by both social and ecological criteria. They have had, overall, little impact on dryland land degradation in the

developing world. Most such activities have become unsustainable after active donor financial and technical support has ceased.
National policies that have led to land degradation at the community level include:

- Pressuring rural communities towards the production of commodities, largely *mono-crops, for national and international market;*

- *Setting-up of commodity pricing that undermines the value of rural products thus* escalating the prices of rural peoples needs;

- Processes of national development, especially programmes of expansion of farmlands for the production of cash crops, that exacerbate conflicts over land and water use and often reduce areas available to marginalized communities;

National policies and strategies should, among other things, attempt to provide real opportunities and incentives for the rural poor to manage their natural resources sustainably. At the same time they have to improve livelihoods. Price, trade, credit, investment, social, fiscal and other policies have to be designed to contribute toward meeting the priority objectives of sustainable development.

A crucial point is that national strategies that directly affect the livelihoods of the local communities will have to be popularly based. Governments should legitimize the existence of community groups by guaranteeing them freedom and autonomy in the management of their natural resources. Local communities should be involved in deciding what research and development programmes should be undertaken in their area as well as in the environmental assessment procedures and recommendations;

## Some cases of success

Although the majority of programmes aimed at controlling land degradation in the dryland ecosystems have not been very successful, largely due to a wrong approach, there have been a few cases of success. There is also experience now of successful interventions and there is growing understanding of sustainable production systems, past and present. There are examples of approaches that have worked, involving local participation; appropriate technology transfer; the use of local institutions and decision-making mechanisms; the strengthening of management methods based on indigenous knowledge systems; the creation of more equitable relationships between men and women, or different social classes, which results in better natural resource management; and the securing of land tenure and access to resources.

UNEP honored eight such cases spread across the world's drylands, in June, this year, by awarding the winners the "Saving the Drylands" certificates of appreciation in Almaty, Kazaskstan, during the celebrations to commemorate the World Day to Combat Desertification and Drought ( UNEP, 1995). These successes need to be replicated in many more similar situations and on a larger scale in order to address the global scale of the dryland land degradation problem (UNEP 1994).

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# Assessing dryland system dynamics : a UNEP perspective

**Miriam Schomaker**

Synopsis
This paper considers the entire array of information and data collection, management, analysis and reporting aspects relevant to assessing dryland system dynamics in the context of sustainable development. Although the paper focuses on drylands, the broader assessment process discussed applies for any region. Issues are highlighted which are important when considering assessment of dryland systems, based on experiences so far.

The paper briefly discusses: (I) results of past global assessments; (ii) the concept of desertification; (iii) the definition of world drylands; (iv) desertification in drylands per geographical region; desertification rate for a number of countries; and (v) a discussion on desertification costs: damage and rehabilitation. Particular emphasis is placed on a) user relevance of assessments; b) the need for integration of environmental and human sub-systems, with reference to indicator development and the Pressure-State-Response framework; c) different perceptions of dryland issues; d) scale issues; e) methodological and science issues; f) data issues; and g) data and information management and assessment capacity building. It then explains how UNEP is currently approaching environmental assessment and reporting in general and for dryland systems in particular

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Key Points
1. To give more visibility to the drylands, we need to present a better picture of the dryland system dynamics, but also ensure that realistic responses can be formulated to reverse negative trends. This applies to any aspect of the dryland system. If we, for instance, want to prove to the world that dryland systems are an important potential carbon sink, we should not stop after providing the "scientific evidence" of such a statement. Such information is only relevant if we can at the same time provide realistic options for action (responses) to ensure that the drylands would indeed become a major sink. Realistic means: directly linked with society. The options should not only be technically feasible, they should also be economically viable and socially and culturally acceptable. In other words: any assessment activity should always keep the ultimate goal, sustainable (dryland) development, in mind.
2. A major problem facing dryland researchers, policy makers and technical advisers is that there is not enough sufficiently reliable data with a global coverage. Many publications are mainly based on expert opinion and ..... guesswork. There is no straight forward, internationally agreed way of assessing dryland systems. Assessment efforts still tend to take place on an isolated case study and/or subject level. There are, among others, different "schools-of-thought", different methodologies, different entry points, data availability and compatibility problems, scales problems, and different ways of presenting information.

Key words: coping strategies, land degradation, farming options, carbon storage , sustainability, bottom-up approach, land tenure

## Introduction

Initially the paper was to discuss "where dryland degradation occurs and how it was mapped". Since there are many recent publications existing in which the "where and how" has been dealt with, this paper will go beyond this and consider the entire array of information and data collection, management, analysis and reporting aspects relevant to assessing dryland system dynamics in the context of sustainable development. No scientific results will be presented. For real science UNEP heavily relies on the scientific community outside its own organization, as for instance present during this workshop.

Though the paper indeed focuses on drylands, the broader assessment process discussed applies for any region. Those issues will be highlighted which are important while considering assessment of dryland systems, based on experiences so far. It will then be explained how UNEP is currently approaching environmental assessment and reporting in general and for dryland systems in particular. In many cases we don't have solutions. By putting the questions on the table we will hopefully be able to move towards answers.

The term dryland system is used instead of dryland degradation because degradation is only one part of the story. With the word system both the environmental sub-system and the human sub-system are considered; only by intimately linking these two sub-systems one can truly study the dynamics of the drylands. To give more visibility to the drylands, we need to present a better picture of the dryland system dynamics, but also ensure that realistic responses can be formulated to reverse negative trends. This applies to any aspect of the dryland system. If we, for instance, want to prove to the world that dryland systems are an important potential carbon sink, we should not stop after providing the "scientific evidence" of such a statement. Such information is only relevant if we can at the same time provide realistic options for action (responses) to ensure that the drylands would indeed become a major sink. Realistic means: directly linked with society. The options should not only be technically feasible, they should also be economically viable and socially and culturally acceptable. In other words: any assessment activity should always keep the ultimate goal, sustainable (dryland) development, in mind (Stafford Smith et al., this volume)

In the next section the "where and how" and the assessment experiences so far are briefly touched upon. In section 3 some of the assessment issues are discussed in more detail. In section 4 UNEP's Assessment Framework is described and some of the major activities UNEP is involved in or planning are highlighted. Through these activities UNEP hopes to be able to contribute to providing a better knowledge base for informed decision making and management *inter alia* related to dryland systems.

## 2 Dryland degradation: where is it and how was it mapped

The most recent UNEP publications on where dryland degradation occurs and how this was assessed date back to 1991 and 1992, prepared as direct input to or as background material for the UN Conference on Environment and Development in Rio de Janeiro, Brazil, June 1992. In addition, many publications exist describing dryland degradation assessments on a case studies basis, covering only one, a few or even part of a country. The main recent UNEP publications are:

*1 Deichmann, U and L. Eklundh (1991). Global Digital Datasets for Land Degradation Studies: A GIS Approach. UNEP/EAD(GEMS/GRID) Case Study Series No. 4.*

This technical report serves as background information to the Atlas listed below rather than as a detailed analysis of the data generated. The report describes UNEP's work in assembling the core data sets used in the production of the global and regional sections of the Atlas. Concepts of geographic analysis are introduced, and methodologies for the production of numeric and cartographic output are presented. The core data sets used (and where necessary modified) for the Atlas are discussed: (i) World Map of Human-induced Soil Degradation (GLASOD) produced by ISRIC (International Soil Reference and Information Centre) in the Netherlands with UNEP funding; (ii) long-term climate datasets prepared by the Climate Research Unit of the University of East Anglia, U.K.; (iii) Normalized Difference Vegetation Index (NDVI) derived from NOAA AVHRR data; population density (Africa only) for which spatial modeling techniques were used. The person who worked on the population density dataset is currently continuing his work at the University of California, Santa Barbara. The digital datasets used for the global and regional part of the Atlas can be obtained from UNEP/EAD in Nairobi.

*2 Dregne, H., M. Kassas and B. Rosanov (1991). A New Assessment of the World Status of Desertification. In: UNEP Desertification Control Bulletin Number 20, 1991.*

This paper briefly discusses: (i) results of past global assessments (UNCOD, 1977; UNEP, 1984); (ii) the concept of desertification; (iii) the definition of world drylands; (iv) desertification in drylands per geographical region; desertification rate for a number of countries; and (v) a discussion on desertification costs: damage and rehabilitation. Data used for this paper originate from the analyses made for the Atlas, from a dryland assessment study done by ICASALS (1991) and from various expert meetings organized by UNEP in preparation for UNEP's input to UNCED. Results presented are mainly descriptive and statistics based. It is a summary of UNEP's input to UNCED (below).

*3 UNEP (1991). Status of Desertification and Implementation of the UN Plan of Action to Combat Desertification. Report of the Executive Director. UNEP/GCSS.III/3.*

In addition to a more extended version of the above Dregne *et al* paper, the report describes: (i) progress made in the implementation of the UN Plan of Action to Combat Desertification; (ii) policy guidelines and course of action for combating desertification; and (iii) financing the plan of action for combating desertification.

*4 UNEP (1992). World Atlas of Desertification. Published by Edward Arnold, London, UK*

In the Atlas an attempt was made to present a general picture of degradation in the drylands on a global scale (Part I) and for the African continent (Part II). In addition, case studies are presented which illustrate some assessment methodologies applied (Part III). Information is presented in maps as much as possible.

A major problem with publications as listed above was that there is not enough sufficiently reliable data with a global coverage; such publications are mainly based on expert opinion and ..... guesswork. Besides, the different character of the above publications, and of the many others not listed, illustrate that there is no straight forward, internationally agreed way of assessing dryland systems. Assessment efforts still tend to take place on an isolated case study and/or subject level. There are, among others, different "schools-of-thought", different methodologies, different entry points, data availability and compatibility problems, scales problems, and different ways of presenting information.

This is no news. Many recommendations in Agenda 21 (UNCED 1992) refer to this, in particular in Section II: Conservation and Management of Natural Resources (Chapters 10-16). Also the new UN Convention to Combat Desertification (adopted in Paris in June 1994) "calls for the need to integrate and coordinate the collection, analysis and exchange of relevant data and information to ensure systematic observation of land degradation in affected areas and to understand better and assess the processes and effects of drought and desertification".

Reference is made in particular to Article 16 on Information Collection, Analysis, and Exchange; to Article 17 on Research and Development; to Article 18 on Transfer, Acquisition, Adaptation and Development of Technology; and to the Urgent Action for Africa part of the Convention. Indeed more collaboration is needed resulting in an agreed umbrella framework and in further development of realistic, compatible methodologies (including indicators) and procedures to improve the knowledge base of dryland systems and their dynamics.

Already during an ESCAP/UNEP expert panel meeting of the Regional Network of Research and Training Centers on Desertification/Land Degradation Control in Asia and the Pacific (DESCONAP - April 1994, Alice Springs, Australia), the limitations of existing methods were discussed after which efforts were made to develop more generally applicable guidelines for assessment and mapping of desertification/land degradation in Asia and the Pacific. No doubt, many more examples of such efforts could be given. Box 1 lists the key questions that were put before the experts to facilitate the discussion; it was copied since it nicely illustrates the complexity of the issue, even though the list is not exhaustive.
The DESCONAP group identified the following limitations of existing methods:

- a high level of subjectivity and therefore not repeatable
- not enough precision to allow comparison of assessments through time
- often based on point assessments which are unrepresentative of larger areas

The group recommended "an approach based on the integration of remotely sensed data, verified by adequate ground referencing, and ancillary data in a GIS, which represents the best currently available technology to produce geographically explicit and precise assessment of dryland resources".

While the list in Box 1 goes into quite some detail some issues have been left out, in particular on the part of the human sub-system (see next section). Issues, often interrelated, involved in data and information management for and assessment of dryland systems are summarized below and will be discussed in more detail in the next section:

1 User relevance

2 Different perceptions

3 Integration

4 Scale issues

5 Methodological and science issues

6 Data issues

7 Data and information management and assessment capacity

**1 What are we trying to measure?**
- General definition. Is the definition of desertification in the draft international convention practical or do we need more specific definitions for measurement and monitoring purposes? If so, what definitions are appropriate and what indicators can be developed to apply them?
- Are we measuring past desertification leading to the current state of the land or should we be concentrating on measuring future change?
- Is desertification a cause or a symptom of land use problems and lack of sustainability? Can we separate symptom from cause? Do we need different methods to monitor these situations?

**2 Basic problems in measuring desertification?**
- Successes and failures in past programs. Review of case studies from different countries and agencies aimed at producing general conclusions on different methods and their appropriateness.
- Problems of short-term environmental fluctuations confounding determination of long-term change.
- Problems of measuring "loss of biological and economic productivity". What does that productivity refer to and can it be measured given all the other factors affecting it?
- Variation of desertification indicators in space and over time and the problems of developing adequate sampling strategies.
- Availability of data for measurement of desertification and the limitations imposed. Can we develop an "art of the possible" or must we wait for data to accumulate in the future?

**3 Techniques for measuring ecological state and biological productivity (ground-based and remote sensing).**
- Is it feasible to measure ecological state and biological productivity directly or must we use indirect indicators or surrogate measures?
- What surrogates and indirect indicators are appropriate? Can we develop measures linking them directly with productivity?
- Measurement precision issues (both ground and remote sensing).
- Scale and sampling accuracy issues (global, regional, local).

**4 Problems of measuring changes in economic productivity resulting from or associated with desertification**
- Availability of statistics.
- Other factors affecting economic productivity.
- Problems of linking economic productivity with the state of the natural environment.

**5 Generalized versus ecosystem or land use specific indicators of desertification**
- Global approaches and their limitations.
- Regional and local approaches for rangelands, cropped areas and mixed systems. Significance of land use change and loss of natural vegetation.

**6 The nature of environmental change**
- The role of climate variability.
- Social, political and economic causes of change.
- The problem of dealing with large areas and spatial variability.

**7 Identifying trend**
- Is measurement of current status enough or should we be trying to detect change and trends?
- Increased levels of precision required to detect change in status compared with current status only.
- What are the data limitations which restrict our ability to monitor change and identify trends? Are there any short methods using information on factors such as spatial pattern or change before and after key events such as droughts or wet periods?

**8 Practical technologies - global, regional and local scales**
- What can be done, where can it be done, and on what land use systems and natural environments?
- Where can techniques for desertification assessment be developed for particular ecosystems or land use types with limited research.

What are the unsolved problems in desertification assessment and how can they be addressed?

Box 1 Key questions during ESCAP/UNEP Meeting on Development of Guidelines for Assessment and Mapping of Desertification/Land Degradation in Asia/Pacific, April 1994, by Dr. G. Pickup of CSIRO

## 3 Main issues involved in assessment of (dry)lands

### 3.1 User relevance

Many assessment activities still take place within the realm of science. More direct links are necessary between the actual users and producers of information. Research and assessment activities should preferably be formulated together with users, in fact upon the request of users (for their purposes). Depending on "who wants to know" different levels of detail and different forms of information are needed. Once the "why" is clear the research agenda and the kind of data needed can be decided upon. In section 3.3 an example is given of a user relevant assessment methodology. Nowadays users are often considered in the context of the decision-making cycle which includes four stages (Figure 1).

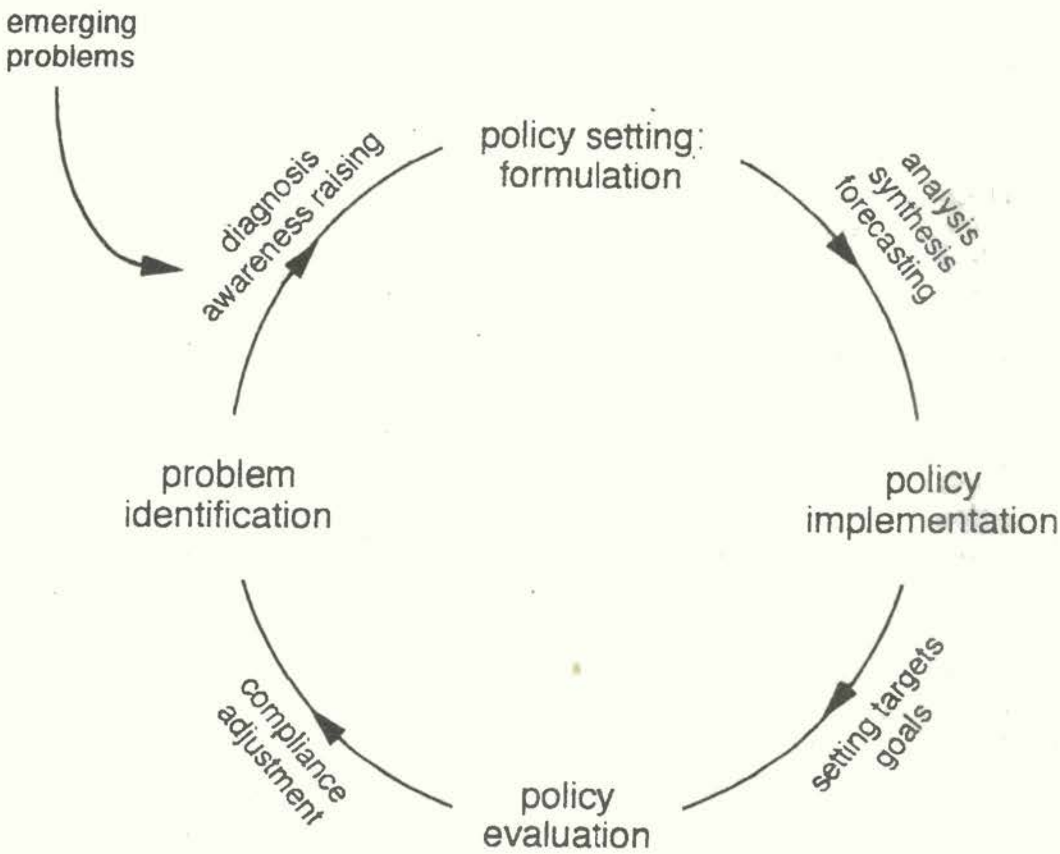


Figure 1 The decision-making cycle (adapted by RIVM from Winsemius 1986, in RIVM/UNEP 1995)

Decision-making processes take place at all levels of government and involve many different cultural, social, institutional, economic and environmental inputs and considerations. Depending on what level and which stage in the cycle, the kind of information needed differs.

For problem identification and awareness raising more general, descriptive indicators are needed on status and trends; and different audiences need customized material. To reach the international policy setting and resource allocation scene one would opt for a different presentation then to reach the general public.

For strategy, policy, project formulation one would need more detailed indicators, also focusing on the causes of a certain problem and on projections of impacts, through modeling, scenarios, cost-benefit and multi-criteria analysis, so that effective, and realistic, responses can be formulated. For the actual implementation of policies, goals and targets need to be established, primarily at national and local level (more quantitative indicators). Here the societal and economic context becomes increasingly important. It will be the people living in the drylands that will have to decide on what, how, and when they want to and can reach certain targets. Assessment and information aspects should focus on negotiation and agreement on targets among all the stakeholders who in some way or another have an interest in the land.

For evaluating the effectiveness of policies and actions taken one needs to find indicators (quantitative) that illustrate how the situation has moved towards the set goals and targets.

Research and assessment activities should preferably clearly target the users and the stakeholders, both considering the level in the hierarchy (from local population to high level international policy makers) and the stage in the decision-making cycle.

## 3.2 Integration

UNEP has, over the past decades been focusing on state and trends, with emphasis on environmental sub-systems. However, though it is indeed necessary to know where a problem occurs, one also needs to know why the problem occurs in order to be able to formulate responses. To do so there is an urgent need to focus much more on the interlinkages between the environmental system and the human system (Figure 2). Not only should research and assessment activities cover both sub-systems; interlinkages between the two are even more important.

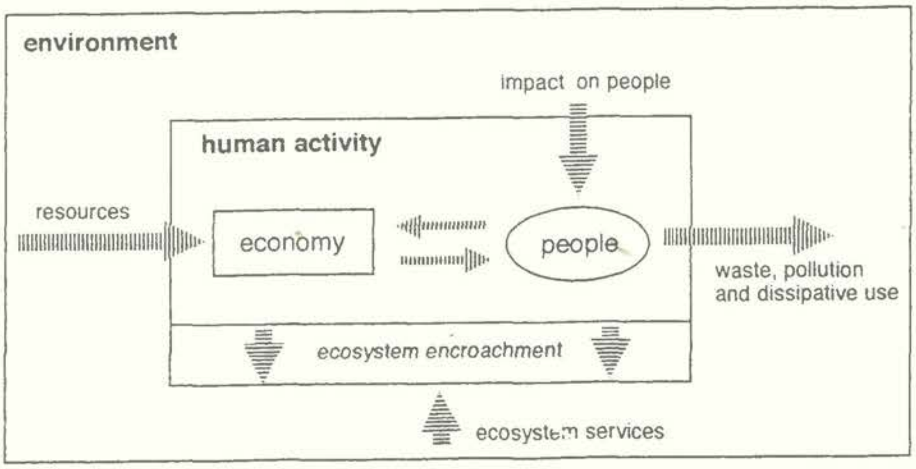


Figure 2 A model of human interaction with the environment (RIVM/UNEP 1995)

Causes of problems are usually predominantly human induced, causing pressures on a system, which often result in a negative trend in the state of the system. Only when the causes and the impacts of the resulting pressures on the system are known can adequate responses be formulated (Figure 3). To qualify and quantify the pressures, state and responses, indicators need to be found that adequately represent the extremely complex situation. The OECD PSR framework (Figure 3) for such indicator development is being adopted widely.

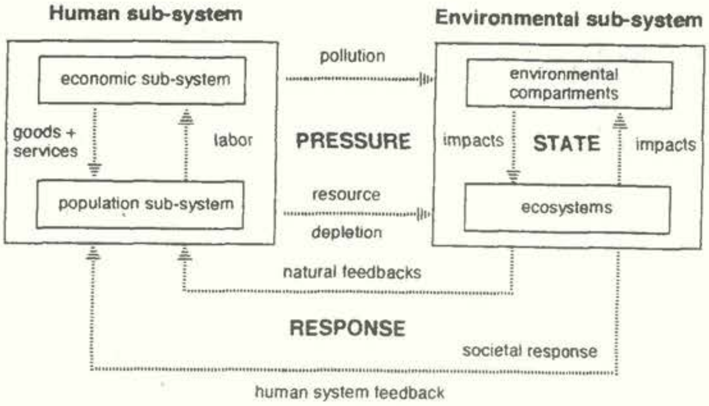


Figure 3 Pressure-State-Response Framework for Indicators (RIVM/UNEP 1995, adapted from OECD)

There are in fact hundreds of indicators which say something on, for instance, land quality and desertification (UNEP/RIVM 1994, UNDP/UNSO 1995, World Bank/FAO/UNDP/UNEP 1995, UNDPCSD 1995 etc). Some cover the causes/pressures part of the system, some focus on change in status and trends and the impacts of such change, and some are more related to responses. The challenge before us is to find those few indicators that are sufficiently representative and at the same time easy to understand and measure on a routine basis. Indicators should be SMART: specific, measurable, achievable, relevant and time-bound. Indicators are needed at different hierarchical levels (see also section 3.1 on user relevance), as illustrated in the Information/Indicators Pyramid below (Figure 4).

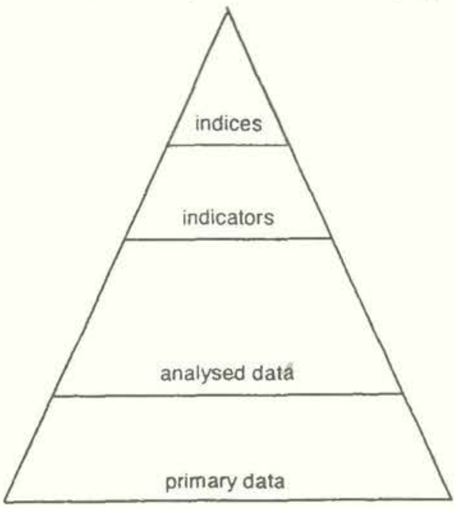


Figure 4 The Information/Indicator Pyramid (SCOPE 1995 and WRI 1995)

3.3 Different perceptions

Our perception of a problem is too often confined to conventional ways of thinking. For instance, we tend to "know" in advance that land degradation (including vegetation) is THE problem of the drylands and we enthusiastically start measuring and monitoring changes in land cover, soil degradation and the like. However, drylands ARE very marginal, highly variable and heterogeneous (both in time and space) ecosystems-systems. There are always pockets in drylands that are under (temporarily) high environmental stress. The traditional societies developed beautiful coping mechanisms to tackle these situations.

One example out of many described in the literature : in the pastoral areas of Eastern Africa people would move their herds around to those areas where their herds could be fed best (eg. the higher dry season grazing grounds). With the introduction of national boundaries, of non-traditional insecurity situations (political struggles, guns instead of bow-and-arrow), of arable farming in the dry season grazing grounds and the like, the total area used traditionally to cope with the harsh environment has decreased drastically. These societies also had traditional "agreements" with related surrounding societies: in case of problems one group could knock on the "door" of the other group and *vice versa*. Often pastoral societies were linked in this way with farmer societies. Since pressures in the arable lands are increasing as well, the farmer societies can hardly keep their promises, reason why these practices are often no longer in use. The traditional movements gave stress areas time to recuperate.

These days pastoralists have to stay in the stress area, because of this the land can not recuperate and will even degenerate further. By the time the local stress situation has eased out again the livelihood of the people has deteriorated, while, if they had been able to move to another area, they would have come out unaffected. With every new small stress situation that occurs (which is regularly), the society becomes more vulnerable because they don't have enough time to recuperate. Where regularly occurring, localized stress situations would go unnoticed in the old times, they now result in a gradual downward trend in the livelihood of the pastoralists.

In other words: the problem is not so much degradation of the environment but degradation of the traditional coping mechanisms resulting in deterioration of livelihoods. When considering the latter as the issue at stake a very different assessment programme would be required. Box 2 briefly outlines an example of an effort to work towards such an alternative approach (the Drought Monitoring Project in Northern Kenya).

The Drought Monitoring Programme (DMP) is primarily an information gathering network. Its main task is to provide decision-makers with data related to drought stress and recovery so as to better inform them on possible action.

Emphasis is on regularity of information and a grassroots approach.

The DMP's information system relies on continuous monitoring of the nomadic and semi-nomadic pastoralist population. From this target group, communities and households are selected using random sampling techniques. The sample population is visited every month by local monitors.

Three main variables are monitored : environment, local economy (livestock production and agriculture) and pastoralist welfare. Each variable is measured by multiple indicators. Changes in the environment are monitored by examining the number of available water sources and observing the condition of the rangelands (trees, browse and pasture). Questions relating to livestock production include animal births, animal mortality, slaughter rates and animal sales. Questions on agriculture refer to the type of crop grown, the condition of the crop and the levels of harvests. Welfare questions include diet (household food consumption), cereal purchase and prices, displacement of household members, population dynamics - eg. the movement of people into and away from the communities under observation - and the nutritional status of children in the age group 1-5 years. Nutritional status is recorded by measuring the mid-upper arm circumference (MUAC).

The project's grassroots style of monitoring best captures changes in the pastoralist's food security because subsistence is dominantly concentrated in household herds. Most matters about the herds are decided at the household; this is true even when households come together to form larger herding villages (communities). Monitors themselves originate from the area that they survey.

Systematic household and community monitoring yields valid, regionally-sensitive information. Additional sources of information are satellite imagery and data from GoK technical departments (eg., morbidity patterns for human and livestock population).

Environmental data obtained through remote sensing is collected by the DMP from the USAID FEWS programme. Data from satellite images on vegetation growing rates (NDVI) is processed and analyzed for 10 day periods. This information is correlated with data from ground monitors who observe vegetation conditions in and around pastoral communities.

Finally, data are collected by means of aerial surveys which are carried out by DMP twice a year. The aerial survey is useful as a rapid means to assess livestock and pastoralist population in the area. The aerial survey is a tool to assess the relative wealth and poverty and hence the level of vulnerability of the pastoralist population.

Box 2 Drought Monitoring Programme in Northern Kenya (personal communication)

Another aspect which seems to dominate our perception of dryland issues is that we tend to focus on the negative side of the story. There are, however, certainly positive stories to be found: programmes and projects that have found ways to improve the situation, to reverse negative trends. It may well be much more effective to look into these success stories to develop responses from these successes which are replicable in other areas. This will be discuss in a separate paper.

3.4 Scale issues

Ideally we would want to have detailed data on "everything". We would want to be able to smoothly move from an abundance of detailed field data to summarized information for national level purposes to even more comprised information for sub-regional, regional and eventually global level purposes (the Indicator Pyramid, Figure 4). For practical reasons (constraints in available time, human and financial resource) we would want to find shortcuts. We would want to be able to determine simple, direct links between field level data, general statistics and remotely sensed data at decreasing levels of detail (through extrapolation, spacial modelling techniques and the like). Once such relationships are established we would be able to monitor over time and we would be able to indicate which temporal scales are relevant for which aspects. Many wishes .....

3.5 Methodological and science issues

Box 1 touches upon many relevant aspects. In summary: most research and assessment work still takes place on an isolated, scientific case study basis. Methods developed are very site specific. Work is often carried out within the university realm (PhD studies and the like): an ideal situation where usually more equipment is available than in the real world, where "free" research staff time is at hand, where time pressure is not a real issue. As a result, methods developed under these circumstances are not easily repeatable, not broadly applicable, not realistic in terms of time, cost, and practical applicability within for instance a government and/or NGO structure, and not suitable to provide an overall picture for larger areas.

3.6 Data issues

There are not enough data available/being collected on a routine basis where on the other hand sometimes data are being collected because they have been collected for so long already while it is not always clear what they are needed for. The quality of data that are available is often questionable: no standardized procedures are followed, guess work is involved and the like. Data are often collected using "self made" definitions and classification systems, as for instance in the case land use and land cover: data from one area are not compatible with data from other areas, which hampers comparison and presentation of an overall picture. Data may well be existing but difficult to get hold of: they are stored in too many different places, are poorly documented and there often is a competition aspect involved. Much data is only available as general statistics and point data while one would ideally want georeferenced information.

3.7 Data and information management and assessment capacity

Apart from the need to further develop methodologies for data collection, data management, data analysis and integration, and data presentation there is a need to strengthen national capacities in all these aspects so that the entire world can contribute to the assessment process on an equal basis.

## 4 UNEP involvement in (dry)land assessment

UNEP's assessment mandate is to "keep under review the state of the world environment (SOE), enhance understanding of the critical linkages between environment and human activities, identify priorities for international action, flag emerging issues and strengthen national, regional and global information handling capacities for sustainable development". UNEP's assessment strategy or framework can be illustrated through a triangle (Figure 5a), in which all groups of activities needed for proper integrated SOE assessments are reflected. For assessments of a more specific issue, like (dry)lands, a comparable, but slightly simplified triangle can be drawn (Figure 5b).

As illustrated earlier, currently most assessment work for larger areas is based on guesses. Only little sufficiently reliable, compatible data exist for larger areas. Besides, no internationally agreed, comprehensive methodologies exist for issues like: (i) integration of natural resources data with socio-economic data; (ii) scale integration (both spacial and temporal); and (iii) change indicators. Nor are we fluent yet in presenting information in formats which are of direct use to / have direct impact on those we target. In the current situation the assessment and reporting triangle looks more like a pinnacle: information originates from a very weak data and analysis basis. Eventually we would want the pinnacle to look like a more stable and reliable triangle.

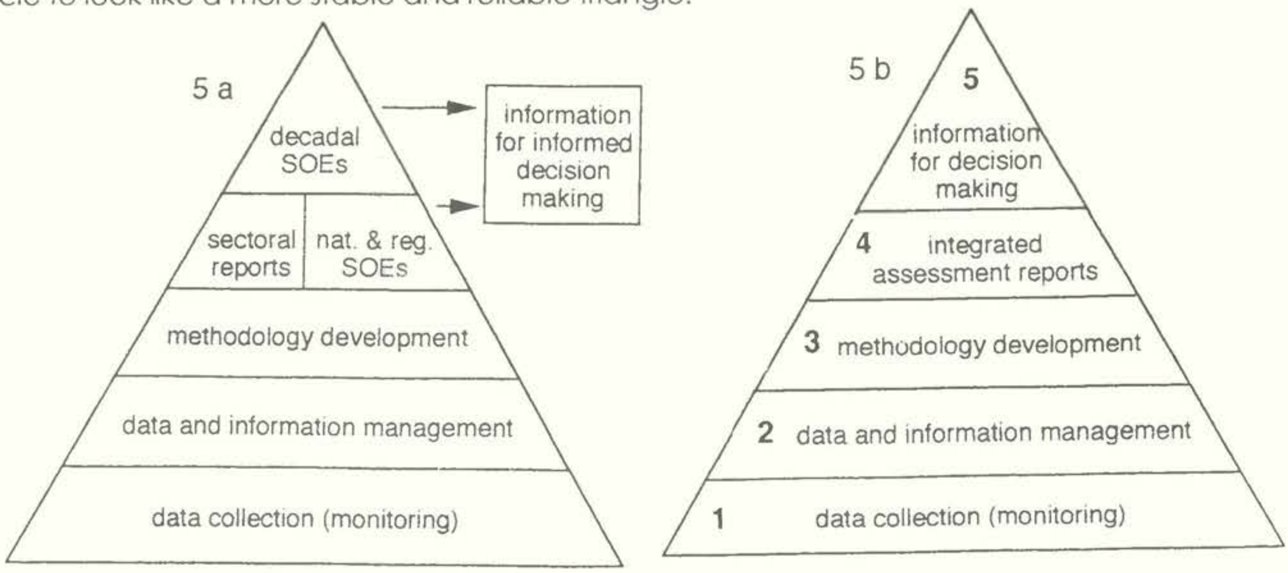


Figure 5 Providing the knowledge base for decision making

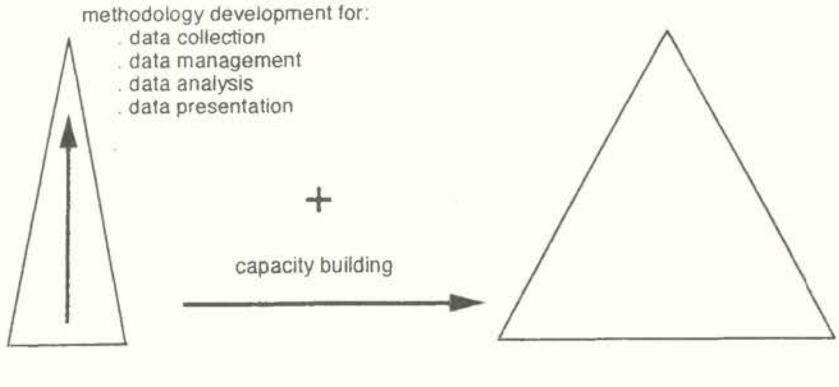


Figure 6 From pinnacle to triangle

To move from pinnacle to triangle we need to: (i) continue efforts to further develop operational data collection and data management tools, analysis, integration and modelling methodologies, and presentation formats; and (ii) continue efforts to enhance capacities in the entire assessment process. An enormous challenge ahead, for which we hope to expand our collaborating networks, for instance with experts as represented in this workshop.

Below, some drylands related activities are listed in which UNEP is involved for each of the five compartments indicated in Figure 5b. UNEP is not an implementing or donor agency though. We can merely try to promote activities, and do so by: (i) providing small financial contributions resulting in joint outputs (methodologies, datasets, ....); (ii) jointly developing frameworks others can link up to, (iii) provide expert advise. All our activities take place as joint efforts with other institutions (both within and outside the UN; both at international, regional and national level).

We mainly use the Environment and Natural Resources Information Networking programme (ENRIN) as the vehicle to strengthen capacities in all environmental assessment and reporting and associated data management. This programme develops umbrella frameworks (along the lines of the triangle) for different regions. It encourages major donor institutions in each region to link up and contribute to activities that fit within the umbrella framework. Emphasis is on increasing collaboration among existing institutions, programmes and networks (to avoid duplication). Use of outputs developed is promoted through this programme (can be any relevant successful output from anywhere): datasets, data management software tools (such as the Soil and Terrain Digital database methodology - SOTER), analysis tools/models, decision-making tools and the like.

1) Data bases and monitoring relevant to assessment of inter alia drylands:
   - further methodological development of the GLASOD approach (in the Asia and Pacific region); more emphasis on causes and impacts of degradation; expected mid 1996
   - (sub-) regional Soil and Terrain Digital databases (SOTER); Latin America and the Caribbean second half 1996; others to follow pending resources
   - contribute to the World Overview of Conservation Approached and Technologies (WOCAT) including the success stories inventory; GLASOD showing the negative side (human-induced degradation) and WOCAT the positive side (successful responses)
   - digital elevation model (DEM): an elevation database from which many products can be derived (eg. watershed boundaries, drainage flow patters etc.); joint USGS EROS Data Centre/UNEP/NASA work; available for Africa; other continents to follow towards end 1995 and in 1996
   - land cover; using AVHRR; joint US Geological Survey - EROS Data Centre; Latin America by end 1995; a related project being implemented for a number of countries in Asia and the Pacific; other continents to follow in 1996 and 1997
   - UNEP is co-sponsor of the planning phase for a Global Terrestrial Observing System (GTOS), comparable and linked to the already existing Global Oceans and Global Climate Observing Systems (GOOS and GCOS). Once operational GTOS would provide an excellent

umbrella mechanism for data collection and sharing for monitoring purposes.

- population distribution (through spacial modelling); 1996-97 pending resources
- contribute to development of new datasets on socio-economic, institutional and political aspects (pressures/responses) - depending on the outcome of indicator and other methodology development work (one of our major challenges)

2) Data and information management
   - have the Mercure satellite receiving system operational (in some developing regions in 1996, others to follow in 1997); it will provide easy links to all UNEP's data and information sources and to the Internet and eg. World Wide Web facilities so improving data accessibility and sharing
   - further develop the integral meta-data system for all UNEP data and information and for referencing to other data "in the world"
   - UNEP/FAO Initiative on Standardizing Land Cover and Land Use Classification Systems, WCMC/ITC/ITE; a flexible attribute based approach (including software)
   - support ICRAF work on software for flexible preparation of climate surfaces from basic climate datasets, for specific, user defined applications.
   - based on recommendation on specific data needs based on indicator and other methodology development work (see 3): develop data management tools for these data
3) Support methodology development for assessment
   - integration of socio-economic and natural resources aspects and scale integration (case study based, eventually to lead to more generally applicable methodology)
   - land quality and desertification indicator development (THE issue that links all assessment components together): contribute to ongoing and new initiatives such as DPCSD (UNSO/FAO/UNEP and many others working on desertification/land quality sheets for DPCSD framework), SCOPE, World Bank/FAO/UNDP/UNEP Land Quality Indicators; UNEP success story analysis; there is an urgent need to bring all these efforts on one line
4) Assessment reports
   - technical reports, datasets, software tools, decision-making tools (eg. indicators) resulting from the activities in 1, 2 and 3 above
   - a revised World Atlas of Desertification (first edition is out of print) by mid 1997
5) Information sharing

   Outputs will be produced as a "family of products". The same base material will be customized for specific target groups: "how too" booklets (eg. related to the success story inventory); brochures; desertification bulletins; radio programmes; videos; electronic information sharing; summary reports for policy and decision-makers; more elaborated technical reports and basic data for technicians and the scientific community; ......

   This is merely a list of some activities which may be of direct relevant to this group. Please do not hesitate to contact us for more detailed information.

The paper hardly referred to global warming. However by presenting a framework for the entire assessment process and by highlighting assessment aspects that still need to be further developed it has hopefully been made clear that there is an urgent need for a more collaborative effort to try and fill the, often still quite empty boxes within the assessment framework. We sincerely hope that experts as present at this workshop will be able to link up and assist in this international effort.

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# Some Features of Land Degradation in the Sahel: examples from the Sudan and elsewhere

A.T. Ayoub

Synopsis

This paper is a literature review on land degradation in the Sahel citing the case of Sudan. Studies show that, historically, vegetation cover varied greatly in responses to rainfall. Most of the studies relate land degradation to the observed variation in vegetation cover and not to vegetation composition or quality. The paper tries to link vegetation quantity, quality and topsoil loss to the recurring episodes of drought and hence to land degradation and its impacts. An account of drylands development is discussed in terms of runoff/runon, pastoralism/cropping and management of natural forests and woodlands.

Key Points

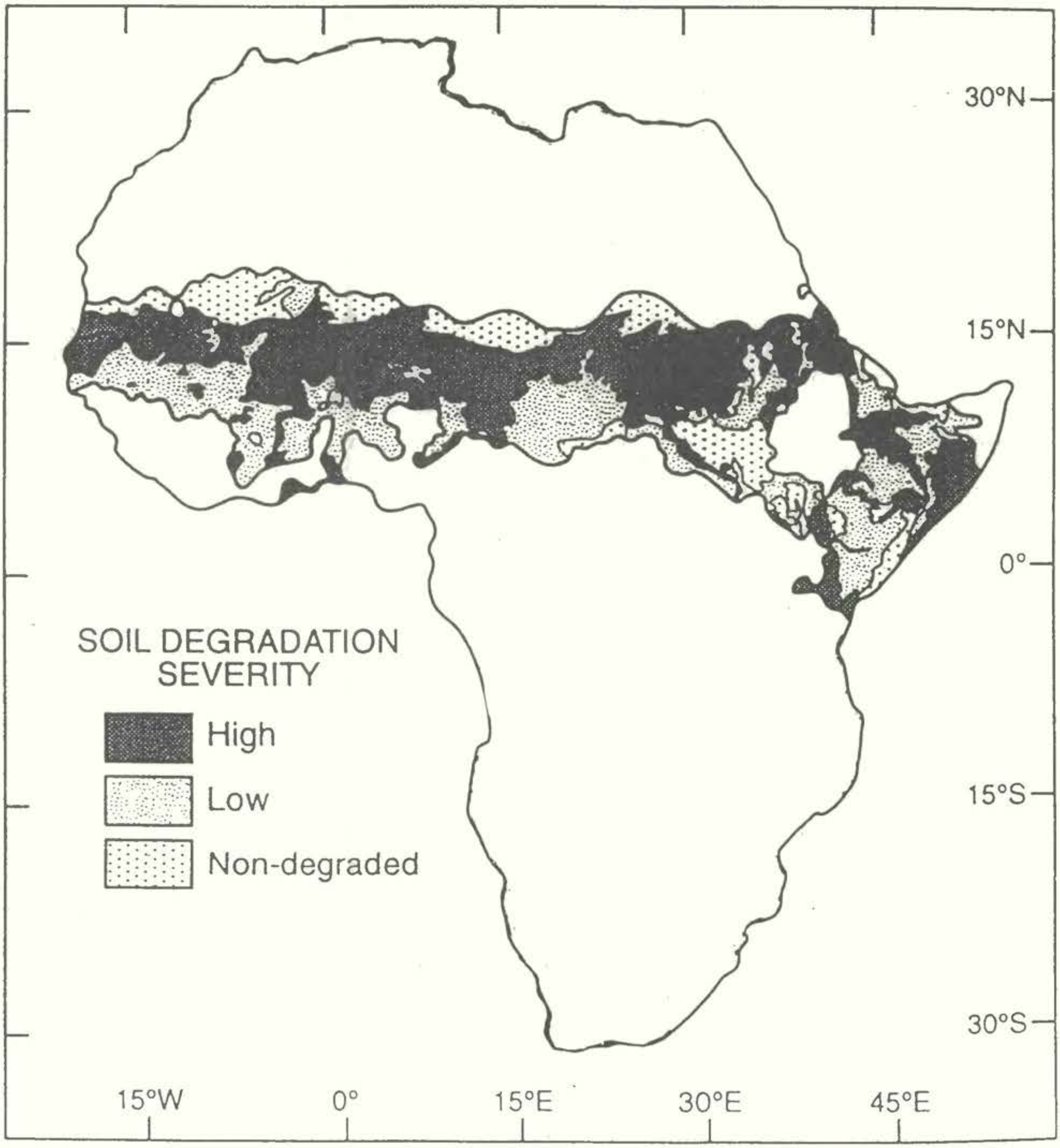
1. The Sahel, extends for approximately 7000 km across a continuous west-east belt of Africa and is over 1000 km wide . It covers 18 countries either completely or partially, and represents about 14% of the global drylands. Land degradation is the most important environmental problem affecting extensive areas of land in the Sahel. The problem of soil erosion is particularly acute while the problem of loss of soil fertility is increasing. Land degradation is serious because the productivity of huge areas of land is declining just when populations are increasing rapidly and the demand on the land is growing to produce more food, fiber and fuel.
2. A total of 124 million hectares is considered strongly degraded. In these areas the original biotic functions are largely destroyed, and the land has lost its productive capacity and is only reclaimed after major investments. Overgrazing of the vegetation by livestock led to decreases in soil cover that increased wind erosion.
3. About 165 million hectares of the Sahelian soils (74% of the total degraded soils) can be classified under light and moderate degrees of soil degradation. About 50% of which is due to loss of top soil by wind erosion and 38% by water erosion. If no restoration of this area is accomplished soon, part of it may become strongly degraded in the near future. Rehabilitation efforts may be justified to save these soils from further degradation. Ancillary benefits in terms of carbon storage would follow.

Keywords: Sahel, drought, land degradation, soil loss, impacts, pastoralist, dryland development

## Introduction

The Sahel, extends for approximately 7000 km across a continuous west-east belt of Africa and over 1000 km wide (See Figure 1). It covers 18 countries either completely or partially, and represents about 14% of the global drylands. The Sahel is most commonly associated with desertification, drought and famine issues, and therefore has become a classical example of desertification. The region includes the susceptible drylands of the ECOWAS countries, of Senegal, Mauritania, Mali, Burkina Faso and Niger which experience severe degradation problems, plus Chad and Central African Republic, and the members of IGADD of Sudan, Ethiopia, Eritrea, Djibouti. On climatic grounds it also includes northeastern Uganda and most of Kenya, but excludes the central and southern portion of the Ethiopian highlands, which though suffering from severe degradation problems fall into the humid category. Most of the Sahel receives its main rainfall in the summer months of June, July and August. It is brought in by moist winds from the adjacent seas, mainly the Atlantic Ocean.

Figure 1. Map of Africa showing the extent and severity of soil degradation in the Sahel (Source:UNEP 1992



## The History of Drought and Famine in the Sahel

The longer-term history of wet and dry periods of the Sahel reveals that the period 9500-4500 BP was a very wet one, while the period 4500-3300 BP was a dry one. Ritchie and Haynes (1987) speculate that the possibility existed that over-grazing in the region may have contributed to the expansion of the Sahara desert during that dry period. It is therefore reasonable to believe that at least part of the Sahel was subjected to recurring episodes of drought for the past millennia.

For the period 1510-1630, at least six famines have been recorded in Ethiopia. Two of these famines were attributed to locusts and three to drought. The great famine of 1888-1893 in Ethiopia was a combination of natural disasters-drought, rinderpest, locusts and caterpillars and war waged by Italy to subdue Ethiopia. Banditry, rebellion and tribal warfare also contributed to the famine. Throughout the 1880s and 1890s, war, serious drought, locust attacks and epidemics of small pox and cholera played havoc with food production in the Sudan. Conditions were greatly aggravated by the Mahdist War of 1896-1899.

The Sahel suffered several-year periods of drought a number of times during this century: in 1903-1905, 1911-1914, 1966-1974 and 1979-1987. A team from Lund University, Sweden examined the records of rainfall for central Sudan since 1900 and revealed that: (a) the period 1900-1920 had wide fluctuations in rainfall, including two or three brief periods of drought, with an indication of a rising trend; (b) the period 1920-1965 had generally above-average rainfall, with relatively minor year-to-year fluctuations and only three scattered years of below-average rainfall; (c) the period 1965-1987 had below-average rainfall, with a declining trend in 1987; and (d) the two years 1988 and 1989 had good rainfall, above average, but 1990-1991 were dry. These data show that rainfall in central Sudan has been below average during the past two decades.

In 1973, it became known to the world that a very severe social and environmental catastrophe of famine had struck throughout the Sahel. That catastrophe of starvation and death among people and their livestock had occurred as a result of a 5-6 year drought, during the period 1968-1973. It caused the death of perhaps 200 thousand people and of millions of cattle in the Sahel from the Atlantic Ocean to the Red Sea. Then drought struck once again, during the period 1979 - 1984 which gave the Western media rich material that it quickly took to the outside world.

The United Nations held its Conference on Desertification in Nairobi in 1977 to address the 1968-1973 episode. It followed it by another meeting in 1984 as a result of the 1979 - 1984 drought. In those meetings, it was acknowledged that the threat of desertification and related misfortunes were worse than ever, and that any attempt to attain sustainable development for such arid region should recognize the links between poverty, inequality and environmental degradation. The principal factors identified in desertification were: (a) natural vulnerability of the ecosystem; (b) over-exploitation of the resources; (c) economic mismanagement that leads to unsustainable land use; and (d) political unrest.

The droughts of the late 1970s to mid 1980s in northern Ethiopia and Sudan caused miseries to the peasant and pastoral lives. The first result was crop failure, and that resulted in major socioeconomic adjustments. Grain prices rose, livestock was sold at continually declining prices; migrations in search of food and jobs continued. Some poorer families even went to the abandonment of their dependents. Failure of the state to provide adequate relief in certain areas led to mass starvation, epidemic diseases and increased mortality. In 1984 in western Sudan, grain production was only 18-24% of the local need, and the livestock was sold at 25% of the pre-drought prices. Some nomads were unwilling to sell their herds, and consequently livestock mortality was massive. When famine reached its intensity, the result was mass population displacement. Thus, famine became not only a crisis for the stomach but also for personal identity. The 1984 famine in the Sudan was a great shock to a country accustomed to producing a surplus of sorghum and millet sufficient for export.

### Top Soil Loss and Land Degradation

Many of the upland areas of the Sahel, such as the northern Ethiopian Highlands are affected by very high severity water erosion. In Eritrea and northern Tigray in Ethiopia, soil erosion by water has reached critical levels, a consequence of deforestation of steep slopes (Table 1). In 1990 IGADD (The Intergovernmental Authority on Drought and Desertification) estimated that soil loss in the Ethiopian highlands was occurring at a rate between 1.5 and 2 billion $m^3$ per annum, with perhaps up to 4,000,000 hectares of the highlands irreversibly degraded (UNEP, 1992). FAO in 1987 reported that sediment rates in African rivers were 5.0% per year in Nigeria, 4.3% in Sudan and 3.2% in Tunisia. Data from studies by ODA (1991) in 1988-1990 report a five-fold increase in silt concentration in the Blue Nile originating from Ethiopian highlands compared to the 1930 levels.

The same data show that about 5 million $m^3$ of this silt settles in the canals of the Gezira irrigation scheme every year. Effective silt removal operations could cost as much as US$ 25 million per year.

Table 1. Sediment rates in African rivers (Source FAO, 1987)

| Country | Sediment rate % per yr |
|---|---|
| Nigeria | 5.0 |
| Sudan | 4.3 |
| Tanzania | 5.0 |
| Tunisia | 3.2 |
| Zimbabwe | 5.0 |

Studies in Sudan between 1988-1990 by ODA show a five-fold increase in silt concentration in the Blue Nile compared to the 1930 levels. About 5 million $m^3$ of this silt settles in the canals of Gezira Irrigation Scheme every year. Effective silt removal operations could cost as much as US$ 25 million per year.

Coastal plain and interior rangelands of Somalia are subject to severe wind erosion. Grazing pressure from sheep, goats, camels and cattle reactivates sand dunes and sand sheets in the coastal regions. The northern parts of the Sudanese provinces of Darfur and Kordofan are the site of much-quoted studies of desertification. The region suffers from very high wind erosion on the soils of ancient sand dune deposits. Here the causes can be attributed to overgrazing during times of drought. Some studies in Kordofan indicate that overgrazing has also caused declines in plant species diversity as well as vegetation coverage, with areas closest to centers of permanent population worst affected, highlighting the importance of the human element in land degradation (Ibrahim 1978). Soil compaction and crusting are the most serious forms of physical degradation affecting several irrigated areas of the Sahel. They have become a major problem in the Gezira, Rahad and Khasm el Girba projects in northeastern Sudan where vast areas of Vertisols are under irrigation.

The southern banks of Lake Chad suffers from similar compaction and crusting problems. The drought periods had also led to concentration of cattle around boreholes leading to soil compaction by trampling. In Sudan's Kordofan and Darfur Provinces, population pressures have led to tree clearance for the introduction of mechanized agriculture, primarily for growing sorghum. Mono-cropping has led to degradation of the sandy soils by nutrient depletion through over-cultivation, with substantial land abandonment occurring as short a time as three years after the start of cultivation.

About 224 million hectares (32%) of the Sahel is affected by soil degradation. 50% of the total degradation area is affected by wind erosion, and 38% is affected by water erosion (GLASOD, 1990). Major causes of soil degradation are overgrazing (53%), overexploitation of vegetation for firewood and other domestic purposes (24%), improper agricultural practices (16%), and deforestation (7%). Out of the total degraded area 74% is light to moderately degraded and 26% is strongly to extremely degraded. The degree to which the soil was degraded was related in a qualitative manner to the agricultural suitability of the soil, to the declined productivity, and in relation to its biotic functions (Tables 2 and 3).

Table 3. Main types of soil degradation in Africa (Source GLASOD, 1990)

| Type | Area affected (millions of hectares) | Area affected (% of degraded area) |
|---|---|---|
| Water erosion | 227.4 | 46 |
| Wind erosion | 186.5 | 38 |
| Chemical deterioration | 61.5 | 12 |
| Loss of nutrients | (45.1) | (8.8) |
| Physical deterioration | 18.7 | 4 |
| Compaction | (18.2) | (3.9) |

Table 2. Main causes of soil degradation in Africa (Source GLASOD, 1990).

| Cause | Area affected (millions of hectares) | Area affected (% of degraded area) |
|---|---|---|
| Overgrazing | 243 | 49 |
| Agricultural activities | 121.4 | 24 |
| Deforestation | 66.8 | 14 |
| Overexploitation | 62.8 | 13 |

The following four degrees of degradation were used:

1. Light. The terrain has a somewhat reduced agricultural suitability, but is suitable for use in local farming systems. Restoration to full productivity is possible by modifications of the management system. Original biotic functions are largely intact.

2. Moderate. The terrain has greatly reduced agricultural productivity but is still suitable for use in local farming systems. However major improvements are required to restore the terrain to full productivity. Original biotic functions are partly destroyed.

3. Strong. The terrain has lost its productive capacity and is not reclaimable at farm level. Major investments and engineering works are required to restore the terrain. Original biotic functions are largely destroyed.

4. Extreme. The terrain is unreclaimable and beyond restoration. It has become human-induced wasteland. Original biotic functions are fully destroyed.

It should be realized that these estimates are qualitative and cannot be directly translated into production losses or costs for rehabilitation. However, this approach may help policy-makers and decision-makers to better focus their attention to priority areas for immediate action.

Overgrazing is the most widespread cause of soil degradation in the Sahel affecting more than half of all degraded susceptible dryland soils. Overgrazing around settlements is often related to the sedentarization of nomadic herders. The settlement of these former nomads has meant that their herds have been concentrated onto grazing around their new homes. Drought conditions have also forced herders to concentrate their animals in areas where drinking water was available causing the complete disappearance of the herbaceous cover in many places, particularly around boreholes, with consequent windblown loss of topsoil and reactivation of ancient sand dune deposits.

Duststorms, and thus wind erosion, occur throughout the year, but are most intense in summer. Middleton et al (1986) estimated the removal of Saharan dust (a great portion of which comes from the Sahel) at about 260 million tonnes per year.

About 150 million tonnes sink in the Atlantic Ocean and as much as 50 million tonnes travel as far as Barbados, 5000 km off the African western shore.

Degradation due to poor agricultural management is largely concentrated in the semiarid and dry subhumid zones since these are the areas most suitable for dryland crops. Inappropriate management of dryland crops stems from a number of driving forces. Dryland farming has been extended into increasingly marginal areas since the nineteenth century when agricultural machinery was introduced which was inappropriate to the fragile ecosystem. Dry cereal cropland continued to expand into the steppe using tractors and multidisc ploughs with consequent degradation largely due to wind action in sandy soils, and hard pan formation in the heavy clays resulting in run-off and crop failure.

Deterioration of soil fertility had also been due to complete negligence of soil conservation measures such as shelter belts crop rotation and fertilizer application. The expansion of cultivation have pushed subsistence crops into areas whose fragility has been exposed by drought. Soil degradation due to the overuse of vegetation for such domestic purposes as fuelwood, fencing and construction is common to the central and western Sahel region very largely concentrated in the arid zone.

Land degradation via soil erosion begins with degradation of the vegetation. As the protective vegetation cover decreases, the top-soil becomes increasingly susceptible to wind and water erosion. The Sahel has been utilized for grazing, cultivation, tree cutting and bush fires for thousands of years, with overexploitation accentuated in times of drought. Unprotected clayey soil surface begins to form a crust as a result of trampling by the livestock. The ground becomes more and more subject to wind and water erosion, with a resulting loss in topsoil, nutrients and water. Abandoned fields that have been overexploited for crops or overgrazed become dominated by the shrub *Calotropis procera*. Not being palatable to livestock, these areas can be said to have undergone "green desertification".

### Impacts of Drought and Land Degradation

Rainfall in the Sahel is extremely variable both in time and space. Annual departures from mean rainfall values commonly range from 30 to 100 percent. Variability may be more of a constraint than low rainfall, longer-term climatic fluctuations overlay the pattern of extreme interannual variability. The rainfall pattern of fluctuation in the Sahel can be measured in decades. The result increase in spatial heterogeneity of soil reserve distribution is the downward spiral suggested by many as the "classic" process of desertification.

An additional pulse of moisture may increase shrub or annual grass biomass, but it is not enough to reverse the downward spiral. Studies of the Sahel based on satellite data show without question that vegetation cover has varied greatly during the past fifteen years in response to rainfall (Tucker et al, 1991). Although the observed changes can be correlated with rainfall, they relate to vegetation cover and not composition or quality. While most of the variation observed in the Sahel can be linked to rainfall, there is a large fraction (up to 17 percent) of unexplained variability that has increased with time and might be linked to deterioration resulting from land management practices (Hulme and Kelly 1993).

Finally, some recent extensive research done in the Sahel reports that degradation indeed has occurred, but the pattern is spatially complex. Rather than degraded, the Sahelian environment was actually quite resilient (Hellden, 1992, Olsson, 1993).

The drought of the current century led to substantial long-term site degradation in about 58 million hectares of the Sahel including depletion of the vegetation cover to the point of diminishing its ability to regenerate and of the formation of sand dunes. Moreover, a number of pre-drought changes in land use appear to have made the land more vulnerable to degradation, thereby exacerbating the drought damage that occurred. For example, growing poverty among certain tribes in the Sahel has for some time been forcing pastoral families to cultivate pastoral areas formerly covered with soil-binding vegetation.

Another result of growing poverty has been an increasing transferal of dependence from camels to sheep and goats. Camels permit a more mobile form of husbandry, less damaging to the soil, less dependent on a permanent source of water and more conducive to the maintenance of self-esteem.

#### a) Impacts on crop production

The Sahelian agriculture occupies around 10 per cent of Africa's drylands and supports about ten times the number of people that live under pastoral production systems. Population pressure and unreliable rainfall often combine to create acute food insecurity. In theory it could be said that if there were no people there could be no anthropogenic causes of environmental damage. Also, fewer people means availability of more resources per capita. Numerous local level case studies, however, suggest that relationships between population dynamics and environmental degradation are much too complex to support sweeping generalizations about cause and effect. Closer examination usually reveals that tenurial relations, the expansion of commercial farming, land alienation and various public policies have much more impact on environmental degradation than does population growth *per se*. It is not, therefore, the sheer numbers of people that determine environmental health, but rather how those people act within particular socio-economic and ecological contexts.

In many cases, population increase seems to have stimulated intensification of production activities and more careful land use practices. A recent study in the Machakos district of Kenya, sponsored by the World Bank, indicates that between 1930 and 1990 population growth was associated with many positive effects on the local environment, including the control of soil erosion on cultivated land, rehabilitation in grazing areas and increase in vegetation cover. The study notes that with a smaller population there would have been less labor available to carry out these production and environmental tasks (Tiffen, Mortimore and Gichuki, 1994).

The agricultural policies in a number of Sahelian countries have focused on large-scale agricultural ventures. Large-scale rain-fed mechanized agriculture has been established in eastern, central and western Sudan at the expense of grazing and forested lands leading to enormous expansion of the cultivated area, thereby achieving increased grain production through expansion of the cultivated area only, not through increased yields. The government policies have favored large-scale solutions, and virtually ignored the traditional sector, which employs about 70-80% of the rural population and produces a significant amount of food and cash crops (Larson and Bromley, 1991). Figures of food production in the Sudan show the ups and downs of food availability and reveal a fragile balance between production and need (Olsson, 1993).

b) Impacts on Pastoralists

Pastoralists are often better off than settled farmers during normal times, but are more vulnerable to adverse environmental conditions. In most of the countries in Sahelian belt affected by the 1983-84 drought, for example, pastoralists suffered more than other groups, and it took them much longer to recover from this crisis and reconstitute their stock. As a result of progressive land degradation in most of the parts of Western Sudan and elsewhere in the Sahel, invasion of grasslands by bushy plants such as mesquite and ushar (*Calotropis procera*) resulted in distinct changes in the species composition, and spatial distribution of vegetative cover. The decline in grass cover resulted in poorer carrying capacity and lower feed quality. Thus, although total plant biomass and cover may have remained about the same or even increased with shrub invasion, the spatial distribution of plants has changed significantly.

Many workers reported greater nutrient concentrations and biogeochemical cycling rates under shrubs than in inter-shrub gaps. By concentrating nutrients around shrubs, recovery of perennial grasses and other vegetation in intershrub gaps becomes less probable (Figure 2).

The pasture available to the herders is being gradually reduced by expansion in the crop lands creating bitter conflicts between herders and farmers. Nomadic grazing systems have evolved in response to erratic climates with limited rainfall. By moving from place to place in relation to the available resources the herders make optimal use of limited water resources and spread the impact of their activities more evenly over the environment. Pastoral communities require relatively large tracts of rangeland to permit grazing rotations and transhumance practices. The development of mechanized farming in the Sahel did not take into consideration the animal resources which were being gradually pushed out of their traditional grazing lands. This, together with the ever increasing livestock numbers, resulted in high pressure on the grazing lands, leading to overstocking and consequently overgrazing.

large tracks of land which used to be good grazing areas in eastern Sudan are now bare ground (Atta Elmoula, 1985).

Each new farm or plantation contributes to the relentless compression of grazing lands and reduces the herders options for mobility - a critical element for their survival as pastoralists. Pastoralists play a critical role in the sustainable management of fragile ecosystems. This need to be recognized. The key is to work with local people to develop policies to protect grazing lands and nomadic rights where pastoralism is both more sustainable and more environmentally land tenure appropriate than farming. Dry spells during the growing season and end-of-season droughts affect the crops, especially during the crucial grain-filling periods, resulting in serious yield reductions even at much higher annual rainfall. Nitrogen is the major limiting nutrient. Hence increasing nitrogen availability will increase yields of sorghum and millet. Phosphorus will also increase yields through increasing water use efficiency by crop plants.

Figure 2. **Model of Shrub Invasion in a Dryland Ecosystem.** (Source: Modified from Grover & Musick, 1990)

Drought and/or intensive grazing

Perennial grass cover depleted

Shrub invasion

Shrub competition

Selective livestock grazing

Fire suppression

Rodent/rabbit activity

Accelerated perennial grass decline

Increased wind erosion

Dune formation

## Conclusions and Recommendations

About 25% of the degraded soils in the Sahel is strongly degraded requiring the highest inputs for restoration, mainly as a result of wind and water erosions. Also as a result of overgrazing unpalatable or noxious shrub species have encroached the grazing land. Improper agricultural practices mainly land cultivation on sloping land without proper anti-erosion measures led to water erosion. Cultivation of land in areas with uncertain rainfall exposed soils during the fallow periods to the aggressive forces of wind and water. Use of heavy machinery on soils with weak structure stability (particularly Vertisols) led to soil compaction and formation of hard pan. Insufficient use of fertilizers and shortening of fallow periods for monocropping of sorghum, millet, cotton etc led to nutrient depletion particularly nitrogen, phosphorus and organic matter.

Ad hoc efforts to control land degradation have had limited success to date. Well planned, more integrated long-term national and regional land conservation and rehabilitation programmes, with strong political support and adequate funding, are needed to solve the land degradation crisis in the Sahel. While land-use planning and land zoning efforts of FAO and others, combined with better land management, should provide long-term solutions, it is urgent to arrest land degradation and launch conservation and rehabilitation programmes in the most critically affected and vulnerable areas in a holistic approach similar to that of the Landcare programme of Australia. Removal and resolving the physical, social and economic causes of land degradation, such as land tenure, pricing structures, incentives and, where appropriate and possible, resources for the participation of local communities in the planning, implementation and maintenance of their own conservation and reclamation programmes, should be encouraged. Periodic surveys to assess the extent and state of land resources and strengthening and establishing national land-resource data banks, including identification of the location, extent and severity of existing land degradation, as well as areas at risk, and evaluation of the progress of the conservation and rehabilitation programmes should be launched.

Agenda 21 recommended implementation of urgent direct preventive measures in drylands that are vulnerable but not yet affected, or only slightly desertified drylands, by introducing (i) improved land-use policies and practices for more sustainable land productivity; (ii) appropriate, environmentally sound and economically feasible agricultural and pastoral technologies; and (iii) improved management of soil and water resources. It further recommended that accelerated afforestation and reforestation programmes, using drought-resistant, fast-growing species, in particular native ones, including legumes and other species, combined with community-based agroforestry schemes should be carried out. In this regard, creation of large-scale reforestation and afforestation schemes, particularly through the establishment of green belts, should be considered, bearing in mind the multiple benefits of such measures. It also recommended urgent implementation of direct corrective measures in moderately to severely desertified drylands with a view to restoring and sustaining their productivity;

In areas with suitable soils and underground water, irrigation can be developed. Accumulation of salts in the soil surface should be avoided. Abandonment of grazing over areas can allow the vegetation to recover in areas heavily grazed and partially damaged by overgrazing. Reforestation of the degraded areas by plating trees and shrubs wildlife management, conservation of biotopes and a build-up of biodiversity suitable species stock be identified.

The wildlife species inhabiting the drylands are indigenous and well-adapted to cope with climate variability and frequent droughts. Ecotourism is an important non-consumptive use of wildlife especially in protected areas and in national parks. In Amboseli National park, for example, the annual gross financial return from Ecotourism yields US$1,500 per hectare. This is far more profitable than livestock production in these areas, where US$2-3 per hectare is estimated. A comparative assessment of wildlife and livestock as a multiple use option of rangeland in southern Africa indicates that many ranches are switching from livestock to wildlife.

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# Drylands Biodiversity: A Case Study From the Near East

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## Synopsis

This paper defines biodiversity and outlines its significance in the world's drylands, with particular reference to agriculture. The important problems leading to land degradation, desertification and loss of agrobiodiversity are listed and discussed. The significance of drylands *vis a vis* rainforests as a potential sink for carbon is recognised. Case studies from drylands within the Near East region are used to illustrate these points.


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## Key Points

1. In drylands productive land is scarce and human pressures threaten its stability. Loss of biodiversity via land degradation is high. Biodiversity is important for two distinct reasons: first, maintenance of the within-species genetic diversity of globally important food plants; second, the combating of degradation of these fragile ecosystems.

2. Only about 3000 of the estimated 250,000 species of flowering plants have been regarded as a food source. Others have provided forage or browse for animals which are hunted or farmed. Some 200 plant species are domesticated, but only about 20 species are of major economic importance. About 40% of the world's food dry matter is provided by crops which originated in these drylands.

3. Dryland species are highly adapted to environmental stress. This makes them a vital source of genetic material for improving crop varieties by increasing their drought and disease resistance. Dryland species are under threat from over exploitation of land and water resources. When a dryland plant or animal, or soil micro-organism adapted to dry conditions is lost, it is very likely that it lost forever. Because germplasm which is well adapted to the drier areas is relatively scarce, the loss is all the greater.

4. A key factor for sustainable development of drylands in the Near East region is the ability to conserve, manage and utilise the drylands biodiversity within the context of integrated natural resource management, in a way that can be both feasibly implemented and profitable to land users. Conserving dryland biodiversity is difficult if local needs are not met. Involvement of the local communities is the first step to promoting changes in land use practices and rangeland management systems that increase vegetative cover, conserve biodiversity and increase land productivity. A holistic approach to biodiversity, resource management and sustainable agricultural production is required.

5. Approaches to sustain acceptable ecological equilibrium must be based on understanding the mechanism of agro-ecological changes. Such understanding can improve our ability to attain a certain level of system sustainability.It is not realistic for a single institution to attempt to address the question of maintenance of biodiversity and natural resources on the scale envisaged in the Near East. A Regional Consortium, intended to link and build on existing biodiversity research and development projects, has a better chance of success.

Key words: Biodiversity, drylands, degradation, desertification, rangelands, Near East

## Introduction

In drylands, productive land is scarce and, just as in many other ecosystems, is under threat from excessive pressure on the land and water resources, deforestation, overgrazing and urbanisation. As a consequence of this, the relative loss of this globally important biodiversity by degradation is very high.

In its broadest sense, biological diversity or biodiversity, encompasses the variety and variability of all living organisms and their habitats. Agriculture uses only a small fraction of the biodiversity. It is presently based on the genetic resources of a handful of species and their wild relatives. The major shifts in biodiversity are driven by climate and land use change.

This paper demonstrates that biodiversity of drylands is as important as that of the tropical rainforests (the current focus of attention) and that, with appropriate national and international action, these drylands can be conserved, managed and protected against impending environmental degradation.

Loss of biodiversity deprives the world of biological resources that provide food and medicines for millions of people, of firewood and other sources of cultural objects so important for the maintenance of cultural integrity. When a dryland plant or animal, or soil micro-organism adapted to dry conditions is lost, it is very likely that it is lost forever, and because species well adapted to the drier areas are comparatively few, the loss is all the greater.

### Influence of desertification on the carbon cycle

Degrading dryland soils could also be a potentially significant source of carbon emissions; however, dryland soils are also known to be important storehouses of carbon. Indeed these dry rangelands are a major sink for carbon, and have the potential to sequester 0.5 to 1.0 giga tonnes of carbon annually (Squires and Glenn, 1995). Furthermore, the potential for increasing the carbon storage is greater in drylands than in agricultural and forested lands where carbon levels are near saturation (Ojima, Stafford Smith and Beardsley, this volume). The implications for combating soil degradation, particularly in rangelands, are therefore enormous.

The impact of human-induced changes in drylands on the earth's atmosphere and global energy balance have begun to be modelled with some success. As scientific research advances, the linkages between climate change and dryland degradation will become much clearer.

### Sustainable Development of Drylands

A key factor for sustainable development of drylands in the region is our ability to conserve, manage and utilise the drylands biodiversity within the context of integrated natural resource management, in a way that can be both feasibly implemented and profitable.

Dryland wildlife, when properly managed, can provide a source of livelihood to rural communities and also help conserve biodiversity. In contrast, intensive pressures of people on land and improper resource use deplete wildlife and biodiversity and cause economic loss. A sizeable part of tourism and trade in Eastern and Southern Africa is based on dryland wildlife, which is under stress. In West Africa, in Mali, the Baoule National Park and Biosphere Reserve is affected by overgrazing by transhumant livestock, burning of vegetation by herders and settlers, and poaching.

Limiting access to protected areas increases pressure on other land. Conserving dryland biodiversity is difficult when local needs are not met. The GEF-funded project "Community-based Rangeland Rehabilitation for Carbon Sequestration and Biodiversity" in Sudan is an excellent example of a project that will promote, with the involvement of local communities, such land-use practices and rangeland-management systems that increase vegetative cover, conserve biodiversity, and increase land productivity.

This paper will give a brief glimpse of three types of dryland ecosystems in the Near East and the possible means to rehabilitation, maintenance and conservation, through the development of a regional UNEP/UNDP/GEF project and consortium.

## CASE STUDY: THE DRYLANDS OF THE NEAR EAST

There are three principal land uses in drylands; irrigated farming, rain-fed agriculture and pastoralism. We will give brief examples of all three from the drylands of the Near East.

### The Environment

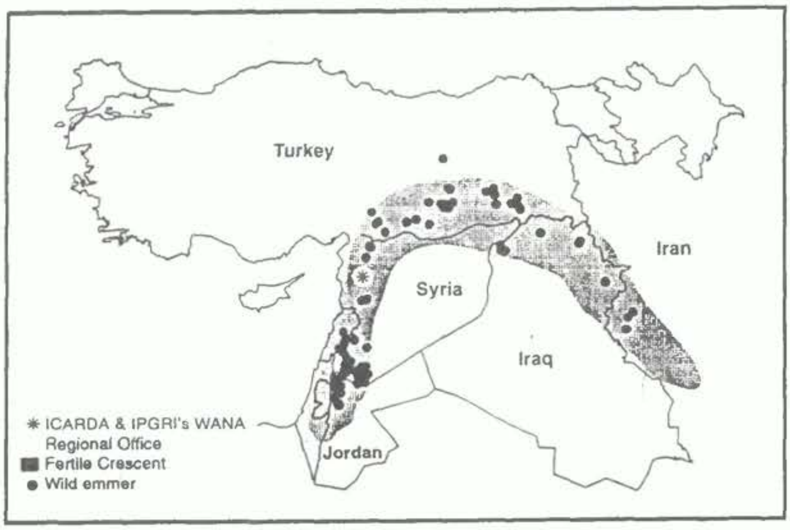
The Near East is generally taken to embrace the geographical area east of the Mediterranean Sea, bounded by the Taurus and Zagros Mountains to the north and northeast and the Sinai Peninsula in the south. The drylands cover about 10 million km$^2$ (60% of the total land). Elevation ranges from about 3000 m above sea level to 400 m below sea level at the Dead Sea. Geologically, the area is marine in origin and the soils, formed from limestone, are generally calcic ( Figure 1).

The region has a typical Mediterranean-type climate with cold winters with rainfall and hot, dry summers. East and south of the bordering mountain chains rainfall declines from approximately 600 mm, initially at a rate of about 3 mm per kilometre, to less than 50 mm in the southeast (giving a sweep of relatively well-watered land in the west and north of the region from which the term the 'Fertile Crescent' derives). This spatial variability in rainfall is accompanied by a high degree of variability in time, both between and within rainfall seasons.

Variability in temperature is associated with the topography, with significant fluctuations from year to year. Within this variability several broad dryland agroecosystems-ecosystems can be identified. As rainfall declines, cropping systems dominated by wheat give way to barley-dominated systems which in turn merge into rangelands and ultimately to desert. Temperature- and soil-related characteristics, and the availability of water for irrigation, add complexity to these very broad categories.

The region supports 529 million head of livestock composed of sheep (47%), goats (33%), cattle (18%) and camels (2%). The total livestock increased 32 % during the period 1975 to 1992. These livestock, mainly resident or transhumant flocks, form an integral part of the ecosystems and are an integrating force among them, as they use low-quality roughage and crop residues and form a link in nutrient cycles. The main sources of feed are natural grazing, stubble, straw, failed crops and fallow grazing. Concentrate feeds, mostly imported, contribute 20 to 50% of all animal feed and is increasing. Almost all goats and camels, 70% of the sheep and about 50 % of the cattle depend on rangelands. Now there is so much pressure on rangelands that many annual plants are consumed before they flower.

Figure 1. The fertile crescent, location of ICARDA and IPGRI's WANA Regional Office, and geographical distribution of wild emmer, the progenitor of durum and bread wheat. The centre of plant domestication and genetic diversity




The Near East is recognised as a centre of genetic diversity, plant domestication and one of the three nuclear centres of agricultural origin, defined by Hawkes (1983). This area corresponds geographically to a region known as the Fertile Crescent, a semicircle, which extends from Palestine through Syria, southern Turkey into Iraq and western Iran.

The early plant gatherers found the local plants a convenient source of concentrated energy and protein, which could be easily carried and stored. Archaeological evidence documents that wheat, barley and, to lesser extent, lentil, pea and other food legumes played an essential role in the rise of the great Near East civilisations. Contemporary geographical distribution of their wild progenitors corroborates the archaeological evidence. For example, a map of the present geographical distribution of a wild progenitor of emmer, durum and bread wheats, *Triticum dicoccoides*, coincides with the geographical limits of the Fertile Crescent (Figure 1). Wheat and barley were the two staples which started food production in the Near East. These two crops together with the domesticated sheep and goats were the basis of a farming system that evolved in the Fertile Crescent in about 7000 BC and which spread as a Neolithic agricultural package quickly from the nuclear region to other parts of West Asia, to the Nile valley and the Balkans. Harlan (1992) provides a short list of 80 cultivated plants or plant genera originating in the Near East, Mediterranean or West Asia.

Today it is estimated that 38% of the world's food dry matter is provided by crops which originated in the semi-arid regions of the Near East (Harlan, 1992). Wheat alone makes up one-third of the global food production with 580 million tonnes. It is now widely recognised that the wild progenitors and relatives of these crops have accumulated, during their long existence, a rich reservoir of genes for adaptation and survival to the harsh natural environment.

Many useful genes have already been transferred from the crop wild gene pool to the cultivated species (Plucknett *et al.*,1987), especially those related to biotic and abiotic stress tolerance. In future, the rich intrinsic genetic diversity may be indispensable in breeding crops adapted to climatic changes. In spite of the recent advances in biotechnology making gene transfer between unrelated organisms possible, the wild progenitors and closely related species of the primary gene pool (Harlan and De Wet, 1971) are still, and will remain for the next few decades, the most feasible wild source of genes for conventional breeding programs, provided they still exist.

## The threat to biodiversity and sustainable agriculture

The Fertile Crescent was one of the major food-producing regions of the world for many centuries; indeed, it was once termed the granary of the Roman Empire. Today, with a population of some 70 million, which is increasing at a rate of around 3.5% per annum, it has become a food-importing region. This rapid increase in the human population, and the accompanying increase in demand for food and the need for housing, jobs and infrastructural development, has, as elsewhere, had an indelible impact on the natural ecosystems of the region. For a majority of the population in this region, agricultural production, including pastoralism, is the principal economic activity. Agricultural land use has been intensified and expanded in efforts to meet national aims of food self-sufficiency, placing huge demands on the natural resource base, and leading to severe degradation of vegetation, land and water sources.

Degradation of biodiversity is attributed to the destruction of natural habitats and over-exploitation of native vegetation. But, in the Fertile Crescent, those same habitats represent the resource base for productive agriculture and are often the sole livelihood of farmers and pastoralists. Most land in the region, with the exception of the desert areas, is used in some manner for production, for example, extensive livestock grazing in steppe lands.

Degradation of natural vegetation is apparent in virtually all parts of the Fertile Crescent. Areas, which in historical times were covered with extensive forests of oak, juniper, pistachio and almond, are now treeless expanses of field crops where, in the dry summers, dust clouds rise as topsoil is eroded by wind (Bobek, 1968; Pearse, 1971).

The herbaceous vegetation of wetter areas is chronically overgrazed, especially in spring when it is the main feed source until crop residues become available after harvest in early summer. Intensive grazing at this time means that seed production of the most valuable species is reduced, plant populations, number of species and consequently vegetation cover in succeeding years are severely reduced. This leads, in turn, to increased runoff of rainfall with consequent soil erosion -- the productive potential of the land is permanently downgraded.

Traditionally, utilisation of the vegetation of the drier areas took place under the tribal 'hema' system which restricted the timing, frequency and intensity of grazing, and was instrumental in the maintenance of the rangeland (Jaubert and Bocco, 1994). Social change has led to a breakdown of tribal structures, and the abandoning of controls on the use of steppe vegetation. Government policies aimed at increasing meat production have supported the provision of subsidised feedstuffs, barley grain and industrial crop by-products, and, with modern transport facilities, feed and water are now transported to permanent populations of people and livestock in the steppe areas (Leybourne, 1994). This has lead to severe degradation of the vegetation through overgrazing and the use of shrubs for fuel. As in the wetter areas, the reduction in the vegetation has caused more runoff of the sparse surface water and an escalation of soil erosion by both water and wind as large areas are laid bare.

Much of the degradation must be ascribed to land clearing with modern machines (Valkoun and Damania, 1992) which has allowed the cultivation of areas that previously were too steep, too rocky, or simply too vast to be farmed. In the higher rainfall areas there has been extension of cultivation on steeper hillsides. Upper slopes and hilltops, often planted with orchards, are now ploughed deeply and regularly; the practice is of questionable value to the productivity of the trees, but has been widely adopted. It renders soil vulnerable to water erosion, and destroys habitats and species within habitats. In the drier areas, vast tracts that previously carried natural vegetation are now sown annually with barley and produce sparse crops as a feed source to substitute for the degraded rangeland.

In summary, the natural vegetation of the region is being massively eroded through the degradation of natural habitats, intensification of the cultivation of arable lands, the expansion of cultivation into previously uncultivated and marginal areas, the replacement of landraces by new cultivars, and the ubiquitous overgrazing of natural pastures and rangelands by small ruminants.

### Irrigated areas within drylands

An important contribution to world food production comes from irrigated lands in the dry, semi-humid and humid tropics. Traditional practices e.g. paddy rice, have been maintained successfully over millennia in Asia, but recent large-scale expansion of irrigated agriculture in Asia and Africa has met with increasing problems, as waterlogging and salinity or alkalinity have built up. Today, about 43 million hectares of irrigated lands or 30% of the total area of the world's drylands (145 million ha) are adversely affected. (Tolba *et al.,* 1992). In the Near East the situation is critical and our case study is from the West Bank and Gaza.

Although only 10% of the whole Territory (5% in the West Bank and 60% in Gaza) of Palestine is irrigated agriculture, this type of cultivation, practised by both Jewish Settlers and by some Palestinian farmers, could damage long-term sustainability.

Intensive discharge of ground water and use of fertilisers, pesticides and other chemicals, present a threat to biodiversity as they are hazardous not only to the soil, but to all plant species and wildlife. All these practices contribute to the pollution of Palestinian natural resources including surface and ground water, land and air -- this has a negative effect on overall biodiversity. Furthermore, because of the highly productivity nature of this type of agriculture, there tends to be a uniformity to the type of seed stocks used, and a tenancy toward hybrid exotic gene stock in place of local seeds.

One of the other problems with the intensive irrigation and water use that has taken place since the middle of this century has been the deterioration of the quality of surface water and ground water sources throughout Palestine. One of the best examples of this is the Jordan River. The once mighty river that flowed from Lake Tiberias into the Dead Sea has now been reduced to a fraction of its original size through the siphoning off of water at the river's mouth. Furthermore, largely because of runoff from sewage and the high-input farms in the Jordan valley, the water of the Jordan river is now highly salinized, so much so that it is unsuitable for irrigation. Clearly this has a negative effect on the traditional plant life along the river banks

Water has always been a precious resource in the Middle East. Irrigated agriculture has been an important part of the agricultural production in the Palestinian Territory (West Bank and Gaza) for more than 2000 years. The ancient civilizations of Jericho, for example, existed largely off crops grown under irrigation from the surrounding springs. The Nabatians in the Negev Desert also developed systems for collecting and diverting runoff water for agriculture. These early irrigated systems were able to flourish, not afflicted by the problems of over-population, shortage and deterioration of water resources, and land shortage and degradation (Hauser and

Jones, 1988; Perrier, 1988).

However, in the 1950s, with rapid technological development and advancement following World War II, the Middle East in general underwent an enormous expansion in agricultural and irrigated area, complete with subsidized water usage to encourage production (Allaya, 1994).

Most of the water resources are shared between the Palestinian Territory and its neighbors. The Jordan River is fed by Lake Tiberias and the Yarmouk River in Jordan, which is in turn fed by the Golan Heights. The Coastal and Mountain Aquifer basin is shared between Israel and the Gaza Strip and is fed by water from the West Bank (Baida, 1986; Shawa, 1994). In the absence of a political agreement on these issues, the allocation of these important resources has been determined to date by virtue of force -- who is able to control the water resources. Clearly this is not desirable if the goal is sustainable use of water resources (Bingham *et al.,* 1994).

While great steps were made in establishing intensive irrigated agriculture and increasing production in these arid areas, with 50 years of hindsight, we can begin to see that this great expansion of irrigation has had negative effects in this region of limited water resources. Likewise, while few studies have been done, indications are that the water quality of the West Bank aquifer in the Jordan Valley is less than adequate for human consumption and is diminishing. This is certainly due to a combination of the sinking of the Jordan River and over-pumping of the aquifer itself causing a water crisis that has afflicted the majority of the West Bank. Indeed a satellite photograph can almost perfectly map the political boundaries of the West Bank and Gaza, simply by showing the driest areas of the land's central region from the Mediterranean to the Jordan River.

The water crisis story in Gaza is well known. While the cause remains controversial, the urgency of the situation is undeniable. After years of over-pumping to supply irrigation both in the Gaza Strip itself, and probably in the Negev Desert, the levels of the aquifers have dropped below sea level and Gaza is now experiencing severe salt water infusion, making most of the water brackish and unusable for either domestic or agricultural purposes. The level of total dissolved solids in water samples taken from wells in Gaza (excluding Israeli settlements) ranges between 1000 and 3000 parts per million (p.p.m.) with an average of 1400 p.p.m., compared with a safe drinking water standard of 500 p.p.m. and an acceptable level for irrigation of 1000 p.p.m. This situation may well have been exacerbated by the siphoning of water from the Hebron mountains that previously fed "Wadi Gaza" providing recharge for the Gaza aquifer (Shawa, 1994).

Other negative effects of irrigated agriculture are due to the fact that relatively more intensive inputs are used in this type of system. On average, Palestinian farmers growing irrigated vegetables use between 750 kg and 1500 kg of nitrogen per hectare, while use by farmers growing rain-fed vegetables is almost negligible. The same applies for the use of pesticides which are both highly toxic and environmentally harmful (Safe *et al.,* 1991). This situation is even worse in the coastal plain of Israel, where the water in the underground aquifer basins is so polluted from years of intensive irrigated agriculture that Israeli water experts say it is near brackish (Gobbay, 1991). Furthermore, many of the crops used in this kind of agriculture are hybrids based in exotic landraces, rather than the local varieties prevalent in non-irrigated farming. Clearly the ideal system will incorporate both wild and exotic races, but too little research has been done on the races in use in West Bank irrigated agriculture.

While certainly the policies of encouraging the expansion of irrigated agriculture here increased production in the short term, it is not sustainable. The best estimates of available water resources compared with projected population growth indicate that by early in the next century fresh water resources will all need to be devoted to domestic purposes. Irrigation will have to come almost entirely from treated waste-water. On the other hand, recent research aimed at improving the sustainability, production and status of rain-fed farming in the West Bank has shown significant improvements in crop yields through improving the selected varieties, techniques of crop rotation and fertilisation, and pest control.

### Rain-fed Croplands Within Drylands

Rain-fed agriculture in drylands covers nearly 36% of the total dryland area in the world and it is being lost through degradation processes at about 3.5-4.0 million hectares a year (Tolba *et al.,* 1992). This rate of degradation is highest in the Near East. Rain-fed agriculture (approximately 68% of the total arable land) expanded from 62 to 80 million hectares from 1980 to 1990, mostly by encroaching on prime rangelands. However, various grades of mutual dependence exist between range and rain-fed land production systems. Expansion was the result of the growing need for cereals and was accelerated by the increased availability of tractors. Crop yields are low (700 kg/ha) and crop failures are common.

Typical examples of this come from our case study in the low (100-200 mm per annum) rainfall (Steppe) zone of Jordan. Recent information indicates that the land resources of this zone will not be adequate to provide more than 14% of the food needs of Jordan by the turn of the century if no additional resources are made available. Unfortunately, any expansion in land resources requires additional water resources, which are either diminishing in quantity or suffering from competition for human use.

Several studies indicate that climatic changes took place 5000 to 10,000 years ago and were followed by more humid climate. It should be noted that the climatic changes and subsequent soil- plant interactions occurred gradually. Field observations in this steppe zone indicate substantial accumulation of calcareous silt. Such sediments reduce the soil water-holding capacity, create drought conditions, and force several more plant species out of the ecosystem.

The deposition of such sediments occurred at the same time as the climatic changes. As the silt content of the soil increased, especially within the surface layers, drought conditions became more prominent. This accelerated the erosion processes and soil was transported toward lower areas. Thus, deep soil and better conditions for plant growth are found on more level areas, generally along wadis where water usually accumulates after rainfall. Steeper slopes usually have shallower soils, and conditions harsh for plant growth dominate such landscapes.

Ecological fragility paved the ground for pronounced human interference which has intensified during the last 5-6 decades. Human interference in this region became more visible as a result of population increase, intensive grazing, heavy traffic movement after the establishment of new urban settlements, indiscriminate ploughing and the cutting of shrubs for fuel. Such activities have converted many parts of this region, described as forest 1000 years ago, to totally treeless areas or to areas deprived of seeds.

The area should now be used as range; however, irrigated farming is practised wherever ground water is available and in the west portion, where rainfall is 200 mm, wheat or barley is now being cultivated. Historically, good yields were obtained only once in every seven years and this practice is accelerating the degradation.

Nevertheless, recent studies have indicated that the soil of this region has good potential for recovery. Monitoring the development of plant cover indicated that the number of plant species increased many folds within three years when protection only was provided. Moreover, it has been demonstrated that on steep slopes with shallow soil (30-40 cm deep) it is possible to support an acceptable plant cover, if water balance is improved through water harvesting. Spots where no visible vegetation was observed earlier have regained excellent natural grass cover within five years.

The improvement of rain-fed farming has been more encouraging on deeper soils where supplemental irrigation was provided or within self-flooded wide waterways called locally "Marab". In both conditions recovery of grass species was spectacular and the production of fruit trees is sustainable.

Approaches to sustain acceptable ecological equilibrium must be based on understanding the mechanism of agro-ecological changes. Such understanding can improve our ability to attain a certain level of system sustainability. The on-site conservation of endangered plant species, as well as the protection of other species, provide good development opportunities that should not be underestimated.

It is a national priority to implement an integrated ecosystem management which maximises and develops non-traditional water resources. It should also aim at preservation, improvement and management of land resources. It is expected that such an approach will encourage measures for intensive conservation and introduction of wide range of native plant, and economic crops, thus addressing national needs for successful biodiversity conservation. The clear economic benefits from such activities will have a significant positive impact if local communities participate fully in the implementation of the program.

## Rangelands within drylands

Seventy three per cent of the world's rangeland is affected by desertification, mostly vegetation loss by overgrazing. Almost a quarter of the affected area is accompanied by soil degradation, mostly erosion by wind and water (Tolba *et al.,* 1992). One of the largest areas of rangelands affected are those of the Near East (Table 1) where the flora and vegetation is also rich (Table 2), containing about 2500 species (Le Houerou and Boulos, 1991).

Range degradation has far-reaching environmental, economic and social consequences on the countries of the region. The loss of vegetation, soil and water resources leads to increased destabilisation of the fragile ecosystems in the arid and semi-arid rangelands. Range degradation not only decreases biological productivity, but has far-reaching ill effects on the overall environment. Declining freshwater resources, increased soil salinization, sand and dust storms are the most evident. Declining returns from range/livestock result in increased poverty and migration to urban areas and encourage farming of marginal areas which in turn increases risks of further degradation. However, the grazing of sheep and goats is one of the few ways to use these lands, short of quarrying the rock, mining for minerals or expanding the urban zones to build upon them.

There are millions of hectares in the Near East where cereals cannot be grown but which still receive between 200-600 mm of annual rainfall (Table 2). This is particularly the case in Syria and this case study will focus on the regeneration of some of these areas with annual legumes.

Table 1. Approximate surface areas of arid and desert rangelands in the Near East (in 1000 $km^2$).

| Zone/ Country | Total area | Non-arid non-desert (P>400 mm) | Arid (400>P>100 mm) | Desert (100>P>50 mm) | Desert wasteland (50>P mm) |
|---|---|---|---|---|---|
| Egypt | 60 | - | 5 | 35 | 20 |
| Iraq | 435 | 48 | 291 | 96 | - |
| Israel | 21 | 9 | 5 | 3 | 4 |
| Jordan | 98 | 15 | 40 | 25 | 18 |
| Kuwait | 18 | - | 18 | - | - |
| Oman | 212 | 2 | 12 | 91 | 107 |
| Qatar | 22 | - | - | 22 | |
| Saudi Arabia | 2150 | 10 | 200 | 1240 | 700 |
| Syria | 185 | 18 | 157 | 10 | |
| Turkey | 781 | 671 | 110 | - | - |
| United Arab Emirates | 84 | - | - | 84 | |
| N. Yemen | 195 | 30 | 140 | 25 | |
| S. Yemen | 332 | 10 | 20 | 232 | 70 |
| Near East | 4593 | 813 | 998 | 1863 | 919 |
| Arid and desert rangelands | | | 2861 | | |

(Adapted from Le Houerou & Boulos. 1991). P= rainfall

Table 2. Flora of Mediterranean-steppic region and the Arabian Peninsula; approximate numbers and areal richness index (log species/log area). (Adapted from Le Houerou & Boulos, 1991)

| Area/ Countries | Area ($1000 km^2$) | Number of species | Number of species per ($1000 km^2$) | Areal richness index |
|---|---|---|---|---|
| Mediterraneo-Steppic Zone | 1448 | 3650 | 2.52 | 0.58 |
| Near Eastern Steppes | 843 | 2000 | 2.37 | 0.56 |
| Palestine (Israel + Jordan) | 70 | 990 | 10.95 | 0.61 |
| Syria | 140 | 1200 | 8.56 | 0.60 |
| Iraq | 291 | 1200 | 4.12 | 0.57 |
| Kuwait | 18 | 250 | 13.90 | 0.56 |
| Oman | 212 | 1000? | 4.70 | 0.56 |
| United Arab Emirates | 84 | 300? | 3.57 | 0.50 |
| Saudi Arabia (Steppe zone only) | 220 | 750 | 3.41 | 0.54 |
| Saudi Arabia (Whole country) | 2150 | 2200 | 1.02 | 0.53 |
| Sahara | 8000 | 2800 | 0.35 | 0.50 |
| Yemen (N and S) | 507 | 1500? | 3.00 | 0.56 |
| Arabian Peninsula | 3013 | 3000? | 1.00 | 0.54 |

The conservation and collection efforts of the past several decades, to save at least a part of the remaining genetic diversity in drylands, have netted a formidable assortment of accessions from hundreds of species. For example, at ICARDA alone there are holdings in the following genera of 7888 *Medicago* (85 taxa), 3405 *Trifolium* (66 taxa), 1849 *Lathyrus* (56 taxa), 5481 *Vicia* (101 taxa) and 8337 accessions from an additional 196 taxa.

There is still hope but it is a guarded optimism for many reasons. The main concern is that many plants are adapted to very specific niches and for that reason commercial seed production and resowing will never provide a complete solution.

An examination of the flora reveals that many of the native forage grasses and legumes can survive in this harsh environment and prosper with a minimum of management, provided that grazing is properly managed during flowering. Even the seed banks of hard-seeded annual legumes have been gradually reduced over the centuries by overgrazing. Unfortunately, the recent generations of farmers of the Near East have come to accept the appearance of degraded lands and they have lost memory of a more productive era. However, we hope this will change.

The use of ecogeographic surveys must serve as a guide to revegetation efforts, focusing on plants that are already adapted (Ehrman and Cocks, 1990). Once this information is known, then a next step must be to increase the seed of these useful species.

We have started on the annual legume species which have very small and hard seeds. The reason for this strategy is simple — more than one-half of the seed of annual legumes such as *Trifolium campestre* or *T. tomentosum* pass through the digestive tracts of sheep or goats (Thomson *et al.*, 1990). The annual legumes also produce seed with high levels of hard seededness, or impermeability of the seed coat. This permits these species to persist over many years in the soil seed bank (Russi *et al.*, 1992 a, b) and is vital in our plan to revegetate and make better use of degraded rangelands in the future. It should also be noted that the within- and between-species genetic diversity of herbaceous pasture legumes for our target area is high and their agricultural exploitation is low.

The next step is to concentrate on prolificacy of seed production so that the plants in the natural environment can create a community effect through dense populations of seedlings. It has been demonstrated with annual medics in Syria that there is a positive relationship between dry matter yield and seedling density (Abd El Moneim and Cocks, 1986).

After establishment on a rehabilitated site, the sheep can be used to spread the seed further. During the period when the annual legumes and their ripened seeds are exposed for grazing, the flocks should alternate between the source of seed on a rehabilitated pasture and the target pasture that needs rehabilitation. These principles are simple and have been successfully followed by farmers in Syria. What is needed are more demonstrations of the principles in the field (Osman and Cocks, 1992; Osman *et al.*, 1991, 1994).

During 1994/95 in El Bab, near Aleppo, Syria, an area with 266 mm of annual rainfall, ICARDA scientists have demonstrated some of these principles to the farmers and revegetated over 100 ha of marginal, uncultivated grazing land in four separate villages using oversowing with adapted clovers, medics, phosphate fertiliser and deferred grazing in the spring of the establishment year. Seedbanks are being monitored with very positive results.

This management still requires that someone create the supply of these local seeds. We must also now overcome negative experiences from the past, such as importation of exotic seeds which fail to work in local ecosystems due to lack of cold tolerance (Cocks and Ehrman, 1987). Inexpensive, mechanised methods are needed for the production of locally adapted pasture seed which can be used by the farmers (Christiansen, 1993).

Each plant has its own characteristics for shedding seed from the plant and there mechanisms must be understood for use in seed production systems. Table 3 shows that the medic species drop their pods and can be easily harvested by sweeping. On the other hand, *T. campestre*, *T. scabrum* and *T. lappaceum* retain their seed in the heads, making it possible to cut them at harvest and carry them to a thresher. Other species such as *T. stellatum* and *Scorpiurus muricatus*, although valuable, need more operations and are therefore more expensive to harvest.

Annual legumes also vary widely in the yield of seed they produce (Table 3). Each of these practical considerations must be considered when transferring seed increase techniques to farming communities.

The principles described are well tested and ready for wide-scale technology transfer to even drier rangelands, such as those in the Syrian steppe. Unfortunately, the current practice of open access and lack of ownership limits the scope of practical intervention. Until such a time that new policies are developed to correct the problems of proper land stewardship in the steppe, we will focus on the villages where control of grazing by sedentary communities can be exerted and rangeland rehabilitation can be successful.

## GENERAL SOLUTION

### A holistic approach

Within each of our three farming systems we have described a holistic approach to biodiversity, resource management and sustainable agricultural production is required. There is a grave risk that much of the inherent biological diversity of the Near East will be lost, unless a holistic approach to the management of these dryland ecosystems is developed. Sustainable development depends upon the conservation of resources (land, water and genetic) upon which agricultural production is based. At the same time, to be adopted, any strategy for the conservation of natural resources (including biodiversity) must be directed towards sustainable, but profitable agricultural production.

By a holistic approach we recognise that all of the ecosystems we have described have been, and remain, subject to immense human interference. Local communities are major components of these ecosystems which they both inhabit and use for agricultural production and pastoralism. Long-term improvements to the management of the natural resources within these ecosystems can be achieved only with the consent and active participation of those users, who need to be convinced that any changes, sacrifices or inputs required from them are ultimately to their benefit (see also Swallow, and Migongo-Bake, this volume).

A 'bottom-up' approach is therefore essential: land-users (and local and national organisations with an interest) are partners. Each has been consulted about the nature of the problems. It is essential that their ideas for solving the problems, and their opinions of the researchers' ideas are utilised. Subsequent research and development work must involve the active participation of these land-users within the following areas of work:

i. **Conservation of plant genetic resources.** Productive land is very scarce, therefore any large exclusionary 'reserves' to conserve biodiversity, which remove land from productive use, are unlikely to be popularly accepted; nor is exclusion necessarily the best means of conservation: for many species, active management may be required to preserve their diversity. Reserves may also need continual national or international support, the antithesis of sustainability. Consequently, existing traditional and multiple land uses, and the livelihoods that they support, must be taken into consideration, and ways devised to build niches for the maintenance of biodiversity in coexistence with sustainable land-use practices.

ii. **Germplasm** collection should continue with renewed urgency, particularly in endangered environments and special protection must be sought for endangered habitats of particular importance. Numerous collection missions have been conducted, mostly by joint teams of scientists from international and national organisations. For example, ICARDA has conducted, in collaboration with national programs, more than 60 missions for collecting and saving drylands biodiversity of West, Central and South Asia and North Africa. This effort yielded 20,000 new germplasm samples, now maintained in the Centre's gene bank, one of the world's largest *ex-situ* collections of food crops wild relatives of Near East origin. The *ex-situ* conservation of genetic resources has its limitations, especially regarding wild species (Frankel, 1975; Plucknett *et al.*. 1987). Additionally, our results suggest that the very high genetic diversity encountered in some natural populations of wild wheats cannot be adequately sampled and maintained as a bulk in a gene bank.

However, the longer view must surely be to seek to protect -- and to inspire and assist local human communities themselves to protect -- the whole landscape, particularly the basic physical resources of soil and water, upon which the natural flora, crops and animals depend. Therefore, a complementary method, *in-situ* conservation in the original habitat, is recommended for wild species, including crop progenitors (Ingram and Williams, 1984).

ii. **Utilisation of plant genetic resources.** The utilisation of indigenous genetic diversity in germplasm enhancement will contribute to the maintenance of diversity in the region's cropping systems. Landraces, selected by generations of farmers, are an invaluable source of specific adaptation to difficult environments. This makes them ideal germplasm for use in breeding programmes aimed at adapting crops to specific environments, where they can be used as recipients of genes that impart desired characteristics without disrupting their adaptation. The use of local landraces in decentralised breeding programmes can be expected to both speed the rate of crop improvement for harsh environments and maintain diversity as a safety measure against the breakdown of specific resistances to biotic stresses. If coupled with a strategy to involve farmers directly in selection, this approach will also ensure that the needs of the end-users of new cultivars are met, thereby ensuring rapid technology adoption and enhanced benefits.

Productivity of food crops is being increased by the adoption of improved cultivars and improved agronomy. Provision of alternative feed sources for livestock to relieve the pressure on natural grazing areas is possible through the use of forage crops which can be incorporated into the systems by replacement of fallow or continuous cropping. Its replacement with forage crops increases the efficiency of water use for biomass production. If legumes in association with appropriate $N_2$-fixing rhizobia, are used to replace fallow they provide good quality feed for livestock and also contribute to the maintenance of soil fertility and organic matter. An increase in the diversity of species within the cropping system is one aspect of biodiversity through which more productive and sustainable cropping systems can be achieved. Furthermore, such production gains, when well understood by land users and integrated into their systems, will encourage better management of the natural vegetation.

**Table 3** Estimates from ICARDA's Terbol Station, Lebanon , showing successive pod or seed harvesting operations and the percentage of seed obtained with each technique.

| Species in groups based on harvest characters | Cut and remove seed with straw | Remove straw then sweep pods or heads | Hand harvest remaining seed | Vacuum remaining seed | Difficulty (1=easy; 5 = hard) | Max kg seed/ha |
|---|---|---|---|---|---|---|
| | (Percentage of seed harvested in sequential operations) | | | | | |
| *Trifolium speciosum* | 85 | 0 | 10 | 0 | 1 | 400 |
| *Trifolium campestre* | 85 | 0 | 10 | 0 | 1 | 400 |
| *Trifolium scabrum* | 75 | 10 | 10 | 0 | 1 | 400 |
| *Trifolium lappaceum* | 85 | 0 | 10 | 0 | 1 | 600 |
| *Medicago ridigula* | 0 | 85 | 10 | 0 | 2 | 1200 |
| *Medicago noeana* | 0 | 80 | 10 | 0 | 2 | 800 |
| *Medicago rotata* | 0 | 80 | 10 | 0 | 2 | 1000 |
| *Trifolium pilulare* | 60 | 25 | 10 | 0 | 2 | 800 |
| *Trifolium purpureum* | 75 | 0 | 15 | 5 | 2 | 800 |
| *Trifolium angustifolium* | 65 | 0 | 20 | 5 | 2 | 800 |
| *Trifolium haussknectii* | 65 | 0 | 20 | 5 | 2 | 800 |
| *Trifolium tomentosum* | 50 | 0 | 20 | 25 | 3 | 400 |
| *Trifolium resupinatum* | 60 | 0 | 20 | 15 | 3 | 300 |
| *Trifolium stellarum* | 25 | 25 | 25 | 20 | 4 | 200 |
| *Scorpiurus muricatus* | 40 | 20 | 25 | 10 | 4 | 900 |
| *Hippocrepis unisiliquosa* | 45 | 0 | 25 | 25 | 5 | 200 |

Note: Numbers do not sum to 100% because about 5-10% of the pods for each species were lost in cracks in the clay loam soil present at Terbol Station in the Bekaa Valley of Lebanon. (Adel Nassar and Ahmed Osman, ICARDA, personal communication)

iii. **Surface water, soil and vegetation management.** The management of these resources is inextricable linked. A particularly important practice is surface water control. In spring, many dry rangeland of Syria, Jordan and Iraq are characterised by large expanses of crusted soil with sparse, diminutive and ephemeral vegetation rapidly withering in the sun. These are interspersed with much smaller areas, in minor wadi beds and small depressions, carrying a rich mixture of annual species, which grow tall, flower and re-seed. Such small green 'oases', often only a few metres across, flourish through the localised runoff of rainwater from the surrounding crusted area. Much of the rainwater, however, is not trapped locally but runs off down the larger wadis to be lost to rivers or through evaporation in larger saline depressions. Productivity of these lands, via the grazing animal, is very low, and their soils and flora are increasingly degraded, while much of the most limiting resource, water, is lost. Improved surface water control, and harvesting of water for productive use, could reinvigorate large areas of this type of land.

There is abundant evidence of erosion of soil by water and wind throughout the region. In large part, control of erosion is a question of surface water and vegetation management, not only in dry areas as indicated in the preceding paragraph, but also in areas with more rainfall. Techniques for water and vegetation management for soil conservation, and *vice versa*, are based on simple principles and are known. The difficult part is their appropriate implementation at the scale of watersheds or landscapes where land holdings are small and ownership is fragmented, land users' resources are limited and land users may not share common goals. The problem is predominantly social and economic rather than technical, and it is clear that its solution will require the consenting participation of land users.

iv. **Common property management.** Groundwater and rangeland are generally community owned or subject to open access. These are natural resources which, in many cases, individuals are free to utilise in an uncontrolled fashion, leading to their depletion and degradation. The problems involved are rarely just technical; usually they relate to land users' economic circumstances, to their perception of the resource and to social institutions, all of these interacting with technical resource limitations. Again, the initial approach to their proper management must involve consultation with, and the agreement of local communities, although final solutions may also call for government intervention. In that case information generated through the activities will aid in policy formulation (Swallow, this volume)..

v. **Land users as partners.** It cannot be reiterated too often that land users, farmers and pastoralists, must be involved as primary participants in agricultural biodiversity management. Innovative approaches to *in-situ* and on-farm conservation will have to be developed alongside appropriate measures for the management of land, water, livestock and vegetation that will rehabilitate degraded lands, while maintaining the productive capacity of the resources and securing the livelihoods of the community. Components of such approaches have been developed. What is now needed is to integrate these components within landscapes and ecosystems. It is clear that, ultimately, this will only be done by those who manage the land day by day, and that their active participation is essential.(Migongo Bake, this volume),.

A further contribution of land users in developing the needed innovations is indigenous knowledge about cultivated species and their wild relatives, native vegetation, and traditional agricultural practices and systems of land, water and livestock management. Most land users do not choose to abuse their resources. Rather, they are forced to behave as they do by economic and social factors over which they have little control, or they are coerced by shortsighted government policies which promise immediate gains, but at the expense of longer-term welfare.

vi. **Interdisciplinary programme units.** All activities will combine in promoting the maintenance of the diversity within farming systems (agro-biodiversity) through genotypic diversity, rotations, diversification of cropping and livestock systems, agro-forestry (woody species, shrubs or trees to provide windbreaks, fodder or fuel, or for commercial use) and the integration of crop and livestock production, all based on soil, water and vegetation conservation measures. To complete these activities will require a diversity of expertise and the concerted efforts of teams of people with comprehensive and complementary skills. Where possible, team members will be drawn from the region. But not all the skills or facilities required are available in the region; nor in the case of expensive and highly specialised equipment, for example, is it necessary, or even desirable, that they should be. Rather, the project will form collaborative links between regional institutions and laboratories in other parts of the world with a comparative advantage in specific topics. The most efficient way to meet the staffing and equipment needs will be by blending regional and non-regional personnel in interdisciplinary programme units, working either in the region or through collaborative links, where key facilities are available. Training is an essential part of this and the collaborative UNEP-funded project "Promotion of Drylands Biodiversity Conservation through Integrated Management" is an excellent example.

### A regional consortium

It is not realistic for any single institution to attempt to address the question of maintenance of biodiversity and natural resources on the scale envisaged in the Near East. For one thing, a wider range of expertise than is found in most, if not all, institutions is needed. For another, the magnitude of the task is beyond the human and financial resources of any one institution. It is proposed that a Consortium be formed to draw together the type and quantity of skills needed to address the complex biological, social and environmental issues involved in the problem. The Consortium was envisaged as an equal partnership among institutions in the Near East and elsewhere (Europe, North America and Australia), each with a comparative advantage in specific aspects related to the work to be undertaken.

The Consortium, which was discussed in detail at two biodiversity workshops held in Amman in June 1994 and February 1995, brings together National Institutions (NRIs, NGOs), International Agricultural Research Centres (IARCs), Research Organisations (Ors) and farmers representatives, in order to develop, through concerted action, applicable strategies for integrated resource conservation and management in the participating countries, thereby curbing the continuing erosion of the productive capacity and biological diversity of the Fertile Crescent and its diverse ecosystems.

Since it commenced operation in 1977, ICARDA has established collaborative partnerships with the Iris, IRAs and ORS and has developed key networks in the region through which the Consortium will operate. ICARDA's headquarters is centrally located within the Fertile Crescent (Figure 1), which geographically makes it ideal for being the Executing Agency in this ecoregional project. IPGRI, the International Plant Genetic Resources Institute, has located its regional office for WANA at ICARDA's headquarters and ACSAD, the Arab Center for the Studies of Arid Zones and Dry Lands, established in 1971 for the management of natural and human resources and documentation concerning Arab arid lands, has its headquarters close by, in Damascus.

The Consortium's activities are not intended to replace but rather to link and build on existing biodiversity research and development projects already operating in the region. Most Consortium activities will be executed through existing regional or national projects or structures within priority ecosystems. Priority areas have been chosen by the National Biodiversity Steering Committees in each country; the choice of agro-ecosystem is determined by a balance between the threat to existing drylands plant biodiversity and national and international concerns. The advantage of the consortium approach is that each partner's comparative advantage can be exploited to form the needed framework for the benefit of the whole region.

The Consortium is seen as providing an opportunity to (a) strengthen national awareness of, and capacity to quantify, as well as tackle, the degradation of agricultural biodiversity and soil and water resources; (b) harness the expertise of individuals and teams through an association of national institutes and international research centres, in fields in which they hold comparative strengths; and, above all, (c) establish a framework to provide coherence and communication among all scientists in the region concerned with agricultural biodiversity and soil, water and vegetation conservation and management.

Besides the planning and targeting of activities, the Consortium will act as co-ordinator in raising additional financial support from the international donor community and national governments, and allocating and dispersing funds for the execution of specific research projects.

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# World Soil Carbon Stocks and Global Change

**N.H. Batjes**

Synopsis
This paper provides new global estimates of current current carbon and nitrogen (N) in soils, with special reference to the drylands. The global soil database, developed for a World Inventory of Soil Emission Potentials (WISE), has been used to compile a global data set of soil carbon and nitrogen pools. WISE currently holds 4,353 globally distributed profiles considered to be representative of the 106 soil units shown on a ½° latitude by ½° longitude version of the corrected and digitized 1:5 M FAO-Unesco Soil Map of the World.

In addition to this, land management procedures for sequestering soil carbon and reducing land based $CO_2$ emissions are reviewed. The possible effects of land use changes and land degradation on C-fluxes from dryland soils to atmosphere and the impact of soil management for carbon sequestration are reviewed. Recommendations for further studies are made.

Key Points

1. World soils are important sources and sinks for $CO_2$, $CH_4$ and $N_2O$, radiatively trace gases which enhance the greenhouse effect. Dominant soils of drylands, mainly from arid and semi-arid tropical and mid-latitude regions of the world, contain about 251 Pg C of SOC and 509 Pg C of carbonate carbon (CAC), amounting to 760 Pg C in total. These totals do not include data for minor associated soils.
2. The total amounts of soil organic carbon (SOC), CAC and N held in the first 1.0 m of dryland soils of the world have been approximated by overlying the WISE data with the digital Holdridge life-zones map. Soils of life-zones corresponding with "dryland" areas of the world, hold 300-369 Pg SOC-C, 473-546 Pg CAC-C, and 41-50 Pg N. Ranges are from 20 Pg SOC-C and 49.6 Pg CAC-C for the Cool Deserts, increasing to 118 Pg SOC-C and 174.2 Pg CAC-C for soils of the Tropical Dry Forest life-zone.
3. Highest SOC pools are found under Boreal forests (346 Pg C), pointing to a possible large release of $CO_2$ from soils of these regions upon predicted climate warming. Globally, the soil carbon pool for the first 1.0 m is estimated to be 2157 Pg C and the nitrogen pool to be 133 Pg N. Large amounts of organic carbon are stored below a depth of 1.0 m, notably in Histosols and some deeply weathered tropical soils. This "deeper" carbon is old, refractory and of limited importance in gaseous exchanges with the atmosphere.
4. Increasing population pressure, land use changes and predicted global warming, through their effects on net primary productivity, organic matter composition and decomposition as well as soil chemical and physical conditions, have important effects on plant composition, SOC pools and atmospheric concentrations of $CO_2$, $CH_4$ and $N_2O$.
5. Soil organic carbon (SOC) reserves in the upper 1.0 m of the world's soils — corrected for amounts of coarse fragments (> 2mm) — is estimated to be 1462 Pg C ($10^{15}$ g), approximately twice the amount held in the atmosphere (» 750 Pg C) and about 2.5 times that held in biota (» 600 Pg C). Although the soil-vegetation carbon pool is relatively small compared with that of the oceans (» 39,000 Pg C), potentially it is much more labile in the short term.

*Keywords*: soil carbon; soil nitrogen; digital soil database; global change; land management; drylands; land degradation; global change

## 1. Introduction

Deciphering the role of terrestrial ecosystems in the global carbon (C) cycle has become increasingly important as policy makers address issues associated with global environmental change, particularly controlling land degradation and climate change (Houghton *et al.* 1990; UNEP 1992). Recent overviews of the carbon cycle may be found in various papers (Houghton *et al.* 1990; Smith *et al.* 1993; Schlesinger 1995; Schimel 1995). Four main reservoirs regulate this cycle on earth: fossil carbon (» 6,000 Pg or $10^{15}$ g C); the oceans (» 39,000 Pg C); the atmosphere (» 750 Pg C); and terrestrial systems including soils and biomass (» 2200 Pg C).

Although the soil-vegetation carbon pool is relatively small compared with that of the oceans, potentially it is much more labile in the short term. The most important natural processes in the exchange of carbon are those of photosynthesis, autotrophic (i.e., $CO_2$ production by the plants) and heterotrophic (i.e., essentially microbial) respiration which convert the organic material back into $CO_2$. The amounts and dynamics of carbon and nitrogen in the world's soils are still relatively poorly known (Houghton *et al.* 1990; Legros *et al.* 1994; Lal *et al.* 1995). Of particular interest is the 'missing sink' of 2.0 - 3.4 Pg C $yr^{-1}$, which arises in global C budgets from the difference in $CO_2$ released by fossil fuel combustion and the annual $CO_2$ increase in the atmosphere (Tans *et al.* 1990; Broeckner & Peng, 1990). Part of the 'missing carbon' has been attributed to a '$CO_2$-fertilization effect' (Sombroek *et al.* 1993; Francey *et al.* 1995), associated with increased atmospheric $CO_2$ levels on plant growth (Bazzaz & Fajer 1992; Idso & Idso 1994).

The favourable effects of organic matter on the physical and chemical properties of soils, on biological activity, and by implication in sustaining soil productivity, are well documented (Russell 1980). Important factors controlling organic matter levels in soils, include climate, hydrology, parent material, soil fertility, soil biological activity, vegetation patterns and land use (Jenny 1980). In the short-term, the carbon balance of terrestrial ecosystems is particularly sensitive to impact of human activities, including deforestation, biomass burning, land use changes and conversion, and environmental pollution (Mann 1986; Mulongoy *et al.* 1993; Fisher *et al.* 1994). During the period 1850-1980 soil carbon pools have decreased by 40 Pg C from the original 1471 Pg C and carbon held in vegetation by 80 Pg C down from 672 Pg C in 1850 (Houghton 1995). Global release of carbon to the atmosphere from land use change in 1990 was1.1 to 3.6 Pg C $yr^{-1}$, as compared to 5.5 - 6.5 Pg C $yr^{-1}$ from fossil fuel combustion (see Houghton 1995). Some 5.7 Pg C of the about 1500 Pg C held in soils is displaced annually due to global soil erosion (Lal 1995 b).

As a whole, the natural resources for food production have shown a marked deterioration during the last two decades; land has been degraded, water resources have been depleted, desertification has increased, biodiversity has decreased, and there have been negative impacts on human health (FAO 1979a; Oldeman *et al..,* 1991; Le Floc'h *et al.* 1992; UNEP 1992).

In view of the scope of UNEP's workshop on "Combatting Global Warming by Combatting Land Degradation", possible effects of land degradation and predicted global warming for soil C and N reserves will be discussed with special reference to dryland regions. Globally, drylands — defined as having a rainfall (R) over potential evapotranspiration (PET) ratio smaller than <= 0.65 (UNEP, 1992) — occupy about one third (49-51 x $10^6$ $km^2$) of the Earth's land surface and are inhabited by about one quarter of the world's population (Le Floc'h *et al.* 1992; UNEP, 1992). The UNCED at Rio de Janeiro in 1992 highlighted desertification as a special problem, with both marked political and technical dimensions.

A comprehensive discussion of the features of drylands and processes of desertification is beyond the scope of this paper and excellent reviews have been published elsewhere (Van Baren 1980; Beaumont 1989; Le Floc'h *et al.* 1992; Mainguet 1994). Drylands have in common their aridic nature and are characterized by low, erratic and infrequent rainfall, intense evaporation and limited water resources. The pervasive aridic nature of drylands limits their carrying capacity and potential for rural development. Increased demands for food, feed and fuel (wood) have increased the pressure on existing arable land and resulted in increased utilization of marginal lands (Wainwright, 1994).

The natural vegetation of drylands is mainly characterized by savannas and some dry-tropical wooded areas. With increasing aridity, the density of trees decreases and thorny, xerophitic and drought-resistant species gain in importance. Most of these species grow slowly and many of these naturally fragile ecosystems regenerate slowly when degraded (Beaumont 1989; Kotschi & Adelhelm 1986; Schutz 1994; Ihori *et al.* 1995). As a result, natural dryland ecosystems are easily damaged in areas where human pressure on the land increases, and this is particularly obvious in arid and semi-arid regions (Van Baren, 1980; West *et al.* 1994).

Livestock rearing and irrigated farming are possible in the arid zones. Rainfed farming is generally practised in the semi-arid zones, yet risky due to uncertain rainfall distribution and risk of drought during the growing period(s). Agriculture requires special methods including dry-farming, growing of fast-growing and drought tolerant crops, or irrigation.

Pressure on the natural resources is highest in the semi-arid zone where livestock rearing and rainfed farming compete for land (Kotschi & Adelhelm 1986), and towards the wetter (subhumid) margins where human population pressure tends to be highest (Biswas 1994; Wainwright 1994). So far, over-exploitation has lead to over 1035 x $10^6$ ha of drylands being degraded, notably by water and wind erosion (Oldeman *et al.* 1991) and by soil compaction and crusting. Sustainable land management practices for maintaining soil quality, improving C sinks and managing long-term C sequestration, which increase the soil's resilience against water and wind erosion, need to be identified (Le Floc'h *et al.* 1992; Kern & Johnson 1993; Peterson *et al.*, 1993).

The aim of this paper to provide global estimates of current carbon and nitrogen in soils, with special reference to dryland regions, using the WISE database (Batjes & Bridges, 1994). In addition to this, land management procedures for sequestering soil carbon and reducing land based $CO_2$ emissions are reviewed and priorities for further work identified.

## 2 Materials and Methods

### *2.1 The WISE soil database*

The primary soil data for this study have been derived from the database developed at ISRIC for the World Inventory of Soil Emission Potentials (WISE) project. The central aim of this activity has been to compile a basic set of uniform soil data for a wide range of global studies, including assessments of crop production potentials, soil vulnerability to pollution, and soil gaseous emission potentials (Batjes *et al.* 1995).

The WISE database consists of two main components (Fig. 1):

(1) Data on the type and relative extent of the component soil units of each terrestrial ½° latitude by ½° longitude grid cell of the world.

(2) Soil profile data for the respective soil units, with associated files listing the analytical methods used and source of primary data.

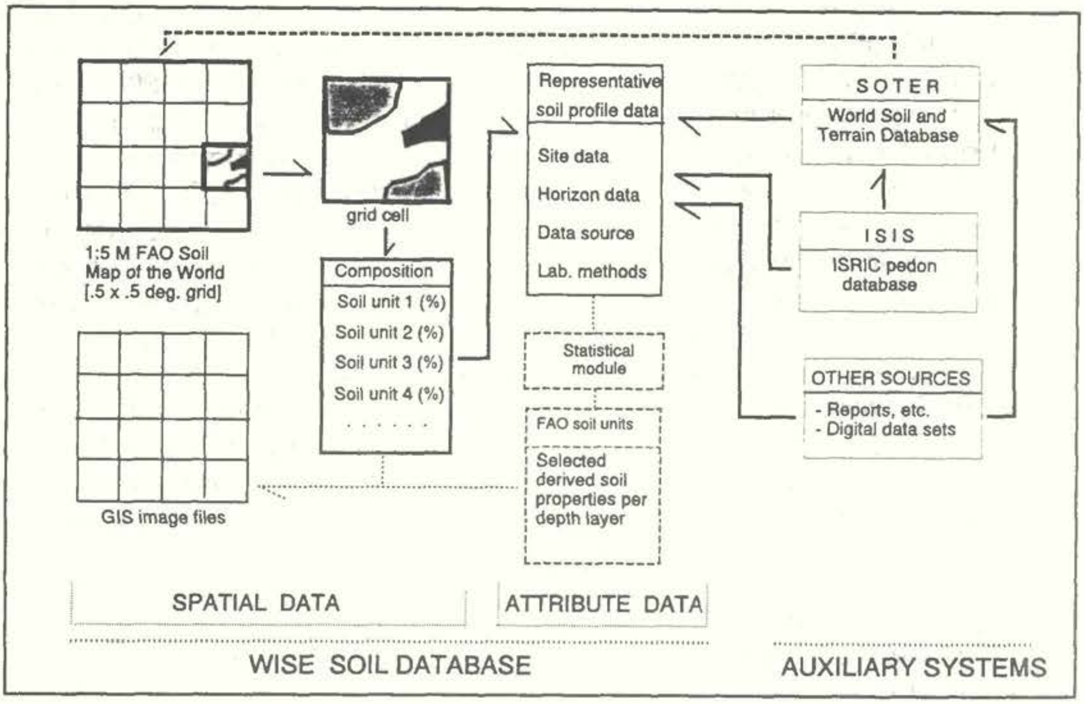


Fig. 1. Schematic representation of the WISE database.

The 1:5 M Soil Map of the World (FAO-Unesco 1971-1981) remains the most consistent map of the world's soil resources, even though sections of it are known to be out-dated (Sombroek, 1990). The cartographic database of WISE has been built up mechanically, by identifying the soil associations which occur in each 5' x 5' grid-cell of an edited and digital version of the Soil Map of the World (FAO, 1991). The next step involved computing the percentage area of each soil unit present in the 36 cells which make up the ½° x ½° grid cell (Nachtergaele, *unpublished data*, 1994), using FAO's standard composition rules. This information was then aggregated to generate the soil geographic data relevant to each terrestrial ½° latitude by ½° longitude grid cell, forming the new cartographic units. Each of these grid cells consists of up to 10 different soil units, of which there are 106 in total (FAO-Unesco, 1974; FAO, 1991). Inclusions of minor soils that may be of critical importance for crop production or greenhouse gas emissions are accounted for, in contrast to what was the case for the Zobler (1986) data file.

The properties of the component soil units of individual grid cells can be quantified using a set of regionally representative and georeferenced soil profiles. These profiles were compiled from 5 main sources: (a) ISRIC's Soil Information System; (b) FAO's Soil Database System; (c) the digital soil data set compiled by the Natural Resources Conservation Service of the United States of America (NRCS); (d) profiles obtained from an international data gathering activity coordinated by WISE project staff, in which national soil survey organisations were asked to supply descriptions and analyses of profiles representative of the units of the Soil Map of the World present in their individual countries; and (e) suitable profiles gathered from survey monographs held at ISRIC's library.

The 4,353 profiles currently held in the WISE database originate from the following regions: Africa (1799); South, West and North Asia (522); China, India and the Philippines (553); Australia and the Pacific Islands (122); Europe (492); North America (266); and South America and the Caribbean (599). No attempt was made in this study to locate where individual profiles occur, because all the profiles were collected to be representative for a particular FAO-Unesco (1974) soil unit. As such, differences in landforms, parent material, land use history and native vegetation are not considered explicitly.

Special attention was given to the systematic collection and recording of data as well as careful consideration of the laboratory methods by which the analytical results were obtained. For each individual set of data, the laboratory name and methods used have been recorded in the database.

### *2.2 Soil carbon and nitrogen files*

In a preceding paper, Batjes (*1996*) studied the variability in soil organic carbon (SOC), carbonate carbon (CAC) and total nitrogen (N) density of the soil units considered on the Soil Map of the World (FAO-Unesco 1974; FAO 1991). First, calculations were made for individual soil profiles taking into account (a) the bulk density, SOC, CAC and N content of individual soil horizons, and (b) the comparability of the analytical methods used. Where measured bulk density data were lacking, a scheme of pedotransfer rules was used to generate surrogate data. In view of the importance of coarse fragments (> 2 mm) in determining actual SOC, CAC and N reserves, a correction factor has been introduced. This was done on a soil unit basis, using the data on coarse fragments held in the WISE database. The resulting 'derived' files provided the basic input for the current study.

Mean SOC, CAC and N density (kg $m^{-3}$) by soil unit, computed for two depth ranges (i.e. 0-30 cm and 0-100 cm), was combined with information on the geographic extent of the corresponding soil units throughout the world, giving estimates of global soil C and N mass. In the case of Lithosols (0-10 cm), Rankers and Rendzinas (0-30 cm) a shallower default depth has been utilized. If there were no representative profiles for a soil unit, a 'best estimate' was assigned to this soil unit with reference to values computed for similar soils.

Software for the various data file selections and computations was written in dBASE IV, the language used for programming the WISE data handling system.

## 3 Soil C and N Reserves

### *3.1 Global estimates*

*Soil organic carbon*

World estimates of soil organic carbon mass, held in the first meter, range from 1000 to 3000 Pg C with three of the most recent published estimates being 1220 (Sombroek *et al.*, 1993), 1431 (Houghton 1995) and 1555 Pg C (Eswaran *et al.*, 1995). Sombroek *et al.* (1993) used the recently corrected Soil Map of the World (FAO 1991) and about 400 soil profiles classified according to the FAO-Unesco (1974) legend. Eswaran *et al.* (1995) used the map of 'Major Soil Regions of the World' (still in press) and about 1000 profiles from 45 tropical countries and a selection of 15,000 profiles from the USA, classified according to Soil Taxonomy (USDA 1994 and earlier versions). Houghton (1995) derived his estimates from soils in major ecosystems.

Table 1 shows estimates of SOC, CAC and soil N pools per 10° latitudinal band. Globally, SOC in the first 1.0 m amounts to 1462 Pg C. This is about 240 Pg C higher than the estimate of Sombroek *et al.* (1993) and about 90 Pg C lower than found by Eswaran *et al.* (1995). In this context, it should be observed that it is unclear whether the above authors used any corrections for coarse fragments. In there is no correction for coarse-fragments, the global estimate would become 1548 Pg C which is very similar to the estimate of Eswaran *et al.* (1995). Nonetheless, estimates presented for individual soil units still vary widely among the various studies.

Calculations of world soil C and N reserves are complicated by a number of factors, notably: (a) the still limited reliability of areas occupied by different kind of soils; (b) the limited availability of reliable, complete and uniform data for these soils; (c) the high spatial variability in carbon and nitrogen content, stoniness and bulk density of similarly classified soils; (d) the comparability of analytical methods used (Pleijsier 1987; Vogel 1994), and (e) the effects of climate, relief, parent material, vegetation and land use.

*Soil carbonate carbon*

The current estimate of 695 Pg C — or 748 Pg C, when no correction for coarse fragments is applied — compares well with the 720 Pg CAC-C published by Sombroek *et al.* (1993). These authors used average carbonate-carbon contents for soil types published by Schlesinger (1982) and soil area estimates derived from the Soil Map of the World (FAO 1991). The initial estimate of Eswaran *et al.* (1995) amounted to 1,738 Pg C of carbonate-carbon, which was about twice the amount estimated by Sombroek *et al.* (1993), the 930 Pg C of Schlesinger (1985) and the present findings. Eswaran *et al.* (1995) estimated carbonate carbon contents arbitrarily where petrocalcic horizons occurred, whereas only actually measured soil carbonate data are used in this study.

Although reserves of carbonate C globally are large, this source of C does not participate in the C flux to other carbon systems as rapidly as organic carbon, except if soils are irrigated or become acidified with increased S and N inputs (Lal *et al.* 1995). Nonetheless, more data are needed on the rates of storage and release of carbonate-carbon in soils under projected climatic and vegetational changes (Schlesinger 1985). Changes in pedogenetic carbonate may have great effects on both phosphorus (P) availability and the biogeochemical cycle of P in dryland systems (West *et al.* 1994). The reason for this is that most inorganic P is present as soluble Ca-phosphates in these soils (Lathja & Bloomer 1988).

*Total soil carbon*

Recently published estimates of soil carbon in the upper 1.0 m of the soils of the world range from 1940 Pg C (Sombroek *et al.* 1993), 2157 Pg C (this study) to 3293 Pg C (Eswaran *et al.* 1995). These figures show that global estimates of carbon are still not very reliable as a result of the lack of uniform and representative data from different parts of the world, notably China and the former USSR, as well as to uncertainties attached to the geographic data used. It is well known that parts of the Soil Map of the World are outdated (Sombroek, 1990), necessitating a concentrated effort in updating the information on the world's soil resources (Oldeman & Van Engelen 1993; Arnold 1995; Madsen & Jones 1995).

Data on amounts of carbon stored at different depths in various soils of the world have been discussed elsewhere (Sombroek *et al.* 1993; Eswaran *et al.* 1995; Batjes, *1996*). Large amounts of SOC occur at greater depth in Histosols and some deeply weathered tropical soils such as the Ferralsols. High concentrations of carbonate-carbon may occur below 1.0 m in petrocalcic horizons. Available data sets, so far, have precluded detailed estimates of these 'deep' reserves worldwide (e.g., Sombroek *et al.* 1993; Eswaran *et al.* 1995).

**Table 1.** Estimated soil carbon and nitrogen reserves per 10° latitudinal band
(Pg of C and N for specified depth zone: A) 0-30 cm, B) 0-100cm; corrected for coarse fragments).

**A)** *d= 0-30 cm*

| | Latitude | | SOC-C | CAC-C | TOT-C | N | area† |
|---|---|---|---|---|---|---|---|
| N | 90 | 80 | 1.0 | 0.0 | 1.0 | 0.0 | 0.1 |
| N | 80 | 70 | 26.3 | 0.9 | 27.2 | 1.7 | 2.3 |
| N | 70 | 60 | 128.4 | 7.5 | 135.9 | 8.3 | 12.2 |
| N | 60 | 50 | 127.4 | 14.8 | 142.2 | 10.3 | 14.1 |
| N | 50 | 40 | 83.6 | 33.7 | 117.2 | 8.6 | 15.6 |
| N | 40 | 30 | 49.2 | 43.3 | 92.5 | 5.9 | 15.1 |
| N | 30 | 20 | 41.0 | 38.7 | 79.6 | 5.0 | 15.0 |
| N | 20 | 10 | 33.3 | 23.7 | 57.0 | 3.9 | 11.2 |
| N | 10 | 0 | 48.7 | 11.0 | 59.7 | 4.5 | 10.0 |
| S | 0 | -10 | 53.4 | 6.8 | 60.2 | 4.8 | 10.3 |
| S | -10 | -20 | 39.1 | 10.5 | 49.6 | 4.0 | 9.4 |
| S | -20 | -30 | 29.7 | 18.9 | 48.6 | 3.5 | 9.3 |
| S | -30 | -40 | 16.4 | 10.0 | 26.4 | 1.8 | 4.1 |
| S | -40 | -50 | 4.7 | 2.0 | 6.7 | 0.5 | 1.0 |
| S | -50 | -60 | 1.9 | 0.2 | 2.1 | 02 | 0.2 |
| S | -60 | -70 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| S | -70 | -80 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| S | -80 | -90 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| **All** | **90** | **-90** | **684.1** | **222.0** | **906.1** | **63.0** | **129.8** |

**B)** *d= 0-100 cm*

| | Latitude | | SOC-C | CAC-C | TOT-C | N | area† |
|---|---|---|---|---|---|---|---|
| N | 90 | 80 | 4.6 | 0.0 | 4.6 | 0.1 | 0.1 |
| N | 80 | 70 | 80.0 | 1.2 | 81.4 | 3.8 | 2.3 |
| N | 70 | 60 | 331.9 | 12.7 | 345.1 | 19.1 | 12.2 |
| N | 60 | 50 | 268.0 | 44.0 | 312.2 | 21.4 | 14.1 |
| N | 50 | 40 | 166.3 | 114.3 | 280.6 | 18.2 | 15.6 |
| N | 40 | 30 | 97.2 | 147.4 | 244.3 | 11.8 | 15.1 |
| N | 30 | 20 | 78.7 | 116.1 | 194.6 | 10.7 | 15.0 |
| N | 20 | 10 | 63.1 | 65.9 | 128.9 | 8.4 | 11.2 |
| N | 10 | 0 | 93.7 | 30.1 | 123.8 | 9.4 | 10.0 |
| S | 0 | -10 | 103.7 | 14.8 | 118.6 | 9.7 | 10.3 |
| S | -10 | -20 | 73.0 | 29.6 | 102.6 | 7.9 | 9.4 |
| S | -20 | -30 | 56.0 | 70.1 | 126.0 | 7.3 | 9.3 |
| S | -30 | -40 | 31.7 | 40.9 | 72.6 | 4.0 | 4.1 |
| S | -40 | -50 | 9.4 | 7.1 | 16.5 | 0.9 | 1.0 |
| S | -50 | -60 | 4.8 | 0.6 | 5.4 | 0.3 | 0.2 |
| S | -60 | -70 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| S | -70 | -80 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| S | -80 | -90 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| All | 90 | -90 | 1462.0 | 695.0 | 2157.0 | 133.0 | 129.8 |

†Area in $10^6$ $km^2$, excluding land glaciers.
SOC-C = soil organic carbon; CAC-C= soil carbonate carbon; TOT-C= (SOC-C + CAC-C);
N= total nitrogen; sums may not exactly add up to totals shown due to rounding

Fisher *et al.* (1994) drew attention to the important role of deep tropical soils and tropical land use in the global carbon cycle. However, in soils where this deeper carbon is old and refractory it will be of limited importance in gaseous exchanges with the atmosphere. Batjes (*1996*) estimated the global SOC pool for the first 2.0 m at 2376-2456 Pg C, of which 616-640 Pg C is stored in tropical soils. No estimates could be proposed yet for carbonate-carbon, because of a paucity of data for carbonate carbon held below 1.0 m depth. The amount of organic carbon held in either resistant charcoal or in litter layers remain difficult to estimate. Litter layers are particularly important in the cooler regions of the world and in soils under forest.

The global amounts of SOC and CAC sequestered in the soils of the world are shown in Fig. 2 and 3. The first map reflects the possible large effects of global warming on C effluxes in high northern latitude areas, notably in peatland, boreal forest and tundra ecosystems (Sampson *et al.* 1993). Soils of arid and semi-arid regions ( Figure 1) have low SOC contents and high CAC reserves, a subject that will be discussed below.

*Soil nitrogen*

Post *et al.* (1985) estimated that the nitrogen pool in the first 1.0 m is about 95 Pg N using an ecosystems approach, which is similar to the 96 Pg N estimated by Eswaran *et al.* (1995) and the 100 Pg N presented by Davidson (1994). Using the WISE database, 133 Pg N is obtained. This higher value is possibly due to the fact that the WISE database contains data for a large number of agricultural soils, where N levels have been increased by chemical fertilizer application. The WISE values are also somewhat higher than the 92-117 Pg of N calculated using an ecosystems approach (Zinke *et al.*, 1984). The total soil N pool is large in comparison to the » 10 Pg of N held in the plant biomass and » 2 Pg N in the microbial biomass (Davidson 1994).

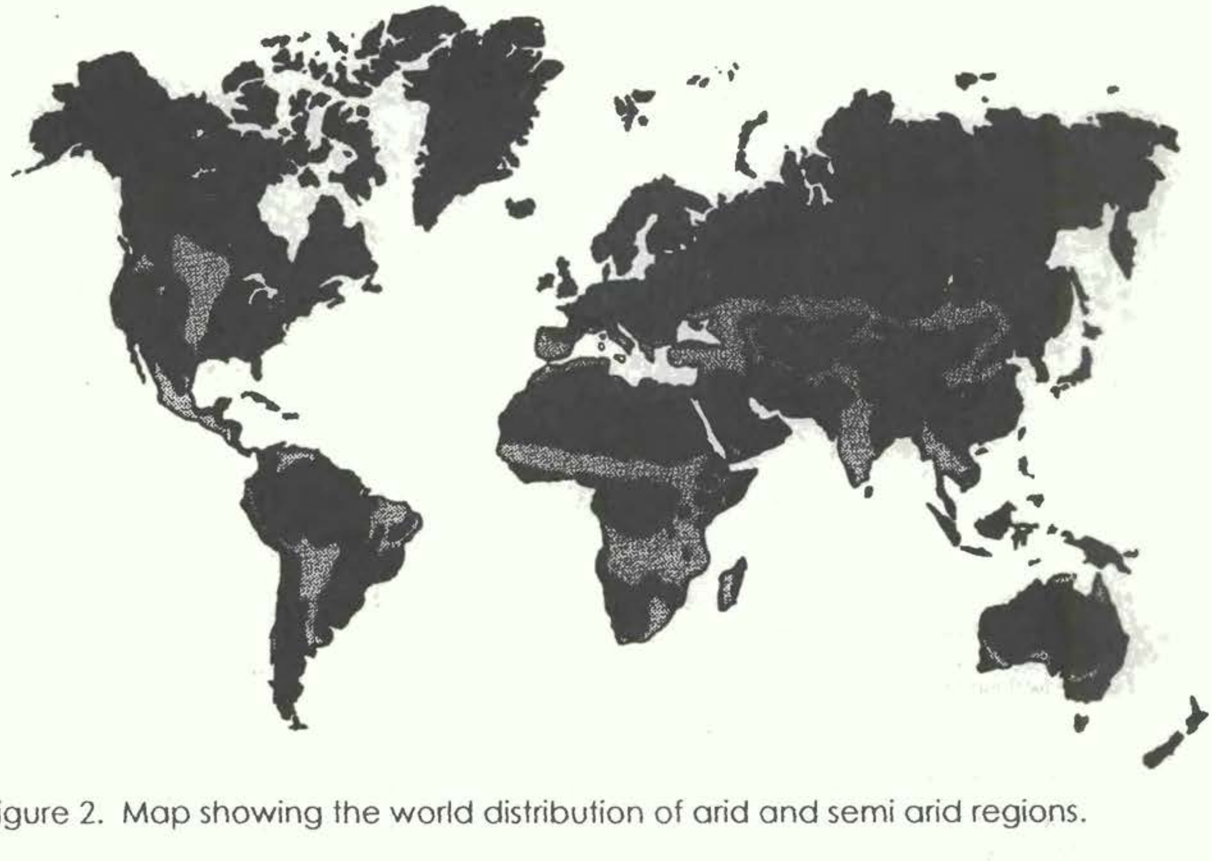

Figure 2. Map showing the world distribution of arid and semi arid regions.

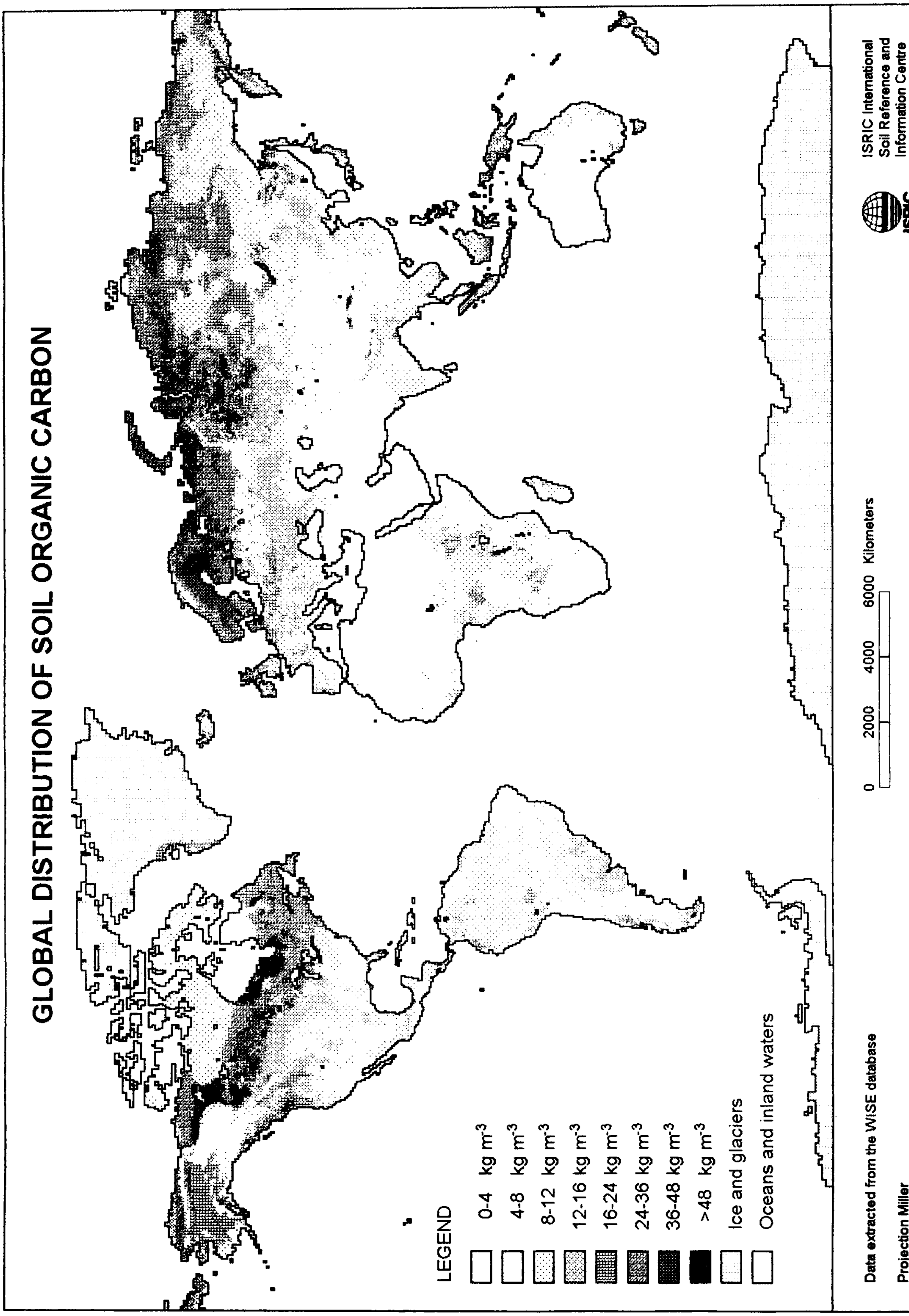


Fig. 3. Soil organic carbon density map derived from WISE database (kg C m$^{-2}$ to a depth of 1.0 m)

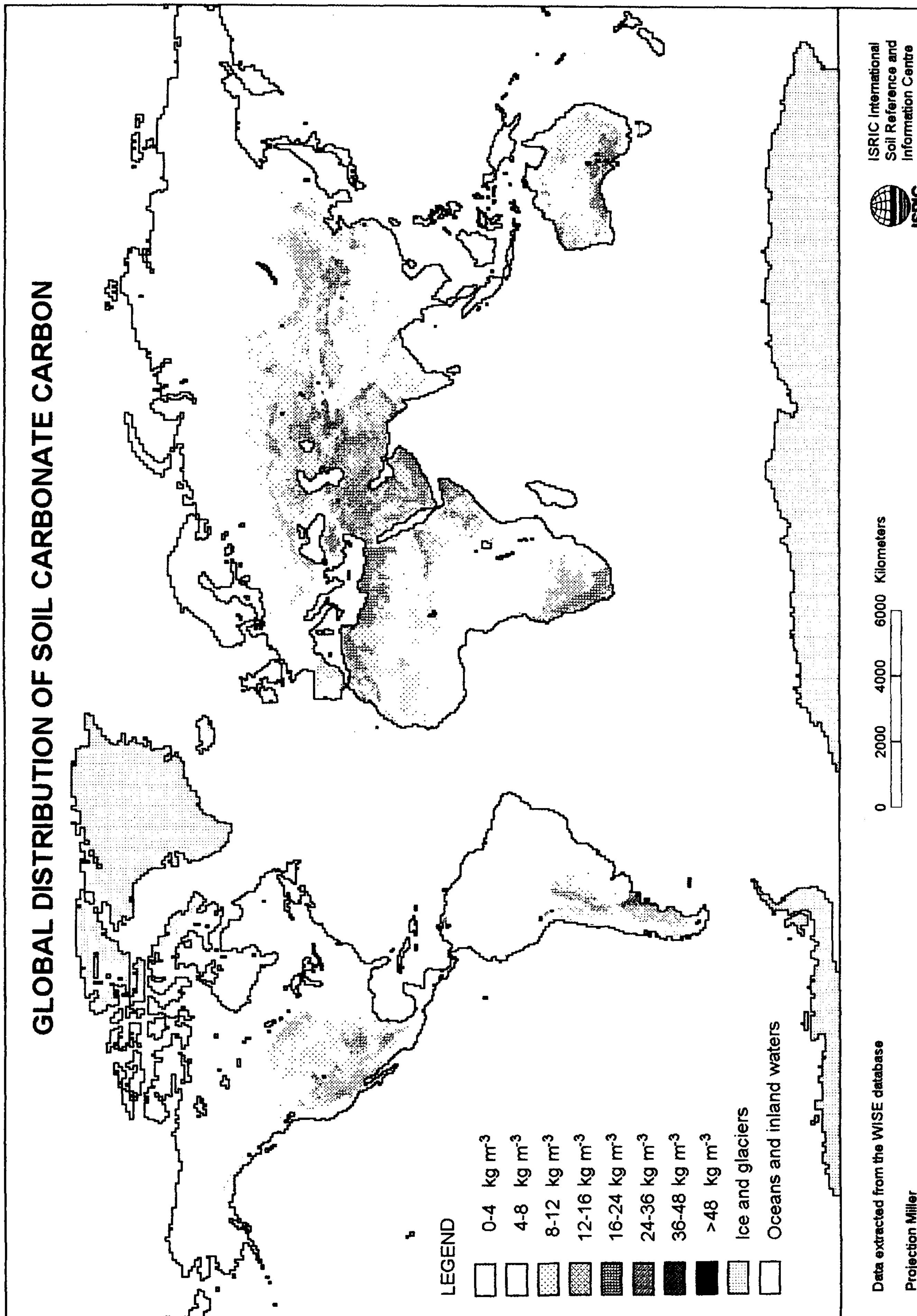


**Figure 4. Soil carbonate carbon density map derived from WISE database. (kg C $m^{-2}$ to a depth of 1.0 m)**

### 3.2 Distribution by Holdridge life-zone

Climate and vegetation are important controlling factors of organic C accumulation in soils (Jenny, 1980). In this section, mean soil carbon and nitrogen contents are estimated using the Holdridge (1947) classification scheme which assigns a life-zone based on climate variables, bio-temperature and annual precipitation. The Holdridge database, simplified to 14 zones by Leemans, was extracted from the Global Ecosystems Database (Kineman 1992).

First the weighted amount of soil SOC, CAC and N was computed for each ½° terrestrial grid, using the grid's full soil unit composition. Areas of oceans, inland waters and glaciers were assigned zero values by default. Next, the weighted grid averages were combined with data on the geographical distribution of Holdridge life-zones worldwide (Table 2). Thereby the current (indirect) approach differs from the one used by Post *et al.* (1982) in which profiles were directly allocated to a Holdridge life-zone, where the life-zones were derived from digital maps. It takes into account that different soil types occur in a given Holdridge life-zone.

Global estimates of SOC and N per Holdridge life-zone presented by Post *et al.* (1982), include a correction for coarse fragments (> 2mm) as for the figures shown in Table 2. Using the figures for Pg C and area estimates, weighted soil organic and carbonate carbon densities can be computed for the various life-zones (Fig. 4). Mean values (to a depth of 1.0 m) range from 3.2 kg SOC-C $m^{-2}$ for soils of the Hot Desert life-zone to 23.2 kg SOC-C $m^{-2}$ for soils of the Boreal Forest life-zone. Comparison of these figures with those of other researchers (e.g., Post et al. 1982; Houghton 1995; Ojima *et al.* 1993) is complicated by differences in definitions and procedures for aggregating life-zones or ecosystems. The relatively large contribution of CAC to soil carbon reserves of dryland systems is apparent from Table 2B, ranging from 77% for Hot Deserts to 7% for Forest Tundra. The relative contribution of soils from the various life-zones to the global soil carbon pool is shown in Fig. 5.

**Table 2.** Estimated organic carbon and nitrogen pools, aggregated per Holdridge life-zone (Pg C respectively N for specified depth zone: A) d= 0-30 cm, B) d= 0-100 cm; corrected for coarse fragments)

**A)** *d= 0-30 cm*

| Holdridge life-zone | SOC-C | CAC-C | TOT-C | N | Areat |
|---|---|---|---|---|---|
| Tundra | 54.0 | 9.5 | 63.5 | 3.9 | 10.16 |
| Cold Parklands | 16.1 | 5.1 | 21.2 | 1.5 | 2.78 |
| Forest Tundra | 74.2 | 8.5 | 82.7 | 5.3 | 8.72 |
| Boreal Forest | 152.1 | 12.6 | 164.7 | 11.1 | 14.94 |
| Cool Desert | 9.6 | 14.4 | 24.0 | 1.3 | 4.00 |
| Steppe | 35.3 | 19.0 | 54.3 | 4.3 | 7.35 |
| Temperate Forest | 63.9 | 11.3 | 75.2 | 6.2 | 9.90 |
| Hot Desert | 36.6 | 71.3 | 107.9 | 4.9 | 20.80 |
| Chaparral | 21.8 | 15.0 | 36.8 | 2.5 | 5.62 |
| Warm Temperate Forest | 15.9 | 2.8 | 18.7 | 1.6 | 3.20 |
| Trop. Semi-Arid | 27.0 | 24.3 | 51.3 | 3.3 | 9.53 |
| Trop. Dry Forest | 61.1 | 18.7 | 79.8 | 6.6 | 14.85 |
| Trop. Seasonal Forest | 70.8 | 6.8 | 77.6 | 6.7 | 15.08 |
| Trop. Rain Forest | 45.7 | 2.7 | 48.4 | 4.0 | 8.46 |
| All ecosystems | 684.1 | 222.0 | 906.1 | 63.0 | 135.39 |

**Table 2 (continued)**

**B)** *d= 0-100 cm*

| Holdridge life-zone | SOC-C | CAC-C | TOT-C | N | A:ea† |
|---|---|---|---|---|---|
| Tundra | 158.8 | 16.4 | 175.2 | 8.3 | 10.16 |
| Cold Parklands | 35.7 | 15.3 | 51.0 | 3.4 | 2.78 |
| Forest Tundra | 180.2 | 14.4 | 194.6 | 11.7 | 8.72 |
| Boreal Forest | 345.9 | 34.9 | 380.8 | 23.4 | 14.94 |
| Cool Desert | 19.5 | 49.6 | 69.1 | 3.0 | 4.00 |
| Steppe | 69.1 | 73.1 | 142.2 | 9.2 | 7.35 |
| Temperate Forest | 120.3 | 31.9 | 152.2 | 11.7 | 9.90 |
| Hot Desert | 66.1 | 231.6 | 297.7 | 11.4 | 20.80 |
| Chaparral | 42.5 | 53.7 | 96.2 | 5.4 | 5.62 |
| Warm Temperate Forest | 30.4 | 6.3 | 36.7 | 3.1 | 3.20 |
| Trop. Semi-Arid | 53.1 | 82.2 | 135.3 | 7.1 | 9.53 |
| Trop. Dry Forest | 118.3 | 55.9 | 174.2 | 13.7 | 14.85 |
| Trop. Seasonal Forest | 133.0 | 21.1 | 154.1 | 13.5 | 15.08 |
| Trop. Rain Forest | 89.0 | 8.6 | 97.6 | 8.0 | 8.46 |
| **All ecosystems** | 1462.0 | 695.0 | 2157.0 | 133.0 | 135.39 |

†Area in $10^6$ $km^2$, excluding land glaciers.
Holdridge life-zones as simplified to 14 classes by Leemans (see Kineman 1992).
SOC-C = soil organic carbon; CAC-C= soil carbonate carbon; TOT-C= (SOC-C + CAC-C);
N= total nitrogen; sums may not exactly add up to totals shown for all life-zones due to rounding.

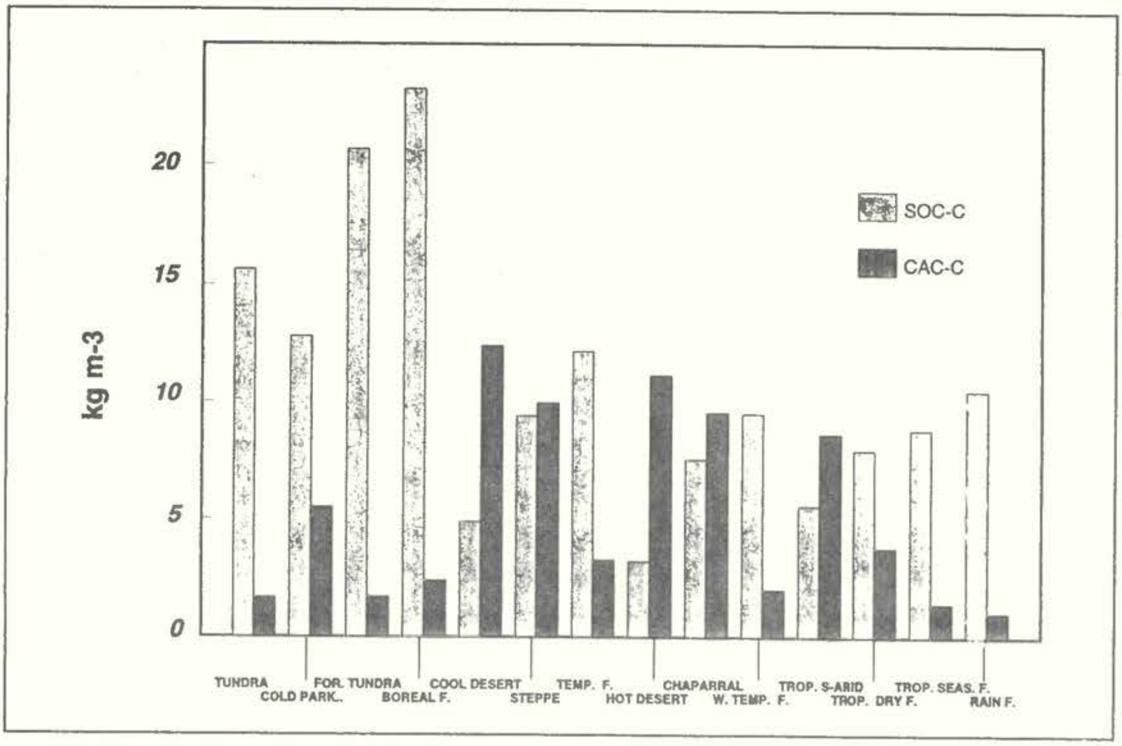


**Fig. 4.** Weighted soil carbon density by simplified Holdridge life-zones (kg $m^{-2}$ to 1.0 m depth; SOC-C= soil organic carbon; CAC-C= soil carbonate carbon; co

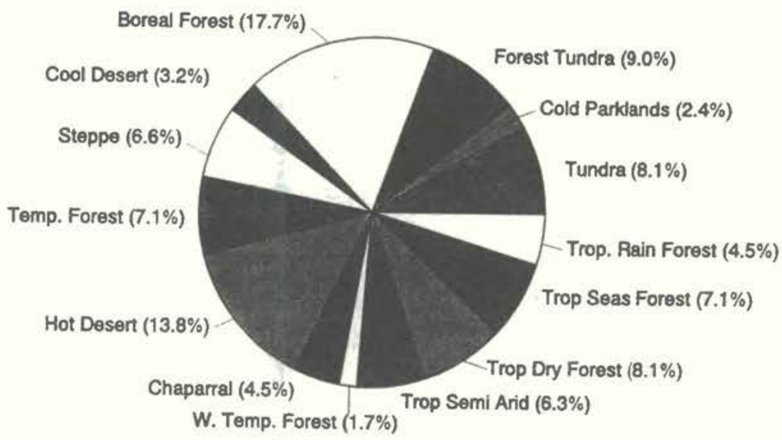


Fig. 6. Relative contribution of soil units, aggregated by Holdridge life-zones, to the global carbon pool (Total is 2157 Pg C to 1.0 m depth, including soil organic carbon and soil carbonate carbon; corrected for coarse fragments).

### *3.3 Analyses for dryland soils*

*Global estimates*

Although a distinct characteristic of dryland regions is their lack of available moisture for plants, it remains difficult to define their extent unambiguously (FAO 1978-1981; UNEP 1992). Particularly because climatic aridity, the feature which drylands have in common, varies strongly according to rainfall distribution and temperature. A useful criterion for defining limits of dryland regions at the regional scale is the aridity index, calculated as the ratio of rainfall (R) over potential evapotranspiration (PET) (UNEP 1992). A shortcoming in using R/PET is that it ignores site specific moisture conditions, as modified by surface run-off, soil water retention and release properties, and capillary rise. Soil moisture regimes, however, are not considered explicitly in the FAO-Unesco (1974) legend, except for Xerosols and Yermosols.

A practical way of defining the extent of dryland soils would be to use the Agro-Ecological Zones map of FAO (1978-1981). Unfortunately, full digital coverage is not expected before the end of 1995 (Van Velthuizen, *pers. comm.*). Similarly, a digital copy of the Aridity Zones map, presented in the Atlas of Desertification (UNEP 1992), was not available from UNEP-GRID for this study. As a surrogate, the Cool Desert, Hot Desert, Chaparral, Tropical Semi Arid and Tropical Dry Forest life-zones of Holdridge will be used to estimate the lower range of global carbon and nitrogen pools for the dryland regions, while Steppes are also included to get the upper range. Table 2 shows soils of the above life-zones hold 300-369 Pg SOC-C, 473-546 Pg CAC-C and 41-50 Pg N in the first 1.0 m, corresponding respectively with about 20-25, 68-79 and 31-38 % of the world's estimated total reserve. Solomon *et al.* (1993) estimated that soils of 'grasslands/deserts' hold 256 Pg C, while Ojima *et al.* (1993) estimated this to be 417 Pg C to a depth of 1.0 m for 'grassland and drylands'.

*Estimates for main dryland soils*

Soils of dryland regions are extremely variable. Low rainfall reduces plant growth and build up of organic matter. Typically, long-term low precipitation enables an accumulation of soluble salts, carbonates and silicates with ascending (evaporating) water in the profile which may lead to surficial accumulation (Buringh 1970; Driessen & Dudal 1993).

Dominant soil units conditioned by limited leaching include Yermosols and Xerosols (which have very weak to weakly developed ochric horizons) and Solonchaks and Solonetz (which are characterized by high salt concentrations) (FAO-Unesco 1974). Important associated soils include: (a) Regosols: immature soils developed on unconsolidated materials, (b) Vertisols: deeply cracking and swelling clay soils; (c) Arenosols: deep sandy soils; (d) Cambisols: a range of slightly to moderately weathered soils (mainly calcic and eutric); and (e) a range of shallow soils limited in depth by hard rock (Lithosols, Rendzinas and Rankers). Inclusions of Calcaric Luvisols, Kastanozems and Chernozems may occur towards the more humid climate types where natural vegetation grades to steppes, but these soils have not been considered in Table 3.

Values in Table 3 again are difficult to compare with the findings of other studies. A major constraint is the different types of classification systems used, e.g. FAO-Unesco legend versus Soil Taxonomy, and the way in which various soil types have been regrouped in various studies.

Sombroek *et al.* (1993) estimate carbonate carbon reserves to be 537 Pg CAC-C in Xerosols, Yermosols and petrocalcic phases, while the present study gives 320 Pg CAC-C for Yermosols and Xerosols alone. This difference should reflect the large amounts of carbonate carbon held in other soils with petrocalcic horizons. According to Eswaran *et al.* (1995), 1,044 Pg CAC-C is held in the world's Aridisols. A better correspondence is found for Vertisols, which are classified using similar criteria in all classification systems, viz.: 19 Pg (Sombroek *et al.* 1993), 25 Pg C (Eswaran *et al.* 1995) and 32 Pg C (this study).

Table 3. Estimates of soil C and N pools of main dryland soils of the world. (Pg C respectively N for specified depth zone: A) d= 0-30 cm, B) d= 0-100 cm; corrected for coarse fragments)

*A) d= 0-30 cm*

| Soil name | SOC-C | CAC-C | TOT-C | N |
|---|---|---|---|---|
| Arenosols | | | | |
| Albic Arenosol | 1.4 | 0.0 | 1.4 | 0.1 |
| Cambic Arenosol | 2.4 | 4.4 | 6.8 | 0.3 |
| Ferralic Arenosol | 3.6 | 0.0 | 3.6 | 0.7 |
| Luvic Arenosol | 0.8 | 1.8 | 2.5 | 0.1 |
| Cambisols | | | | |
| Chromic Cambisol | 3.5 | 1.2 | 4.7 | 0.4 |
| Eutric Cambisol | 10.8 | 3.9 | 14.8 | 1.6 |
| Calcic Cambisol | 2.2 | 3.7 | 5.9 | 0.3 |
| Vertic Cambisol | 1.3 | 0.9 | 2.2 | 0.2 |
| Fluvisols | | | | |
| Calcaric Fluvisol | 2.6 | 5.8 | 8.3 | 0.3 |
| Eutric Fluvisol | 8.2 | 2.8 | 11.1 | 1.3 |
| Lithosols | 38.8 | 41.8 | 80.6 | 4.5 |
| Regosols | | | | |
| Calcaric Regosol | 2.7 | 6.5 | 9.2 | 0.6 |
| Eutric Regosol | 8.4 | 2.1 | 10.5 | 1.2 |
| Rendzinas | 3.9 | 2.0 | 5.9 | 0.3 |
| Solonchaks | | | | |
| Gleyic Solonchak | 0.8 | 2.2 | 3.0 | 0.1 |
| Mollic Solonchak | 0.5 | 0.6 | 1.1 | 0.1 |
| Orthic Solonchak | 2.9 | 8.0 | 10.9 | 0.4 |
| Takyric Solonchak | 0.4 | 1.5 | 2.0 | 0.1 |
| Solonetz | | | | |
| Gleyic Solonetz | 0.5 | 0.0 | 0.5 | 0.1 |
| Mollic Solonetz | 2.7 | 0.0 | 2.8 | 0.4 |
| Orthic Solonetz | 3.6 | 4.5 | 8.1 | 0.5 |
| Vertisols | | | | |
| Chromic Vertisol | 6.0 | 4.7 | 10.7 | 0.7 |
| Pellic Vertisol | 7.0 | 2.9 | 9.9 | 0.8 |
| Xerosols | | | | |
| Haplic Xerosol | 3.5 | 10.3 | 13.9 | 0.4 |
| Calcic Xerosol | 5.6 | 13.5 | 19.1 | 0.7 |
| Luvic Xerosol | 1.3 | 2.5 | 3.7 | 0.2 |
| Gypsic Xerosol | 0.2 | 0.4 | 0.6 | 0.0 |
| Yermosols | | | | |
| Haplic Yermosol | 4.8 | 7.5 | 12.3 | 1.0 |
| Calcic Yermosol | 2.6 | 13.1 | 15.7 | 0.3 |
| Luvic Yermosol | 2.6 | 4.4 | 7.0 | 0.3 |
| Takyric Yermosol | 0.1 | 1.6 | 1.7 | 0.0 |
| Gypsic Yermosol | 0.5 | 4.1 | 4.6 | 0.1 |
| All units | 136.4 | 158.6 | 295.0 | 17.9 |

† Sums may not exactly add up to totals shown due to rounding.

*B) d= 0-100 cm*

| Soil name | SOC-C | CAC-C | TOT-C | N |
|---|---|---|---|---|
| Arenosols | | | | |
| Albic Arenosol | 1.9 | 4.1 | 6.0 | 0.2 |
| Cambic Arenosol | 5.8 | 14.9 | 20.7 | 0.8 |
| Ferralic Arenosol | 8.4 | 0.0 | 8.4 | 1.4 |
| Luvic Arenosol | 2.0 | 5.9 | 7.9 | 0.4 |
| Cambisols | | | | |
| Chromic Cambisol | 6.6 | 3.0 | 9.6 | 0.7 |
| Eutric Cambisol | 21.6 | 12.5 | 34.1 | 2.9 |
| Calcic Cambisol | 5.2 | 12.6 | 17.8 | 0.7 |
| Vertic Cambisol | 2.7 | 2.6 | 5.3 | 0.4 |
| Fluvisols | | | | |
| Calcaric Fluvisol | 6.8 | 19.3 | 26.1 | 0.9 |
| Eutric Fluvisol | 21.4 | 8.8 | 30.2 | 3.2 |
| Lithosols | 38.8 | 41.8 | 80.6 | 4.5 |
| Regosols | | | | |
| Calcaric Regosol | 7.7 | 16.0 | 23.7 | 1.5 |
| Eutric Regosol | 13.8 | 2.5 | 16.3 | 2.3 |
| Rendzinas | 3.9 | 3.5 | 7.4 | 0.3 |
| Solonetz | | | | |
| Gleyic Solonetz | 1.0 | 0.2 | 1.2 | 0.1 |
| Mollic Solonetz | 5.3 | 0.6 | 5.8 | 0.4 |
| Orthic Solonetz | 7.4 | 17.0 | 24.4 | 2.6 |
| Solonchaks | | | | |
| Gleyic Solonchak | 1.9 | 8.1 | 10.0 | 0.4 |
| Mollic Solonchak | 1.0 | 3.0 | 4.0 | 0.1 |
| Orthic Solonchak | 6.9 | 27.5 | 34.3 | 1.0 |
| Takyric Solonchak | 1.2 | 6.4 | 7.6 | 0.1 |
| Vertisols | | | | |
| Chromic Vertisol | 15.1 | 16.4 | 31.5 | 1.6 |
| Pellic Vertisol | 16.8 | 10.1 | 26.9 | 1.7 |
| Xerosols | | | | |
| Haplic Xerosol | 7.5 | 34.8 | 42.2 | 0.9 |
| Calcic Xerosol | 10.5 | 76.9 | 87.4 | 1.4 |
| Luvic Xerosol | 3.3 | 11.5 | 14.8 | 0.3 |
| Gypsic Xerosol | 0.5 | 1.3 | 1.8 | 0.0 |
| Yermosols | | | | |
| Haplic Yermosol | 12.5 | 30.9 | 43.4 | 3.6 |
| Calcic Yermosol | 7.7 | 86.4 | 94.1 | 0.5 |
| Luvic Yermosol | 4.7 | 19.3 | 23.9 | 0.5 |
| Takyric Yermosol | 0.4 | 5.7 | 6.1 | 0.1 |
| Gypsic Yermosol | 1.2 | 5.4 | 6.6 | 0.1 |
| All units | 251.1 | 508.9 | 760.0 | 35.4 |

## 4 Combatting Land Degradation and Global Warming

### 4.1 *Vulnerability to land degradation*

As shown by Jenny (1980) all soils are formed and evolve under the influence of time, climate, parent material, topography, vegetation, fauna and human influences. Although many effects of human influence have been positive — causing an accumulation of C, N and P and improving porosity in ancient agricultural soils (Sombroek 1966; Sandor & Eash 1995) — and increased food production, other activities have lead to extensive soil physical and chemical degradation (Oldeman *et al.* 1991; O'Hara *et al.* 1993). The latter include: industrialization with concomitant release of toxic substances into the atmosphere (e.g., SOx, NOx), producing acid [illegible] management practices such as 'short cycle' slash-and-burn, excessive fertilizer application of N and P, and poor irrigation and drainage practices leading to salinization.

As the 21st century approaches, it has become increasingly clear that the drylands of the world will be subjected to even greater land use pressures as a result of the continued growth of the population (Beaumont 1989; Biswas 1994). Crucial factors are the wealth of a nation, the political system, and the relative importance of drylands in the economy (Beaumont 1989; Westing 1994). The key issue is that the growing population must be fed and housed, requiring conservation of the natural resources.

Soils of drylands are especially vulnerable to human-induced degradation in view of the slow speed of their recovery after a disturbance. This is mainly due to the limited water supply and associated low rates of *in situ* soil formation and plant growth. Vegetation and soils differ in their resilience to disturbance. Although vegetation communities are easily disturbed or degraded, recovery rates remain fast when compared with soil formation (Arnold *et al.* 1990). Compared to soils of humid areas, dryland soils have a low resilience to human disturbance (UNEP 1992).

### *4.2 Main causes of soil degradation*

In many countries, external social, economic and technological influences have contributed significantly to social and ecological imbalances in land use systems, leading to land degradation (Biswas 1994). Five main causes of human-induced land degradation are deforestation and removal of natural vegetation, over-cultivation, over-grazing, mismanagement of water resources, and industrial development (Grainger 1985; Kotschi & Adeljhelm 1986; Thomas & Middleton 1994; UNEP 1992). The adverse effects of intensive tourism in uplands and national parks are also well known.

Local-scale variability in organic matter oxidation, N-mineralization and soil loss through cultivation are strongly dependent on management practices (Veldkamp 1993; Herrmann *et al.* 1994; Ihori *et al.* 1995) and seasonal rainfall distribution. Climate-induced changes, causing major differences in ground cover, could degrade structure and decrease porosity in most soils, leading to increased runoff and erosion in sloping land. Associated, adverse off-site effects are sedimentation and flooding in low lying areas, and increased wind erosion in dry areas.

Since the beginning of the 20th century, there has been an overall tendency for increased aridification in the Sudano-Sahelian zone (Grainger 1985; Carbonnel & Hubert 1992). These changes should not be seen in isolation from human-induced processes of global change. Atmospheric circulation patterns over deforested tropical regions, for example, have been shown to prompt climate changes distant from the disturbance, both in tropical, middle and high latitudes (McGuffie *et al.* 1995). Soil erosion from arid and semiarid systems influences nutrient status in the rest of the earth's ecosystems (Simonson 1995) and may alter global climatic patterns (Schlesinger *et al.* 1990).

### *4.3 Main processes of land degradation*

The main processes of soil degradation associated with desertification may be summarized as follows:

(a) increased land pressure leads to local loss of vegetation, cover and increased area of bare patches (increase in patchiness (Schlesinger *et al.* 1990)). Removal of crop (residues) for fuel or fodder reinforces this trend.

(b) direct exposure of topsoil to solar radiation increases soil temperature and the rate of organic matter decomposition (Jenkinson & Ayanaba 1977), where soil moisture is not limiting.
(c) loss of organic matter causes soil structural degradation (e.g., porosity, aggregate stability), reduces water holding capacity, causes compaction with decreased infiltration and increased runoff, thereby reducing the system's resilience against erosion.
(d) decline in organic matter content decreases nutrient storage properties. Nutrients are lost by percolation and the efficiency of chemical fertilizers is reduced.
(e) impact of rain and sun on bare topsoil results in crusting, water infiltration is further reduced, and percentage of runoff increases.
(f) sediments, with enclosed nutrients, are carried away by erosion. Effective soil depth accessible to plant roots decreases, leaving exposed restrictive soil layers or bare rock.
(g) exposed soil is eroded by wind, crops are destroyed by dust bearing winds (off-site effects), and dunes may encroach on arable land. An important off-site effect is the silting-up of dams and river beds, and increased flooding risk in low lying areas. Additionally, airborne dust from dryland regions can form an important source of nutrients (Simonson 1995) and alkalinity (Rodá *et al.* 1993) in areas where it is deposited on soils.
(h) in the worst-case scenario, gradually degraded patches link up to form extended areas of bare and degraded land. At this stage, reclamation becomes virtually impossible.

### *4.4 Combatting land degradation*

Controlling desertification involves all aspects of environmental management, including water and soil management. In most cases, changes in soil properties by direct human influence, whether intentional or not, are far greater than effects induced by direct climate change (see Scharpenseel *et al.* 1990). Soil management measures designed to optimize the soil's sustained productivity should therefore be adequate to counteract degradation of agricultural land by climate change (FAO 1993). When new land is to be developed, or land use is to be changed, it is important to survey the resources and to make an assessment of their suitability for specific uses (FAO 1979b, 1985; Kassam *et al.* 1993). Thereby, expansion of cropping onto marginal land can be reduced. Better range management and development of new livestock breeds will improve productivity and reduce pressures on the land, particularly if critical dry-season fallows are re-introduced (Grainger 1985). Restoring tree and woodland cover will stabilise cropping and pastoralism by reducing soil erosion and providing improved supplementary fodder (Grainger 1985).

Erosion control measures vary considerably according to ecological and socio-economic conditions. Traditional strategies which are generally well suited to the physical environment, are partly becoming obsolete because of population growth. The use of modern farm equipment in contemporary agricultural systems has often failed because of lack of compatibility with populations concerned. Roose (1992) proposes a rural development strategy (water and soil fertility conservation) based on the requirements, traditions and economic capability of farmers.

Sustainable irrigated agriculture is possible in arid zones, even when water quality is mediocre, provided site conditions are properly evaluated, adequate drainage facilities installed, and monitoring systems are put in place (Szabolcs 1991). However, these measures are costly.

Linkages between drought, food production, famine and migration make monitoring of dryland degradation an important issue. Improved monitoring systems are being developed particularly in the field of remote sensing (Millington *et al.* 1994). Harrison & Carg (1991) demonstrate the problems of classifying the vegetation of complex semi-arid landscapes using satellite data, showing that ground observations of changes remain critical for 'ground-truthing'. Important indicators of soil quality that should be monitored include soil structure, soil permeability, soil organic matter content, base saturation, pH and changes in available nutrients or salts in relation to water management.

Studies of land degradation processes have resulted in various qualitative and quantitative models (see De Jong 1994). These models are useful to test scenarios and to explore alternatives to current land uses, provided they have been validated for regional conditions. Uniform databases of the main environmental factors are critical in order to develop, to test and to run these models (Van den Berg 1992). Although models are effective tools for understanding complex interactions among various system components and for visualizing different scenario's to decision makers, many remain poorly suited to the highly site-specific needs of farmers (see Powell *et al.* 1994).

Land users and planners should protect soil resources from global change (e.g., climate, atmospheric pollution, drought, erosion) by:

(a) managing soils with a view to ensuring maximum physical resilience is retained by preserving a heterogenous system of macropores, by conserving soil (micro)biological diversity (Beese *et al.* 1994; Steinberger 1995), and by maintaining a closed ground cover (FAO 1993; Lal & Kimble 1994; Sauerbeck 1994). This can be done with soil management practices already available, including conservation or minimum tillage, water management and erosion control (Kovda 1980; Kern & Johnson 1993; Lal 1995a) and the planting of hedgerows (Kiepe 1995). Tillage methods that minimize depth and extent of soil disturbances will have the least impact on soil gaseous emissions (Reicoksy & Lindstron 1995).

(b) using integrated plant nutrient management systems to balance the input and removal of nutrients over time (FAO 1993; Smaling 1993; Sauerbeck 1994). Fertiliser costs for farming systems with an inherently low biological production potential, however, are likely to exceed the benefits in terms of increased crop and livestock output (Powell *et al.* 1994). Reducing nutrient losses via runoff, leaching, volatilization etc. can enhance the profitability of using external nutrient sources.

(c) restoration of degraded and eroded lands by proven methods, such as afforestation, agro-forestry, improved pastures with low stocking rates, use of chemical fertilizers, ecologically compatible farming systems, conservation tillage, mulch farming techniques, fertility maintenance, and integrated pest management (Kovda 1980; Lal & Kimble 1994).

(d) developing and promoting viable, science-based alternatives to subsistence and resource-based agriculture (Lal & Kimble 1994). More research is required in areas related to vegetation dynamics and feed availability from natural pastures (Grainger 1985; Powell *et al.* 1994). This research should take into consideration the needs and wishes of the rural population and their cultural and societal values. These solutions should, for example, take into account the spatial variability of rainfall and the risk-minimizing behaviour of rural families.

(e) taking into consideration socio-economic issues which limit the land users ability to remedy land degradation (e.g., lack of secure tenure/ownership of the land, limited access to inputs, and possible lack of knowledge and technology to effectively use these inputs).

*4.5 Possible effects of climate change*

Understanding how soil organic matter and nutrient status change in dryland soils in response to climate change requires knowledge of severbiogeochemical input and output processes. The effect of global climate change on soil organic matter content, soil organic matter quality and nutrient pools depends on the relative sensitivity of photosynthesis, autotrophic and heterotrophic respiration to climatic changes. There is increasing evidence that water or temperature stressed plants are more responsive to $CO_2$ increase — because higher $CO_2$ reduces transpiration — than unstressed plants. According to a review of mainly short duration experiments (Idso & Idso 1994), it appears that the relative growth-enhancing effects of atmospheric $CO_2$ enrichment is greatest when resource limitations and environmental stresses are most severe. Idso & Idso (1994) therefore argue that it could well be that the percentage growth response of natural ecosystems to atmospheric $CO_2$ enrichment will be greater than that of managed agricultural systems. In how far these experimental trends apply also to longer-term periods in ecosystems remains to be assessed (Smith *et al.* 1993). In general, it seems that global systems without effective management intervention will become much stronger C sources under doubled $CO_2$ climate (Sampson *et al.* 1993).

Ecosystem models discussed by Schimel (1995) suggest that plant growth, under increased atmospheric $CO_2$ concentrations, may eventually become nutrient-limited; sequestration of C resulting from $CO_2$-enhanced growth will store nitrogen and other nutrients in wood or unreactive soil organic matter. In the long-term, this can cause these elements to become limiting for plant growth. In principle, this problem can be addressed by larger applications of chemical fertilizers and promotion of integrated plant-nutrition measures in agro-ecosystems (Sombroek *et al.* 1993). Effectiveness of the $CO_2$ fertilization effect could also be modified by atmospheric N deposition and increased levels of UV-B radiation. Indirect changes caused by increased UV-B — such as changes in plant form, biomass allocation to different parts of the plant, timing of developmental phases and secondary metabolism — may be equally or more important than direct damaging effects of UV-B (Caldwell *et al.* 1995).

Changes in plant communities associated with changes in climate will affect litter quality, which will lead to either positive or negative feedbacks depending on whether lignaceous perennials or non lignaceous annuals take over. With respect to drylands, West *et al.* (1994) expect a decrease in shrub abundance, an increase in half-shrubs and herbaceous species, an increase in $C_4$ plants, an increase in C/N ratio in plant tissues, and a decrease in succulents under all scenarios of climate change. Climatic shifts may be accompanied by increases in temperature, seasonal rainfall distribution, atmospheric N deposition, burning and so on, making it difficult to make predictions as other factors are included. Different microbial responses to these changes are reasonably well known, but complex interactions such as effects on N-fixation, N-mineralisation, denitrification, cation leaching and ratios of trace gas emissions remain difficult to predict (Davidson 1994). Climatic change may further affect the population dynamics and status of insect pests of crops, affecting possibilities for C storage (Cammel & Knight

1992). Since the $Q_{10}$ for litter decomposition is greater (1.3-4.0) than the $Q_{10}$ for primary production (1.0-1.5), litter decomposition is projected to proceed relatively more rapidly under scenarios with increased temperature (Kohlmaier *et al.* 1990), provided the water and nutrient supply do not become limiting. Jenny (1980) has also shown that soil organic matter content decreases as temperature increases along precipitation transects. One way to proceed in understanding these complex interrelationships is to use computer simulation (Goldewijk *et al.* 1994; Schimel 1995). Sampson *et al.* (1993) estimate the current C flux from grasslands, savannas and deserts at between 0 to +0.6 Pg C $yr^{-1}$ (+ = sink), which would change to -0.3 to +0.1 Pg C $yr^{-1}$ under 'doubled $CO_2$ climate', and to a sink of +0.1 to +0.5 Pg C $yr^{-1}$ with 'doubled $CO_2$ climate and optimum vegetation management'.

Soil salinity and salinization are affected by both precipitation and temperature changes, because of the direct link between leaching, evapotranspiration and salinity (Szabolcs 1991; Varallyay 1994). Even without climatic change, there would be an increase in dryland salinisation in the near future (West *et al.* 1994).

Wind and water erosion in drylands are expected to increase over the next decades, but it remains difficult to say how the rate will respond to different scenarios of climatic change (West *et al.* 1994). While water-driven erosion is likely to decrease as rainfall decreases, total erosion could increase as wind-driven erosion increases. Increased occurrence of extreme weather events such as storms, however, could increase the erosive impact of scarce rainfall showers. Thereby, the overall effect of water erosion can still be high.

## 5 Discussion and Conclusions

This paper estimates the present mass of carbon and nitrogen in the soils of the world, with special attention to dryland areas. Although current estimates of SOC mass in the 1.0 m depth range (un-corrected for coarse-fragments) are similar to those computed by Eswaran *et al.* (1995), large differences in estimates can be observed when individual soil units are compared. With respect to estimates of carbonate-carbon and soil nitrogen, the uncertainty is even higher. The estimates presented in this paper are average values suitable for global assessments. They are considered unreliable for presenting national statistics, which require regionally explicit data sets. Revisions will be made as new data for under-represented soil units are added to the WISE database.

The C and N data presented, were compiled from field samples collected during the last 20-30 years. As such the information does not represent the carbon content of the soils of the world at any single moment in time. As a consequence these carbon data cannot be used directly to assess changes in soil C during this period, for which paired studies are needed (Mann 1986; Veldkamp 1993; Ihori *et al.* 1995). The soil carbon data held in the WISE database, however, can provide primary input for process-based models to predict changes in carbon content resulting from land degradation, climate change and management practices (Goldewijk *et al.* 1994; Schimel *et al.* 1995). A data set, with a first selection of profiles derived from WISE has been released to the scientific community for such purposes (Batjes 1995). It has been proposed to serve as the basis for a global soil database to be developed by IGBP-DIS (Scholes *et al.* 1994).

Refinement of the WISE database is scheduled to continue by incorporation of new profiles, notably soils collected worldwide in the framework of ISRIC's programme on National Soil Reference Collections and

Databases. Several spatial data sets (e.g., soil pH, organic carbon, soil water retention properties), with a resolution of ½° latitude by ½° longitude, have been prepared using the format of the Global Ecosystems Database (Kineman 1992). On the longer-term it is envisaged that the WISE database could be used to refine studies of crop production potentials (e.g., Luyten 1995) and soil gaseous emission potentials (e.g., Bouwman *et al.* 1994; Bachelet & Neue 1993) which of necessity used the 1° by 1° resolution data file which Zobler (1986) derived from the original, printed Soil Map of the World (FAO-Unesco 1974).

The need remains for additional information on the world's soil resources. This is critical as parts of the Soil Map of the World (FAO-Unesco 1971-1981; FAO 1991) are outdated (Sombroek, 1990; Oldeman & Van Engelen, 1993). An internationally endorsed methodology (SOTER) for such a global update has been available since 1993. The SOTER procedures (Van Engelen & Wen 1995) are currently being used by UNEP, FAO, ISRIC and several soil survey organizations to update the South and Central American section of the Soil Map of the World, inclusive of the Caribbean. Similar activities, at scale 1:5 M, are planned for North Asia (Russia, China and Mongolia) as a joint activity of FAO and IIASA. The current update for the Southeast Asian Soil Degradation Assessment (UNEP, FAO, ISRIC) may be used as a physiographic template for a Southeast Asian SOTER.

A full update of the information on the world's soil resources in a 1:5 M scale SOTER would provide the IGBP community with much of the primary soil data it would require for its global studies of terrestrial agro-ecosystems. Also these databases can be used with dynamic models and GIS techniques to identify areas vulnerable to specific types of land degradation and land use change scenarios (Batjes *et al.* 1993)

Human activities have led to about $1964 \times 10^6$ ha of degraded lands in the world (Oldeman *et al.* 1991), many of which occur in dryland areas (UNEP 1992). In these soil organic matter content is decreasing and as a result cation exchange capacity, water holding properties and structural stability decrease also, seriously reducing the possibility for sustainable agriculture. Nonetheless, much can be done by human beings to manage ecosystems, to adapt farming systems and to reduce the impacts of land degradation and climate change. It is likely that vegetation-based C-management or mitigation measures (e.g., afforestation, limited tillage) cannot fully offset the existing anthropogenic disturbances of the global C cycle. The potential for the terrestrial biosphere to store carbon will also be limited by degradation of natural vegetation and soils in response to human population growth and agricultural expansion, notably in the tropics. As the population continues to increase, the potential for vegetation shifts and $CO_2$ fertilization (Bazzaz & Fajer 1992; Idso & Idso 1994) to create a carbon sink on land (Sombroek *et al.* 1993) may become largely academic (Schlesinger 1990) since large sections of originally fertile land may become too degraded to be reclaimed. Also, it has been argued that under increased atmospheric $CO_2$ concentrations and higher temperatures, nutrients may become limiting for net primary production and thereby ultimately reduce soil C stocks. Process-based models are needed to provide possible answers to the above questions (e.g., Solomon *et al.* 1993; Goldewijk *et al.* 1994; Schimel 1995). While the amounts of carbon that can be sequestered in vegetation and soils in semiarid regions is severely limited by low rainfall and generally high temperature, small changes in total amounts will be important for global warming in view of the large extent of semiarid lands. This would require excellent conservation and management of the land under consideration.

Improving the science base for soil and ecosystem management, with attention to the C, N and P cycle impacts of management practices, is needed for all agro-ecosystems to help managers understand and incorporate C-related management factors into land use decisions (Sampson *et al.* 1993; Campbell & Zetner 1993). Several priority issues need to be addressed as reviewed by Beaumont (1989) and Lal & Kimble (1994).

Policies must be developed to reduce the fundamental causes of land degradation and greenhouse gas increases: non-sustainable land use practices and increasing fossil fuel use, both of which are driven by growing human population and economic development, must be curtailed. In assessing prospects and time scales for combatting land degradation in dryland regions, it must be kept in mind that the strategies formulated by scientists and planners can only be effective with the willing participation of the rural people directly concerned. Many projects have failed because planners did not find out the local communities affected what their views and needs are. Hence the need for a holistic design and a participatory approach that directly involves the local community in the planning and implementation process (Kotschi & Adelhelm 1986; Westing 1994; Catizzone & Muchena 1994). The administrative and logistical challenges of implementing a truly participatory approach, however, are considerable requiring a concentrated effort.

## Acknowledgements

The invitation by UNEP is gratefully acknowledged. Special thanks are due to F.O. Nachtergaele (FAO) for developing the spatial griding algorithms and to E.M. Bridges (ISRIC) for contributions in compiling the WISE profile database. I thank J.H.V. van Baren and E.M. Bridges for their constructive comments. The final GIS maplets were prepared by Ms. J. Reesink. The WISE database has been developed at ISRIC with initial sponsorship from the Netherlands National Research Programme on Global Air Pollution and Climate Change.

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# The carbon budgets in drylands: assessments based on carbon residence time and stable isotope formation

**H.W. Scharpenseel, J. Freytag and E.M. Pfeiffer**

## Synopsis

This paper examines and reviews knowledge on the magnitude and nature of important carbon pools, sinks and sources. The carbon compartments of soils from semiarid and arid regions are characterised according to soil Sub-orders or Soil Groups. Values are tentatively extracted and integrated from published -C estimates and from field studies by the authors in North Africa, India, Australia and Argentina.

## Key points

1. Dryland soils cover about one-third of the world's ice free land surface but receive less attention, because of their lower productivity, than the soils with a more favourable moisture distribution. This applies also to assessments of the soil organic matter and the carbon budget.
2. The highest carbon pools are in Histosols with others ranked Iceptosols > Oxisols > Alfisols > Aridisols. Despite their lower ranking, Aridisols, plus other soils of the semiarid and arid climatic belt, may sequester more carbon than estimated so far. This is because of recent as well as fossil carbon sequestration in the lower layers of dryland soils i.e. in paleosols and in carbon from deep roots of drought-resistant perennials
3. Alternating pluvial and arid phases in much of the world's drylands, especially North Africa, means that there is much sequestered fossil soil organic carbon in the 0-100 cm layer. New estimates favour raising the published values above 110 Pg C for Aridisols and 184 Pg C for other dryland soils.
4. The carbon content of dryland soils depends on the total level of precipitation and on rainfall distribution. The high potential evaporation rate typically converts small amounts of more equally distributed rain mostly into water vapour, only poorly into biomass and even less so into phreatic groundwater levels and deeper groundwater stores.
5. Carbonate carbon forms a significant pool with an annual sequestration of carbon in soil carbonates/caliches of 10 Tg C. $y^{-1}$ About 1000 Pg of soil carbonate is present as either finely dispersed or enriched forms in carbonates/caliches derived from lithogenic carbon and respiratory derived carbon. With rising atmospheric $CO_2$ concentration the $CaCO_3$ solubility will increase considerably, though synchronously rising temperature by greenhouse forcing will produce a lowering effect of similar magnitude.

Key words: North Africa, India, Australia, Argentina, paleosols, caliches, vertisols, carbon dating

## GENERAL OVERVIEW

Dryland soils have generally received less attention from soil scientists, because of their lower productivity, than the soils with a more favorable moisture distribution. This applies also to assessment of the soil organic matter (SOM) and the carbon (C) budget. In the light of current concerns about global climate change and the need to sequester carbon this neglect may prove to be unwarranted. For example, Table 1 contains a representative sample of some important C pool sizes, sources and sinks for comparison and reference.

The C compartments of dryland(semi)arid climate-related soil Suborders or Soil Groups are calculated from soil -C estimates by Eswaran et al (1993) (Table 2) and

Table 1: Carbon pools, sources and sinks , an overview

| CARBON COMPARTMENT | MILLION HA | C - POOL in Pg | SOURCE/SINK in Pg $C.y^{-1}$ | AGE / RESIDENCE TIME ESTIMATE |
|---|---|---|---|---|
| $CaCO_3$ in sediments 3), 7) | | $35 \times 10^6$ | | |
| $CaMg(CO_3)_2$ in sedimenst | | $25 \times 10^6$ | | |
| Organic sediments,kerogen-C | | $15 \times 10^6$ | | |
| $HCO_3^- + CO_3^{2-}$ in ocean water | | $38 \times 10^3$ | | |
| in upper 100 m | | 700 | | |
| Fossil C (from oil,gas,coal) | | $10-15x10^3$ | | > Holocene |
| Foss.C econ. accessible 6) | | ca 1000 | | |
| C in atmosphere | | ca 740 | | ca 118 % modern |
| C in living biomass | | ca 650 | ca 1.2 S | ca 120 % modern |
| Org.C in pedosphere 0-100 cm | 4) 12,800<br>8)<br>5) | 1430<br>1180<br>1550 | | only as paleo-sols > 10000 BP |
| Pedogenic secondary Carbona-te - C, (semi)arid landscape with caliches ) esp. 250-550 mm rainfall | 13)<br>1)<br>12)<br>5) | 1700<br>780-930<br>717<br>1730 | ca 0.01 11) | young carbonate C 1/2 fossil,1/2 modern;measured C-14 age minus 5600 yrs=(T 1/2) |
| Forest,temperate climate | 2,500 | ) 787 | | |
| Forest tropical | 14) 1,700 | | | |
| C in forest vegetation | 14) | 359 | | |
| Forest C-sequestration pot. | | | ca 2.5 | |
| Grassland | 3,200 | 300-350 | 0.3-0.5 9) | > forest soils |
| Wetlands | 1,020 | ca 650 | 20-225 g $C.m^{-2}$ 10) | low, fast C re-cycling |
| Riceland | 136 | ca 15 | < 28°C mild C-sink | |
| Drylands | 5)ca 4,000 | ca300 OM-C<br>ca1200 Carb-C | | |
| Biomass Productivity of crop-land in comparison with same land of indigenous vegetation | 2) | | ca -40 % | |
| C-sink from leaf droppings and vegetative relics | | 18 - 24 | | modern |
| C-source from (wood)clearing | | ca 24 | | ca age of trees |
| Lacustrine C sedimentation | | | ca 0.25 | low age of sur-face near humus |
| Small C sinks by aquatic transport (in total) | | | ca 1 | |

1)Amentano & Menges,1986
2)Aselmann & Lieth,1983
3)Berner & Lasaga,1990
4)Buringh,1984
5)Eswaran & Cow. 1993
6)Graßl & Klingholz,1990
7)Kempe,1984
8)Kimble,1990
9)Scharpenseel,1993
10)Schlesinger,1982
11)Schlesinger,1985
12)Sombroek and Cow,1993
13)Yaalon,1981,a,b
14)Dixon & Cow.,1994

Table 2 Organic C, Carbonate C, Total C in upper 100cm of (semi)arid soils (acc. to Eswaran et al, 1993)

| | | Organic C Pg | | Carbonate C Pg | | Total organic + Carbonate C Pg |
|---|---|---|---|---|---|---|
| GLOBAL | | 1.555 | | 1.738 | | 3.293 |
| (Semi)arid soils ARIDISOLS | | 110 | | 1.044 | | 1.154 |
| Other (Semi)arid soils (TORR-, XER-, UST-) | ERT $32^{3}$ | | $24^{3}$ | | $56^{3}$ | |
| | ULT $42^{3}$ | | $0^{3}$ | | $42^{3}$ | |
| | ALF $59^{3}$- | 184 | $71^{3}$- | 184 | $130^{3}$- | 364 |
| | OLL $30^{3}$ | | $55^{3}$ | | $85^{3}$ | |
| | (PSAMMENT) $21^{3}$ | | $30^{3}$ | | $51^{3}$ | |
| (Semi)arid soil-C ARIDISOLS + TORR-, + XER-, + UST- | | 294 | | 1.224 | | 1.538 |

Table 3 Organic C, Carbonate-C, Total C (Carbonate-C in different Soil Groups (acc. to Sombroek et al,1993) (0-100 cm)

| | Organic C, Pg | Carbonate C, Pg | Total Org. + Carbonate C,Pg |
|---|---|---|---|
| GLOBAL | 1.215 | 717 | 1.932 |
| REGIONAL: Xero-, YERMO-, Petrocalcic phases | | 536.7 | |
| Calcaric Fluvisols, Calcaric Regosols, Calcaric Gleysols | | 37.6 | |
| Chernozems, Kastanozems | | 99.4 | |
| Calcic Luvisols, Calcic Cambisols | | 24.7 | |
| Vertisols | | 19.0 | |

At first glance it is surprising that the highest C pools occur, after the first ranking Histosols, in Inceptisols > Oxisols > Alfisols > ARIDISOLS. The darker soils (Mollisols, Vertisols, Andosols, Spodosols), because of their limited areal extent, make the lowest contribution.

But Aridisols, plus other soils of the (semi)arid climate belt, may turn out to sequester even more C then previously estimated. This is based on estimates of recent, as well as fossil C sequestration, in deeper layers of the arid land soils, i.e. in paleosols and C from deep reaching roots. It could be analogous to findings by Fisher et al. (1994) in South American deep rooted savanna grassland. Reports from Amazon rain forests during the dry season (Nepstad et al 1994) show C sequestration down to a depth of 8 m. More C was sequestered below 1m than from 0 to 1m layer.

The significance of C storage in the lower layers of dryland soils should not be overlooked. Evidence of C sequestration is seen in paleosols. For example, palegleyization in old sandy/silty/loessic dunes of North Africa. Here (semi)arid lands very much reflect intermittent alternations of pluvial (> 600mm rainfall) and interpluvial phases. During the humid regime the sturdier vegetation suppressed and annihilated the scanty and more patchy vegetation of the arid phases, so that now large stretches of land are almost devoid of vegetation.

Historical and anthropological evidence is relevant here. Cave paintings in the Algerian Tassili Mountains, depict steppe plants, steppe animals and even sacrificial rites. These originate from three periods. The simplest and oldest period is called the "round head phase" possibly about 20,000 years ago. Then came the mid Holocene "Hunter and Collector phase" and finally, the "Tuareg period", that suggests a steppe-like environment till about 4000 yrs B.P.. By contrast, the arid lands in South West Africa had a sustained arid climate with adapted vegetation. This provided a more continuous, although often sparse and depauperate, vegetative cover with some C sequestration.

## CARBON POOLS; CARBON RESIDENCE TIME IN (PALEO)SOLS AND GROUNDWATER WITH EMPHASIS ON THE NORTH AFRICAN DRY BELT

In most of the North African arid belt wet and arid/hyperarid phases alternated. Evidence for this comes from an assessment of climate records via $^{14}C$-dating. Wet periods are revealed by the C-residence time of organic C in biomass relics as well by the presence of dissolved respirative $CO_2$ in ground water deposits.

A wet phase, ending around 20,000 years ago, was followed by (hyper)aridity in the remainder of the late Pleistocene. After the Alleroed there was a wetter climate, interrupted again by some major but shorter dry spells (Table 5).

There is a low rate of SOM turnover in arid lands. For example, desert scrub and semiarid grassland, ecosystems have a low biomass -C content of 0.3kg $C.m^{-2}$,and 1.5kg $C.m^{-2}$ respectively (Sombroek et al 1993), By comparison, Yermosols and Kastanozems have a SOM -C load of 2.1 and 12.4 kg $C.m^{-2}$ respectively. Despite this, there are features of drylands which make them interesting as potential sinks for carbon. The typical regime for biomass/SOM in soils, the often young sediments of arid lands and the well preserved paleosols as well as the C-enriched argillic horizons of Argids point to a great probability for considerable amounts of sequestered fossil SOM-C. It is almost certain to go beyond the estimated 110 Pg C in Aridisols plus 184 Pg C in other (semi)arid soils from 0 to 100cm depth (Table 2).

Groundwaters provide further evidence. For example, the residence time of C ($^{14}C$-dates) and tritium bomb-$^{3}H$ (as indicators for rejuvenation since the 1950s) in the three groundwater levels, embedded in recent-, Mio/pliocene- and Cretaceous sediments, plus also in recent soil and paleosol are worthy of study. Figure 1 shows $^{14}C$-dates (as a histograph) for sites from Tunisia (Scharpenseel & Ohling, 1974; Scharpenseel et al, 1980; Scharpenseel & Schiffmann, 1985).

The carbon content in arid lands depends on the total level of precipitation and on the rainfall distribution. Because of the high PET (Potential Evapo-Transpiration) small amounts of more equally distributed rain is mostly converted into water vapour, only poorly into biomass and even less so into refill of phreatic groundwater levels and deeper ground water stores. Bomb-$^{3}H$ (tritium) measurements and also $^{14}C$-dates of the dissolved respiratory $CO_2$ reflect the recent to young ground water age in many areas.

There are various views about the fate of the so-called "Missing Carbon" (Hudson, Gherini and Goldstein 1994; Gifford 1994) Large inputs will go into more biomass/vegetation of higher latitudes as well as into humid tropical ecosystems. There will be some contribution to dryland biomass increase, possibly in conjunction with rainfall shifts (Table 4) This aspect is little explored but needs more attention and basic investigation.

**Figure 1. Histogram of all hitherto produced $^{14}C$-dates (soil, sediment, groundwater) in Tunisia (soil and groundwater dates by Scharpenseel, dates of sediments and fossils by various other authors; Note: Groundwater dates corrected according to Tamers, 1967.**

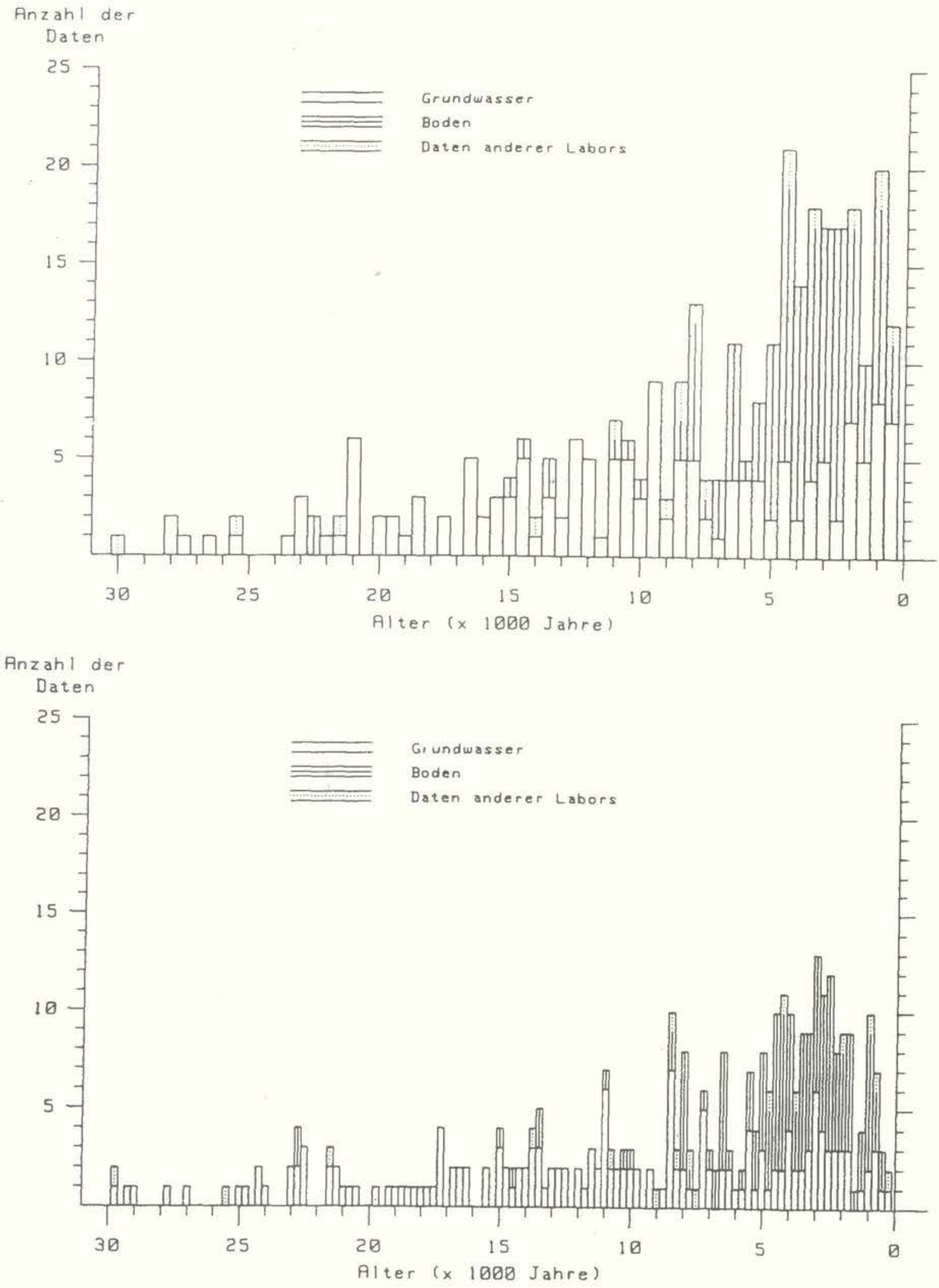


Table 4 Alternatives for the organic carbon cycle's missing C fraction (1.5-2.5 $Pg.y^{-1}$)

| UPTAKE BY OCEANS | ADDITIONAL TERRESTRIAL BIOMASS FORMATION |
|---|---|
| KEELING, PIPER & HEIMANN, 1989: Carbon used to sustain oceanic C pump N -> S while preindustrial atmosphere had gradient S -> N of ca 1.0 Pg $C\text{-}y^{-1}$ | WATSON, RODHE, OESCHGER & SIEGENTHALER, 1990: MCF -> mainly enhanced biomass production |
| BROECKER & PENG, 1992: Confirm Keeling, Piper, Heimann . Deep water N-Atlantic conveyer circulation estimated to transport preindustrially ca 0.6 Pg $C.y^{-1}$, rising trend with modern anthropogenic C inputs | TANS, FUNK & TAKAHASHI 1990: MCF goes mainly in rise of biomass production in northern higher latitudes |
| HARRISON, BROECKER & BONANI, 1993: Reflections about "Missing Carbon Fraction" vs $CO_2$ fertilization vs SOM loss by cultivation against background of SOM/biomass functional compartments, C-residence time, based on natural $^{14}C$-measurments . Kind of compromise between ocean uptake and rise of biomass production. | ESSER & LIETH, 1993: Rise in biomass production exceeds wood clearing and land use related $CO_2$ output (global) |
| | ENTING & MANSBRIDGE, 1991: MCF sink goes also in more tropical biomass |
| | MELILLO, McGUIRE, KICKLIGHTER, MOORE, VOROSMARTY & SCHLOSS, 1993: NPP increase under rising $CO_2$ in tropical forests ca 50 % |
| | NEPSTAD, de CARVALHO, DAVIDSON, JIPP, LEFEBVRE, NEGREIROS, da SILVA, STONE, TRUMBORE & VIEIRA, 1994: Tropical forest (Amazon) with rainy and drier season subsoil-C by deep roots till 8 m depth; more C below 1 m than 0-1m depth |
| | FISHER, RAO, AYARZA, LASCANO, SANZ, THOMAS & VERA, 1994: Large C sequestration by deep rooted grasses in S-American savanna ; annual C sequestration 100 - 507 Tg $C.y^{-1}$ |
| | SCHOLES & HALL, 1993: NPP increase under rising $CO_2$ up to 25 % in tropical grassland; high importance also of wood burning residues |
| | HESSHEIMER, HEIMANN & LEVIN, 1994: By bomb-$^{14}C$ measurement indication for a ca 25 % lower C-input within the C budget in to oceans |
| | SAMPSON, APPS, BROWN, COLE, DOWNING, HEATH, OJIMA, SMITH, SOLOMON & WISNIEWSKI, 1993: In grasslands + savannas + deserts C flux of 0 to 0.6 Pg $C.y^{-1}$ as C sink ( $\geq$ 15 % of MCF). |

Table 5 C-[14] dates for wet and dry phases with various impact om C-sequestration (Tibesti, Chad, Central & East Sahara, W Nubia)

| REGION OF N AFRICAN ARID BELT | $^{14}$C-DATES PHASE OF $^{14}$C DATES | REFERENCE |
|---|---|---|
| S of Tibesti, deltaic sediments | ca 10 160 B.P. | Servant, 1970 |
| late Pleistocene/Holocene | ca 4 500 B.P. | Ergenzinger, 1978 |
| N and S of Tibesti lakes and aquaeous fluxes similar to S of Tibesti | | Pachur, 1975 |
| Datable material in Fezzan | < 3 000 B.P. | Pachur, 1975 |
| SW of Tibesti, Caliches-C only | > 20 000 B.P. | Hagedorn, 1971 |
| Great Sand Lake, Egypt, carbonate-C | > 18 000 B.P. | Hagedorn, 1971 |
| C in carbonates | ca 32 700 B.P. | Pachur & Brown, 1986 |
| Central and East Sahara wet phase | Holocene | Pachur, 1987 |
| Earlier hyatus hyperarid | Late Pleitocene | Pachur, 1987 |
| preceeding wet phase $^{14}$C-datable | ca 25 000 B.P. | Pachur, 1987 |
| C-14 dates of mud in ancient lakedeposits, f.ex. Wadi Howar, E Sahara | 5 000->9 000 B.P. | Pachur, 1987 |
| W Nubia, Sudan; presently arid to hyperarid < 20mm N, $y^{-1}$ | | |
| Wet phase in E Sahara | Late Pleistocene | Pachur & Hoelzmann, 1991 |
| Lake and swamp environment in W Nubian Basin of ca 20 000km$^2$; semiaquatic landscape | 9 300 - 4 000 B.P. | Pachur & Hoelzmann, 1991 |
| Lake sediments | 30 000 - 21 000 B.P. | " |
| Phase of hyperaridity without $^{14}$C-dates, similar to present climate regime, with eolian deflation | 21 000 - 11 000 B.P. | " |
| Sets of $^{14}$C dates, prepared mainly by M.A.Geyh, more moisture | 10 000 - 3 000 B.P. | Geyh & Jäkel, 1974 |
| Beyond 21 000 further $^{14}$C-dates mainly of lake carbonates | > 21 000 B.P. | " |
| Chad Basin, major dry spells | 11 000 - 13 000 B.P.<br>8 000 - 9 000 B.P.<br>3 000 - 4 000 B.P. | Zahn, 1994 |

## PROCESSES OF CARBON DYNAMICS IN VERTISOL AND CALICHE FORMATION

Vertisols are possibly the soil order with deepest organic C penetration due to the self mulching process in the course of the shrinking during the dry season. It correlates to the clay content and the share of HAC and LAC clay minerals. The SOM, mostly complexed with the clay minerals, resists alkali extraction and dissolution quite strongly and must belong to the "passive C fraction" with its superior C residence time (Parton et al, 1987; Ojima, Stafford Smith and Beardsley, this volume).

In the Sudan a great geomorphological inhomogeneity of wadis, terraces, limnic sediments and piedmont plains (pediments, glacis) corresponds with a great diversity of soil carbon (SOM as well as $CO_3^{2-}$-C) load. After an uplift during the turn of the Cretaceous to the Tertiary and some basaltic volcanism the Nubian sandstone was partly eroded and a pre-Nubian basement peneplain emerged.

In the middle or lower Pleistocene a pediment developed with some isolated Inselberg complexes and increased incision of the river Nile and its tributary wadis. Parallel to this development, before 20,000 B.P. fine wadi sediments were deposited. Semiarid conditions have dominated since the turn of Pleistocene and well into Holocene times. Strongly argillic soils emerged mainly on the terraces, even some fresh water lakes developed around 7000 B.P. (Pflaumbaum, 1987). The dominantly smectitic (however with quite different %-rates of kaolinite as well as grades of salinity and alkalinity) Vertisols of Gezira and Nile Alluvium are low in C, mostly < 0.5 %, rarely reaching 1 % C. From the total area of distribution they are intermediate to the two other largest Vertisol areas in Australia and the Dekkan Plateau of India, comprising about 40 mil ha, ca 925,000 ha irrigation land (Dudal, 1965), having been formed according to Tothill (1946) since Alleroed time (ca 11.000 B.P.).

On balance though, the Vertisols despite the depth to which SOM layers reach and the dark colour are low in organic C concentration. Their total store of 32 Pg organic C is only ca 2 % of the global 1.555 Pg organic C in the Pedosphere (Eswaran at al, 1993, Table 2). Despite this moderate to low SOM content in most Vertisol areas the turnover of biomass, according to studies in the Indian fringe area between Usterts (basaltic Dekkan Plateau) and Ustalfs (southward adjacent granitic / biotite-gneissic area) under use of uniformly $^{14}C$-labelled sorghum and peanut straw (Singer & Scharpenseel, 1993; Scharpenseel & Singer, 1992) shows higher stability, slower decomposition and higher C-residence time of the biomass/SOM in the smectitic Vertisols than in the kaolinitic Alfisols (Table 7).

An analysis of "age versus depth down to 600 cm" comprising Vertisols and country of origin regression lines of Germany, Sudan, Tunisia, Argentina, Israel, Bulgaria, Italy, Spain, Portugal and Australia indicates residence times mostly up to 10,000 B.P. (some until 20,000 B.P). Frequently the C residence time increase with depth is only slight in the self-mulching depth range with a subsequent steeper increase to greater depth (Scharpenseel et al, 1986). The steepness of the regression line, indicates that age of the C stores increase with depth. Overall, the following order emerged : Bulgaria > Argentina > Germany > Israel > Italy > Australia > Spain > Portugal > Tunisia > Sudan (Figure 4, Table 7). There is a confounding effect of rejuvenating younger C. Figure 5 shows the relationship between sample provenience vs steepness of age/maximum profile age to depth. The order now becomes : Portugal > Argentina > Spain > Germany > Australia > Italy > Bulgaria > Israel > Sudan > Tunisia (Fig. 5). To derive an understanding for C residence time, sustainability of SOM presence in the deep humus Vertisol profile, the steepness of age versus depth or age/maximum profile age to depth are valuable parameters.

Figure 2. Age versus depth relationships in vertisols

Fig. 4

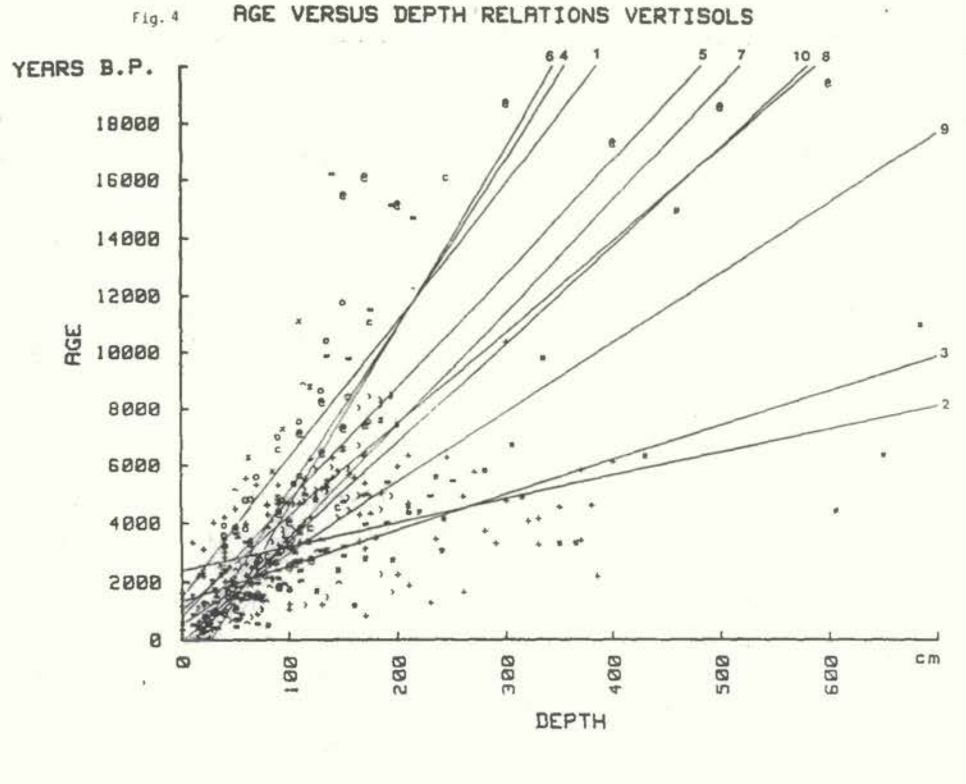
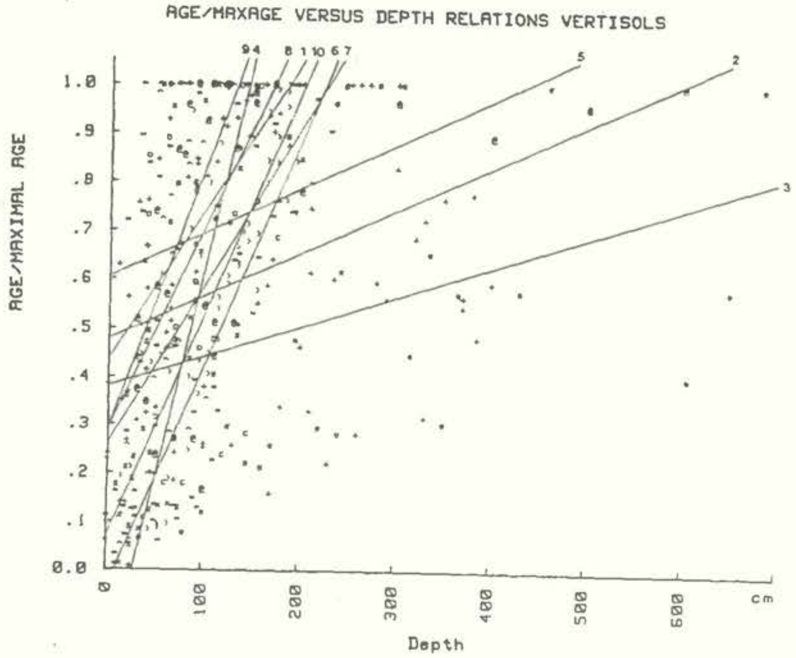


Figure 3. Age/maxage versus depth relations in vertisols




In the 350 mm rainfall area of Kaniva, Victoria, Australia (locations Miram and Lillimur) $^{14}C$ dating of Vertisols with a gilgai relief as a profile scan (82 measurements in 4 profiles) was distinguishing between analog profiles in gilgai mounds and gilgai depression sites. In both locations the C-residence times in the mound position exceeded the analog ones of the depression-profile samples (Blackburn et al, 1979).

The Vertisols are mostly old cropland soils and have often undergone various phases of vegetation with different mechanisms of photosynthesis. Freytag (1985) as well as Scharpenseel et al (1986) describe layerwise d$^{13}C$ measurements in Gezira-Vertisols of Sudan. In the lowest section (Figure 6) a d$^{13}C$ there is evidence of a mix towards $C_3$ and CAM mechanism (-25 ‰, respectively -17 ‰ of d$^{13}C$). These derive from the old Nile alluvium. Closer to the surface, the longer $C_4$ durra refelects a period of indigenous land use. Nearer the soil surface,there is a decrease of d$^{13}C$ due to the impact of the British cotton industry. Cotton is a $C_3$ plant. The low, very stable humus content of 0.5 - 1.0 % was sustained during the various culture/vegetation changes.

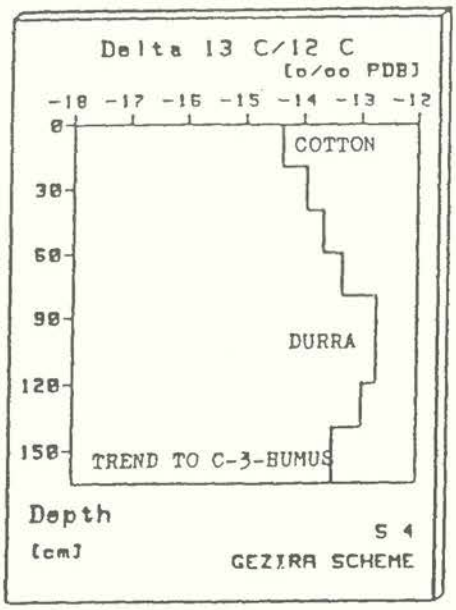



Figure 4 $\sigma^{13}C$ profile curve of a Sudanese vertisol

## EFFECT OF SOIL MANAGEMENT ON SOIL C LOSSS

Stewart (1995) in a recent overview of soil management in semiarid regions refers to a 30 months incubation test of 19 surface soils, revealing quite different ranges of C loss : 22-33 % vanished in soils dominated by kaolinites (see also table 6 from semiarid India), 16-33 % from soils of smectitic clay mineralogy, 13-20 % from Oxisols with mainly oxidic SOM complexes and only 2 % from Andosols with their stable allophane / SOM complexes. For semiarid regions estimates are < 10,000 kg soil organic matter-carbon exist in 0-15 cm depth. Also the return of C, about parallel to wheat grain yields of < 1000 $kg.ha^{-1}$, to the soil is low, of the order of 300 kg $C.ha^{-1}$. Such C return would, according to Stewart (1995), just suffice to maintain the soil organic matter level in a soil of about 0.6 % organic C (see also Grace et al; this volume). Cole et al (1993) see the C sequestration potential in semiarid soils near 0.5 kg C per square meter.

Since the 0.2-0.5 % SOM in arid, the 0.5-2 % SOM in semiarid soils are often deep-seated, due to the penetration depth of roots of the drought-resistant plants, conversion of macchia and garrigue to pasture or cropland requires a cautious approach. Inevitably, crops and pastures need fertilizers. In some legume-based pastures merely applying phosphate fertilizers is sufficient.

The importance of the SOM-content for maintenance of soil fertility has been stressed by Tiessen et al (1994). They used C turnover rates and $^{14}C$-dating to derive an economic span of land use (without supplementary fertilization) of about 65 yrs in temperate prairie soils, and about 6 yrs in tropical semi-arid forest ecosystems.

### *Climatic variability and its consequences*

The extreme climatic variability of semiarid lands and the consequences are stressed by Harrison (1987). Drylands are chracterised by unpredictable long drought periods, e.g. the Sahel with its record of dry spells in the early decades of the 19th century, in the 1960s and 1970s. In southern and eastern Africa and in Australia there are alternating dry and wet periods on a cycle of about 8-15 years. The ENSO (El Nino Southern Oscillation) phenomenon is implicated in this (Squires 1995).

A drastic change of albedo, aerosol concentration and biomass production occurs as alternating cycles of wet and dry occur. Under population pressure the alteration of the ecological conditions leads (during moist periods) to migration phases into the normally arid regions. These have limited cropping and grazing potentialin normal years. These migrations cause hardship to the original nomadic population in the arid fringe areas because they find new settlers in their old grazing lands. When a phase of rising aridity approaches the land is often denuded by the new settlers as drought and famine overtake them (Ayoub, this volume).

## THE CARBONATE CARBON

The ca 1000 Pg of soil-carbonate-C (Table 1) are manifested in finely dispersed distribution or enriched in form of calcretes/caliches as a mixture of lithogenic C from primary carbonates with $d^{13}C$ near 0 ‰ and respiratory C derived of the phases of vegetation during the more humid periods with $d^{13}C$ between -25 - 28 ‰ or -10 - 11 ‰ ($C_3$ plants -25 - -28 ‰; CAM plants -17 ‰; $C_4$ plants -10 - -12 ‰).

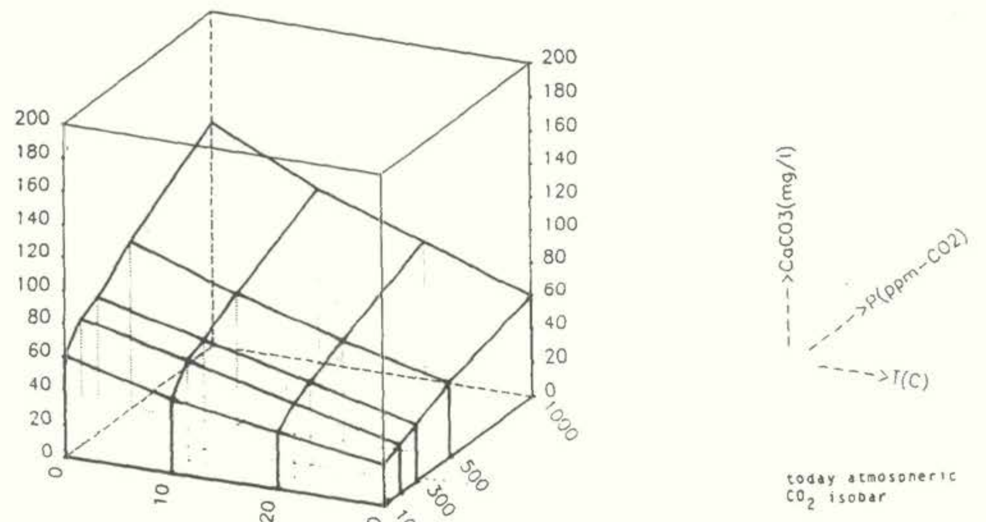
Due to strong representation also of CAM and $C_4$ plants (especially in saline environments) the $d^{13}C$ of caliches can vary over several ‰ between about -13 and -6 ‰. While finely dispersed carbonate -C can be also directly related to the lithogenic carbonates of the soil's parent material, the carbonate in the caliches depends almost always also on organic C sources, contributing the $CO_2$-C required for transient dissolution of the lithogenic carbonates into bicarbonate before reprecipitation with diminishing soil moisture.

Obviously, increasing C sequestration in caliches depends entirely on the additional $Ca^{2+}$ release by weathering processes. This alone can increase the total primary carbonate plus bicarbonate plus secondary carbonate pool. Major phases of caliche production in North Africa were the Villafranca (transition from Tertiary to Quarternary) and Tensiftien (Rissian, Illinois). The mechanism of caliche formation was studied by layerwise measurements of $d^{13}C$ and $d^{18}O$ in caliches of southern Tunisia (Freytag, 1985, as well as Schleser et al, 1983). Schlesinger (1985) estimates the annual sequestration of carbon in soil carbonates/caliches as approximating 10 Tg of C (see also Schlesinger, 1982), the author arrived, based on 4-6 bil ha and storage of 0.5 g $C.m^{-2}.yr^{-1}$ worldwide at 23 Tg C $.y^{-1}$).

Figure 5 shows typical examples of caliches formed by the ad ascensum, ad descensum and catenary mechanisms. The ad ascensum caliches near the surface probably form in a climate with an extended rainy season without an abrupt end to the season. The dissolved bicarbonates migrate slowly to the surface with the decreasing rains and are finally precipitated as a mix of primary carbonate and originally organic respiratory C. Isotope discrimination towards higher $d^{13}C$ and $d^{18}O$ in the caliche develops near the surface. The ad descensum caliches form under conditions where the rainy season is shoter and the end more abrupt. Reprecipitation occurs near the maximum depth of rainfall penetration. There is a lack of time for the slow upwards movement. The isotope discrimination is visible in that greater depth. Catenary caliches, parallel to slopes, show no isotope discrimination.

Figure 6 shows the solubility of $CaCO_3$ in mg/lt in pure water as a function of $CO_2$ pressure and temperature (after Schleser, 1986, personal communication). It reveals, that with rising atmospheric $CO_2$ concentration the $CaCO_3$ solubility would increase considerably. Although synchroneously rising temperature (by greenhouse forcing) would produce a lowering effect of similar magnitude.

Figure 6 Solubility of $CaCO_3$ ( mg\l) in pure water (seawater chacteristic is similar) as a function of $CO_2$ pressure and temperature



## POTENTIAL OF C SEQUESTRATION, MITIGATION OF LAND DEGRADATION

Non forested drylands occupy about 43 % of the total land surface. About 70 % of this vast area is affected by the process of desertification, and 3.5 % is lost annually for economic production. Glenn et al (1993) believe, that by revegetation and other innovative methods in the rangelands a net C sequestration in dryland soils of 0.5 - 1.0 Pg $C.y^{-1}$, at costs of $ 10-18 per ton of C could be achieved.

Any estimate of potential C sequestration is inherently linked to human endeavours geared to improve the soil moisture conditions. Drylands, due to the superior light climate, can, with proper water managenment, be highly productive. Biomass production by intensive agriculture in desert fringe loess or older loamy/marly alluvium soils can be 25 t/ha (Kopp, 1975). Drylands with supplemental irrigation, that is economised and optimised by neutron probe scan against the background of the known pF curve (suction potential) can exceed productivity and C sequestration of most fertile Mollisols.

Even hyperarid regions such as the Sahara have vast water resources (ca 40 bil $m^3.y^{-1}$) Of this, 10-20 bil $m^3$ are estimated to reach the ground water. Ground water reserves exceed $15 \times 10^{12}$ $m^3$ in the Sahara, mainly below the sandy Ergs.

Mitigation efforts against the rise of $CO_2$ as well as wind erosion and desertification, are always intrinsically correlated with the capability to provide enough fresh and/or low saline water as well as modern water management know-how. The so-far neglected efforts of sea water desalinization could also add significantly to the water available for biomass production in dryland areas. Improving economy and techniques of sea water desalinization by solar energy could be a major challenge for improvement of food production and C sequestration in the (semi)arid potential cropland and pasture areas (Hodges et al 1993).

## ACKNOWLEDGEMENTS

We are grateful to UNEP for the invitation to attend the Workshop. One of us (HWS) received a travel grant. Dr Vic Squires assisted with the editing of the manuscript.

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# Factors affecting carbon storage in semi-arid and arid ecosystems

**D. S. Ojima, M. Stafford Smith and M. Beardsley**

Synopsis

The objective of this paper is to identify the factors contributing to the degradation of arid and semi-arid ecosystems, An evaluation of the land use practices that contribute to recovery of degraded lands and the implications for increased storage of soil carbon and its sequestration in degraded ecosystems is also made. In this study the CENTURY model (a model of plant-soil ecosystems that simulates the dynamics of grasslands, forests, savannas and field crops) was used to simulate nine different arid semi-arid ecosystems and different land management practices. Data from nine case studies, on sites ranging from cool dry regions in the semi-arid and arid areas of Asia to tropical regions in Africa and Australia, are presented.

Key Points

1. Soil carbon storage is one of the largest pools of carbon in the ecosystem. Soil carbon cannot be considered a homogeneous mixture of organic material. Instead it must be viewed as a mixture of labile and more resistant material in order to fully understand the dynamics of soil carbon relative to climate and land management changes. The short term fluxes of carbon from soil are derived primarily from changes in the labile soil organic matter pools. The size of this pool is relatively small but it accounts for more than 66% of the soil carbon respiration under "steady-state" conditions.

2. Soil texture influences the relative proportions of slow and passive (recalcitrant) soil carbon. Clayey soils tend to accumulate more passive soil carbon than do sandy soils. When a change in land management occurs, the slow pool often is affected the most. Changes in the slow pool will not only affect the carbon flux from an ecosystem but it will also change soil fertility levels. Changes in long-term productivity may follow.

3. Climatic factors affect a number of key processes of an ecosystem that determine carbon fluxes and storage. For example, decomposition rates, plant production, and respiration rates of soil and vegetation. Rainfall amount tends to control the plant biomass whereas temperature determines decomposition rates. Rainfall and temperature interact. Climate also plays an important role in determining the frequency and intensity of disturbances. Lightning, wind storms and pest outbreaks are often associated with changes in the climatic factors. Changes in climatic variability may also have adverse impacts on ecosystem dynamics and on carbon fluxes from an ecosystem.

4. Changes in land use practices can also dramatically alter carbon storage and flux rate. Conversion of forests or grasslands to cropland can result in rapid decline in carbon stores (Tishkov, this volume; Grace, this volume). Up to 50% of the soil carbon and woody plant biomass of a forest can be lost in the process of conversion to cropland. Land under field crops, forest plantation or rangeland can vary greatly in carbon storage and fluxes.

5. Forests and rangelands under good management and proper rotations may effectively be neutral relative to carbon fluxes when averaged over a large area. Croplands are vulnerable to large carbon losses due to removal of vegetation and soil disturbances on an annual basis. However, not all cropping

management practices have this impact. Recent no-till practices have been developed to lessen the impact on the soil carbon losses from croplands (Grace, this volume). The historical land management usage greatly determines the current carbon storage and potential fluxes from these managed systems.

**Key Words: global carbon cycles, land use change, savanna, Africa, prairies, carbon flux, modelling**

## Introduction

The ecological community is faced with an enormous challenge over the next decade of developing an understanding of the interactions between global change and terrestrial systems. Marked alterations in the Earth's environment have already been observed in climate, atmospheric chemistry, the biosphere, and in patterns and degree of human exploitation of natural resources during the past few centuries (Malone and Roederer 1985, NRC 1986, WCED 1987, IGBP 1986, 1990). These presage even greater changes as the role of human interference in natural processes increases (Houghton et al. 1990).

Global-scale climatic change has, of course, occurred throughout the Earth's history and has caused natural changes in terrestrial ecosystems, for example, during the period following the last glacial maximum (COHMAP 1988). As the ice-sheets retreated, grasslands in the central US developed and trees migrated poleward (Davis 1981, 1989, Huntley and Birks 1983, Shugart et al. 1986). Global mean temperatures have varied less than 1°C during the past 7000- 8000 y (Webb et al. 1985).

Human activity is also a major factor contributing to global change. In addition to changes in ecosystems brought on by natural variations in climate during the past few 1000 y, human activity has taken a larger role in modifying terrestrial ecosystems by altering the composition and distribution of plant communities through purposeful and inadvertent activities. Industrial emissions during the last century have rapidly altered atmospheric composition as evidenced most dramatically by acid deposition rates in Europe and North America and by increasing atmospheric concentrations of $CO_2$.

Land management practices, such as fire, grazing, and land conversion to crop-lands affect ecosystem composition, cycling of nutrients, and distribution of organic matter (Houghton et al. 1983, Clark et al. 1986, Detwiler 1986, Goudie 1989). Growing human populations will continue to exert pressure on terrestrial ecosystems as demands for food, fuel, fiber, water, and other resources increase to accommodate growing needs of the people.

Global scale changes in climate will affect land use patterns, and in turn, land use changes may be an important factor in affecting global change. These activities are in many cases overriding natural controls of the global environment and accelerating rates of global change to an unprecedented level (SMIC 1971, Bolin and Cook 1983, Bolin et al. 1986, WCED 1987).

The complex pattern of climate and human activities that are observed in this region provides a challenging task of developing a management scheme that is able to integrate across environmental and social-political factors. The objective of this report is to characterize these factors controlling carbon dynamics in the terrestrial ecosystems of the semiarid and arid regions and to discuss the role of modeling the biogeochemical feedbacks of these ecosystems.

Semiarid and arid lands are vulnerable to human-induced land use changes (Pielke and Avissar 1990). Agriculture, livestock production, and fuel wood collection alter land surface characteristics and biosphere-atmosphere interaction. These land uses affect soil carbon storage, soil fertility, soil erosion rates, dust loading into the atmosphere, trace gas exchange, and water and energy balances. These changes have also influenced local climate patterns due to changes in ET rates and land surface energy exchange relative to estimated values from pre-disturbance land cover (Charney et al. 1975, Pielke and Avissar 1990).

These land use changes modify biological constraints (plant nutrients, grazing) and these will change at very different time scales from abiotic controls (light, water, temperature), leading to a hierarchical scheme of limiting factors and subsequent feedback to the atmospheric system (Schimel et al. 1991a). We are challenged by the need to understand how short-term (hours to days) adjustments of photosynthesis or stomatal dynamics are constrained by long-term regulation of biological processes related to vegetation dynamics and resource availability. Biological factors, including plant community shifts, affecting N control over photosynthesis and biomass partitioning will change slowly (i.e., months to decades). Abiotic controls, on the other hand, regulate instantaneous rates and will change rapidly. This inter-dependence of short- and long-term regulation is important to understanding how an ecosystem will store or lose carbon in response to changing climatic conditions during transient periods of environmental change.

## Ecosystem Analyses

*Field Studies, Modelling, Regional analyses*

Analysis techniques which link process studies across a range of environmental factors, such as climate, soils, land use, in a geographically explicit manner are currently being used. This analytical structure allows us to assess the regional impact of global environmental changes on soil carbon storage and other ecosystem feedbacks to the atmospheric system (e.g., $H_2O$, $CH_4$, and $N_2O$). This structure takes information from ecosystem process studies and from spatial and temporal data of soils, climate, land use, and remote sensing, and organizes it so that various models can be linked in a dynamic fashion.

This analytical approach provides a mechanism to quantify the level of biotic and abiotic controls on land surface-climate interactions. Regional data bases of climate, soil, and land use characteristics have been applied to represent the spatial distribution of land surface properties across a region using ecosystem models (Parton et al 1987, Burke et al 1990, Schimel et al 1990, Ojima et al. 1993 ). These are linked to remote sensing observations which are used to cross-verify the spatial variation and temporal dynamics of land surface processes that control ecosystem C flux, including plant productivity and canopy nutrient content (Aber et al. 1990, Sellers et al. 1990, OIES 1991).

Modeling enables us to quantify the interactive effects of microclimate, herbivores, and nutrients on land surface/atmosphere exchanges of energy and matter. The biological system (i.e., soil / microorganism / plant system) regulates carbon dioxide, other trace gases, water and energy exchange at the land surface. The extent to which biotic interactions modulate these exchanges is variable but can be substantial. Current models assume that allocation of above and belowground C and N reflects the relative importance of light, water, and nutrients as limiting resources.

Plant functional attributes also play an important role in modeling ecosystem dynamics. Changes in species composition from $C_3$ to $C_4$ vegetation (Ojima et al 1991) or structural changes resulting from shifts between grasslands and savannas affect nutrient dynamics, water utilization, biomass allocation, and other characteristics which modify the biosphere-atmosphere linkage (Eagleson 1986). Modifications of resource use efficiency among various vegetation communities are important to projecting how an ecosystem will respond to increased atmospheric $CO_2$, change in climate, or increases in atmospheric deposition of N. Our ability to model the interactions between these characteristics in a systematic fashion allows us to make better estimates of how the terrestrial biosphere responds to environmental change and to project how this feedback may be enhanced or dampened with modifications to plant attributes.

Development of models that link photosynthesis with other ecosystem properties, such as canopy structure, plant N concentrations, and allocation patterns, would provide a framework to study the rapid abiotic responses and the slower biotic controls of plant structure and nutrient cycling.

Regional modeling is an essential step in scaling plot measurements of biogeochemical cycling to global scales for use in coupled atmosphere-biosphere studies, including evaluation of ecosystem responses to directional climate change. The CENTURY model (Parton et al. 1987) simulates the spatial and temporal dynamics of temperate and tropical grassland, forests, and savanna systems in North America, Eurasia, and Africa. Spatial heterogeneity in the landscape due to topographic characteristics or due to land use factors (e.g., cultivation practices, fire, grazing, etc.) modify nutrient cycling, water and energy fluxes, and biogenic trace gas fluxes. Spatially explicit modeling of the landscape patterns can be implemented with ecosystem models such as CENTURY or LINKAGES (Parton et 1987, Pastor and Post 1988, Schimel et al 1990, Burke et al. 1990). Extrapolation of biogeochemistry to large areas requires the coupling of regional models, remote sensing, and geographical information systems.

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## Key issues between global change-terrestrial ecosystem carbon storage

### *Importance of scale*

The manner in which global change will affect terrestrial ecosystems is complex. Complexity arises from interactions between elements of the global changes, interactions between responses of different processes within ecosystems to change, and the feedback responses back to the global environment. The appropriate use of spatial and temporal scale relationships between these factors can help clarify and simplify this complexity.

*Response time and magnitude of ecosystem change is related to spatial and temporal characteristics of the impact.* The response of terrestrial ecosystems to the climate- weather system is dependent on the spatial scale of the interactions between these systems (e.g., storm-size relative to forest disturbance) and the temporal scale that links the various components (Dickinson 1988, Kittel and Coughenhour 1988, Woodmansee 1988).

Current concerns about climatic change and two-way-interactions between *the biosphere and the atmosphere are changing the kind and spatial scale of* the information needed to understand the response of ecosystems in these interactions. However, the spatial heterogeneity in biotic and abiotic variables across regions makes it difficult to use the small numbers of traditional site-level ecological studies to assess cross-regional effects.

Ecologists are thus beginning to combine studies using large numbers of spatially dispersed sites (Parton et al 1987, 1988, Cole 1988, Schimel et al 1990, 1991a, Burke et al 1990, Ojima et al 1991, Parton et al. 1993) with tools utilizing technologies being developed in geographic information systems and remote sensing.

*Temporal and spatial expression of climate change must be viewed in* seasonal and geographical terms to be meaningful to the ecosystem being studied. Mean annual changes in regional precipitation and temperature patterns have little relevance to plant productivity and decomposition rates, when these changes take place unevenly throughout the annual cycle or geographical region. In addition, ecosystem development is more sensitive to characteristics of climate such as first frost occurrence, duration of the wet or dry season, timing of thaw out, and beginning of the wet season which are not well represented in average values of climatic parameters (Dickinson 1988).

These characteristics are crucial cues or constraints on processes such as germination, growth initiation, senescence, and mortality. Climatic or weather patterns need to be expressed in daily, weekly, or up to monthly time frames to capture these biotic processes. Changes in climate will not be the same in every region of the Earth nor within any specific region; how changes in precipitation and temperature vary geographically and temporally need to be explicitly considered.

Another aspect of the scale of global change impacts is the inherent variability of weather (Rind et al. 1989). The frequency of disturbance for a particular ecosystem is in part a function of these weather patterns. As global change alters weather patterns, the frequency and timing of disturbance may be altered and alter ecosystem dynamics (Overpeck et al. 1990). In addition, fire frequency, pest outbreaks, and other ecosystem perturbations may also shift relative to changes in climatic or biotic properties. This potential increase in variability resulting from global change illustrates the non-linear nature of the Earth system which affect biogeochemical processes as well as ecosystem community development.

Changes in the patterns of disturbance affect both ecosystem structure and function (e.g., plant community composition, physiognomic expression, biogeochemical cycling; Noble and Slatyer 1980, Vitousek 1990, Shugart et al. 1986, Ojima et al. 1990, Cohn 1989). The extent to which weather variability will change is unclear (Rind et al. 1989) and further research into how the frequency of extreme events will change relative to ecosystem development must be carried out.

## Modelling the effects of climate and land use change on grassland ecosystem dynamics

Parton et al (1993) highlight three ways in which changing climate might influence grassland ecosystem function and structure. First, many grassland regions face 'desertification', or the degradation of ecosystems to less productive states.

In many grassland regions, significant reductions in precipitation or increases in temperature may accelerate degradation and lead to the replacement of grassland vegetation by woody species. Soil erosion rates may increase as plant cover decreases, leading to a nearly irreversible loss of productive potential. The above transitions are almost always exacerbated by intense human land use, as occurs in most grassland and savanna regions (OIES 1991, Archer 1994). Desertification can thus occur through both functional (e.g., reduced primary productivity) and structural (e.g., perennial ->annual vegetation) change.

Second, nutrient cycling in grasslands is largely driven by climate. Rates of carbon gain, decomposition and nutrient recycling are all sensitive to temperature and moisture (Parton et al. 1987, Schimel et al. 1990). Soil organic matter storage, the primary reserve for nutrients in grasslands, is sensitive to temperature, and generally decreases with increasing temperature. If temperatures increase in grasslands worldwide, then soil carbon storage is likely to decline.

Declining soil carbon would likely lead to nutrient loss, in the long-term, and could lead to degraded hydrological properties and increased erosion. Losses of soil carbon, and increases in temperature, could also lead to higher trace gas emissions as the soil N cycle is accelerated. Thus, carbon storage and nutrient cycling in grasslands could change significantly if climate changes; indeed interannual variability in climate is clearly reflected in observed and modeled production and decomposition.

Finally, the balance between herbaceous and woody vegetation is sensitive to climate in most grassland/savanna regions, and the productivity and human utility of these systems is very sensitive to this balance.

Globally, changes in shrubland and woodland extent have been widespread over the past century or so, and increases in woody vegetation generally, though not always, decrease suitability for grazing by cattle. In areas dependent upon livestock for subsistence, or income, this is a negative change. It is not necessarily bad to increase woody cover, if livestock grazing practices using goats, camels or other browsers can be implemented, however there are few examples of such a successful substitution.

While the most important driver for woodland expansion appears to be grazing or 'overgrazing', changes in climate (Archer 1994, Schlesinger et al. 1990) and possibly increasing $CO_2$ (Johnson et al. 1992), could increase the rates or extent of grassland conversion. This latter effect would occur through the increase in productivity of $C_3$ photosynthetic pathway shrubs at the expense of $C_4$ pathway grasses.

**Modeling framework**

Ecosystem models have been used extensively during the last 20 years to simulate the impact of different management practices (e.g., grazing levels, fire management and fertilization), and different climatic patterns on plant production, nutrient cycling and animal weight gains for grasslands. We have used ecosystem models for site specific impact assessment, regional impact assessment and the ecosystem models have been incorporated into total assessment models where economic models, social models and ecological models are all included.

Major changes in soil organic matter and nutrient supplying capacity have occurred as a results of conversion of Great Plains grasslands to croplands (Haas et al. 1957, Tiessen et al 1982, Burke et al 1990). Initial values of grassland productivity and soil organic matter levels varied widely across the region from north to south reflecting differences in rainfall and temperature (Jenny 1980, Parton et 1987, Schimel et 1990). Soil texture also had a large effect on productivity and organic matter accumulation.

Soil organic matter is an important integrator of climate, geology, topography, and ecosystem dynamics (Jenny 1980), and net changes in soil C is an integrator of productivity and decomposition and as an index of soil fertility. Soil organic matter in grassland ecosystems is estimated to be an important reservoir of terrestrial carbon (Anderson 1991). Soil organic matter changes due to land use practices and to climatic patterns. The response time of soil organic matter varies according to the perturbation of which pool of soil organic matter is affected.

In order to study the factors controlling soil organic matter and ecosystem dynamics, the CENTURY model was developed (Parton et al. 1987, 1993). CENTURY is a general model of the plant-soil ecosystem that has been used to represent carbon and nutrient dynamics for different types of ecosystems (grasslands, forest, crops, and savannas). A brief description of the model structure and the scientific basis for the model is given here, and a more detailed description of the model is contained in different papers (Parton et al. 1987, 1988, 1993 and Sanford et al. 1991).

The CENTURY model is set up to simulate the dynamics of Carbon (C), Nitrogen (N), Phosphorus (P), and Sulfur (S) for different Plant-Soil Systems. The model can simulate the dynamics of grassland systems, agricultural crop systems, forest systems, and savanna systems, and implement various management practices appropriate for the ecosystem. The grassland, crop and forest systems have different plant production submodels which are linked to a common soil organic matter submodel (Figures 1 & 2). The savanna model uses the grassland and forest subsystems and allows for the two subsystems to interact through shading effects and nitrogen competition.

The soil organic matter submodel simulates the flow of C, N, P, and S through plant litter and the different inorganic and organic pools in the soil. The model runs using a monthly time step and the major input variables for:

(1) monthly average maximum and minimum air temperature,
(2) monthly precipitation,
(3) lignin content of plant material,
(4) plant N, P, and S content,
(5) soil pH and soil texture,
(6) atmospheric and soil N inputs, and
(7) initial soil C, N, P, and S levels.

Figure 1 CENTURY model plant production submodel structure

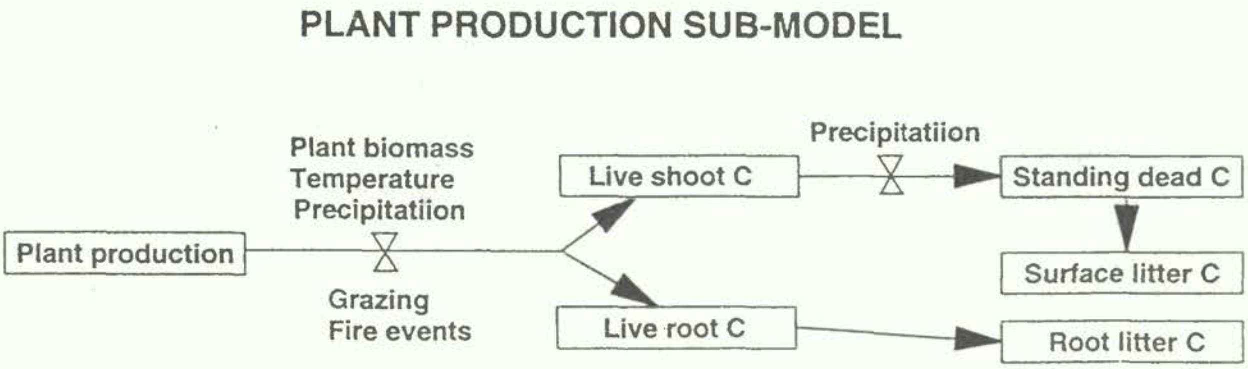
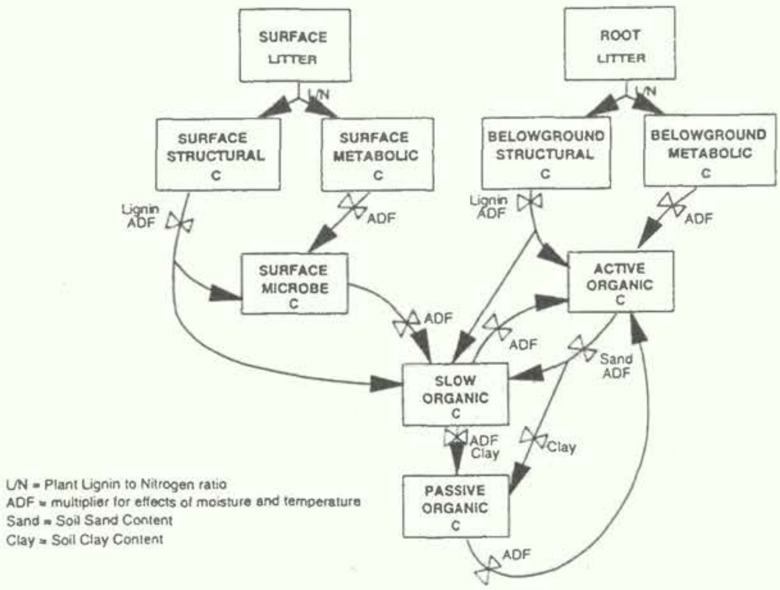



Figure 2. CENTURY model soil organic matter submodel




The input variables are available for most natural and agricultural ecosystems and can generally be estimated from existing literature.

### Soil organic matter submodel

Soil C storage is one of the largest pools of C in an ecosystem. Soil C cannot be considered a homogeneous mixture of organic material. Soil C must be viewed as a mixture of labile and more resistant material in order to fully understand the dynamics of soil C relative to climate and land management changes. The short-term fluxes of C from soil is derived primarily from changes in the labile SOM pools. The size of this compartment is relatively small but accounts for more than 2/3 of the soil C respiration under 'steady-state' conditions.

The majority of the SOM is stored in more recalcitrant material, and these soil C pools also have different degrees of reactivity, and have been defined by Parton et al. (1987) as 'slow' and a 'passive' SOM pools. The slow and passive pools constitute approximately 90% of the total soil C, with an estimated 3:2 ratio between the slow and passive pools.

Soil texture influences the relative proportions of slow and passive soil C, clayey soils tend to accumulate more passive soil C than do sandy soils. When a change in land management occurs, the slow pool often is affected the most. Changes in the slow pool will not only affect the C flux from an ecosystem, but will also change soil fertility levels. So that changes in long-term productivity may occur.

### Land use change

Land use practices can dramatically alter C storage and fluxes. Conversion of grasslands or forests to cropland can result in a rapid decline in C stores, Up to 50% of the soil C and the woody biomass of the forest can be lost due to cropland conversion (Haas et al. 1957, Cole et al. 1989, 1990). Land under cropland, forest plantation, or rangeland management can vary greatly in C storage and fluxes. Forests and rangelands under good management and proper rotations may effectively be neutral relative to C fluxes when averaged over a large area. Cropland areas are vulnerable to large C losses due to removal of vegetation and soil disturbances on an annual basis. However, not all cropping management practices have this impact. Recent no-till or reduced till practices have been developed to lessen the impact on the soil C losses from croplands. The historical land management usage greatly determines the current C storage and potential fluxes from these managed systems.

### Impacts on human resources

In developing countries, high population growth is causing people to expand into areas with marginal soils and inappropriate climates for certain land use practices, which is placing unprecedented pressures on their natural environments (WCED 1987, Parry et al. 1988). Certain social systems are better adapted to changing situations due to adaptive strategies for coping with variable climatic patterns as are, for instance, certain pastoral systems in East Africa (Ellis and Swift 1988).

However, many third world people are living in economies somewhere between traditional-subsistence and modern-industrialized levels, where they have neither inherent resilience to environmental change nor the resistance provided by mature social and economic government programs (Migongo-Bake, this volume; Swallow, this volume).

The process of modernization for many of these "transitional" societies has resulted in less economic self-sufficiency due to greater dependency on regional and international markets over which they have no control (Fleuret and Fleuret 1980, Lamb 1988). Brought about by global change, ecological perturbations in these transitional societies can lead to political strife, famine, and other social disturbances (Redclift 1987, Whitehead 1989).

The ability of humans to adapt to these rapid and unprecedented changes will be tested during the coming decades. The coupling of research and policy communities for the purpose of developing mechanisms to adapt to these future changes which we have brought upon ourselves urgently needs to be established (Baron and Galvin 1990; Peacock et al this volume). The soil organic matter (SOM) submodel simulates the dynamics of C, N, P, and S in the organic and inorganic parts of the soil system. The flow diagram (Figure 2) for the soil organic C model shows that soil organic C is divided up into three major components which include active (microbe), slow, and passive soil C.

Active SOM includes live soil microbes plus microbial products (total active pool is ~2 to 3 times the live microbial biomass level), the slow pool includes resistant plant material (lignin derived material) and soil stabilized plant and microbial material, while the passive material is very resistant to decomposition, and includes physically and chemically stabilized SOM.

The model also includes a surface microbial pool which is associated with decomposing surface litter. Flows of carbon between these pools are controlled by decomposition rate and microbial respiration loss parameters, both of which may be a function of soil texture. The turnover time of these pools varies with the soil abiotic decomposition parameter (function of monthly precipitation and temperature).

The abiotic decomposition parameter is calculated by multiplying the soil temperature function times the soil moisture function. Across the North American region, decomposition rates are the highest in the southeast and lowest in the northwest.

Average monthly soil temperature near the soil surface is the input for the temperature function and the ratio of stored soil water (0-30 cm depth) plus current precipitation to potential evapotranspiration is the input for the moisture function. Typical turnover times for a grassland site are 2, 40, and 2000 years; respectively for the active, slow, and passive pools.

The inputs of C to the soil are from plant residue (shoot and roots) and are partitioned into structural and metabolic plant components as a function of the lignin (L) to nitrogen (N) ratio of the dead plant material. High L:N ratios result in more structural material. Metabolic material has a much higher decomposition rate and structural material contains all of the lignin compounds.

The decomposition rate of the structural material is a function of the fraction of the structural material that is lignin. Decomposition of the metabolic material and the non-lignin fraction of the structural pool material is transferred to the active SOM pool. The lignin fraction of the plant material does not go through the microbes and is assumed to go directly to the slow C pool as the structural plant material decomposes.

The model represents the decomposition of aboveground and below ground plant residue separately and includes a surface live microbial pool that is in close association with the surface litter. SOM1 (1) is the surface microbes and SOM2 (2) is the active SOM pool.

The surface microbial pool turnover rate is independent of soil texture, and flows material directly into the slow SOM pool. The soil texture influences the turnover rate of the active soil SOM (higher rates for sandy soils) and the efficiency of stabilizing active SOM into slow SOM (higher stabilization rates for clay soils). The formation of passive SOM is a function of the clay content (higher for high clay soils) and is primarily controlled by the stabilization of active SOM into stable clay associated microaggregates. Some passive SOM is also created by the decomposition of slow SOM (10 to 30%) and includes a similar effect of clay on passive SOM formation.

We also include the flow of soluble organic material out of the top 30 cm of the soil. Leaching of organic matter is a function of the decay rate for active SOM, and the clay content of the soil (more loss for clay soils) and only occurs if there is drainage of water below the 30 cm soil depth. Anaerobic conditions (high soil water content) cause decomposition to decrease and the soil drainage factor allows a soil to have differing degrees of wetness. A detailed description of the structure of the model and the way in which the model parameters were estimated are found in Parton et al. (1987, 1993).

The CENTURY model includes a simplified water budget model which calculates monthly evaporation and transpiration water loss, water content of the soil layers, snow water content and saturated flow of water between soil layers. If the average air temperature is less than 0øC monthly precipitation occurs as snow. Sublimation and evaporation of water from the snow pack occurs at a rate equal to the potential evapotranspiration rate. Snow melt occurs if the average air temperature is greater than $0^{0}$C. Snow melt is a linear function of the average air temperature.

The potential evapotranspiration rate (PET) is calculated as a function of the average monthly maximum and minimum air temperature and is modified by a user specified multiplier . Bare soil water loss is a function of standing dead and litter biomass (lower for high biomass levels), rainfall and PET.

Interception water loss is a function of aboveground biomass (increases with biomass level), rainfall and PET. Transpiration water loss is function of the live leaf biomass (exponential function of leaf biomass), rainfall and PET. Interception of water and bare soil water losses are calculated as fractions of the monthly precipitation and are subtracted from the total monthly precipitation, with the remainder of the water added to the soil.

The field capacity and wilting point for the different soil layers is calculated as a function of the bulk density, soil texture, and organic matter content. Water going below the profile can be lost as storm flow or leached into the subsoil where it can accumulate or move into the stream flow at a specified rate.

Average monthly soil temperature near the soil surface is used to calculate maximum soil temperature as a function of the maximum air temperature and the canopy biomass (lower for high biomass) while the minimum soil temperature is a function of the minimum air temperature and canopy biomass (higher for higher biomass). The actual soil temperature used for decomposition and plant growth rate functions is the average of the minimum and maximum soil temperatures.

Leaching of labile mineral N, ($NO_3$ + $NH_4$), P and S pools occurs when there is saturated water flow between soil layers. The fraction of the mineral pool that flows from the upper layer to the lower layer is a function of the sand content and the amount of water that flows between layers. Monthly watershed loss of $H_2O$, inorganic and organic N, P and S are simulated by the model.

The N submodel has the same structure as the soil C model. The C:N ratio of the structural pool (150) remains fixed and the N content of the metabolic pool varies as function of N content of the incoming plant residue. The C:N ratio of newly formed surface microbial biomass is a function of N content of the material being decomposed (increases for low N content).

The soil C:N ratios vary as a function of size of the mineral N pool for the newly formed active (3-15), slow (12-20), and passive (7-10) pools. The C:N ratio for slow material formed from surface microbial biomass is a function of

C:N ratio of the surface microbes. The C:N ratio for the passive pool is low for clay soils and high for silty soils.

The N flows follow the C flows and are equal to the product of the carbon flow and the N:C ratio of the state variable that receives the carbon. The N:C ratios of the state variables receiving the flow of material (newly formed SOM) are a function of the mineral N pool. The N attached to carbon lost in respiration (30% to 80% of the carbon flow is respired) is assumed to be mineralized. Given the C:N ratio of the state variables and the microbial respiration loss for each flow, decomposition of metabolic residue, active, slow, and passive pools generally result in net mineralization of N, while decomposition of structural residue immobilizes N.

The model also uses simple equations to represent N inputs due to atmospheric deposition and soil and plant N fixation and also includes fertilizer N inputs and N inputs through plant residue additions. The model has the option of calculating soil N fixation rates as a function of the mineral N to labile P ratio (high fixation with lower ratios).

The losses of N due to leaching of $NO_3$; losses accumulate in the layer below the last soil layer or lost in the stream flow, leaching of organic N compounds, gaseous losses of N compounds, crop removal, and erosion are also represented. N losses due to $NO_3$ leaching are calculated as a function of the water flow between soil layers (higher for larger flows) and the fraction of $NO_3$ that leaches out of the layer (see water flow model description).

In order to investigate the ecosystem response to elevated $CO_2$, the plant production parameters were modified for both $C_3$ and $C_4$-type grasslands under a doubled atmospheric $CO_2$ by changing production relative to PET and to nitrogen use efficiency (NUE). The $C_3$ specific modifications related to $CO_2$ induced changes of the assimilation efficiency of the rubisco C uptake pathway was not simulated.

For the current analysis we uniformly implemented the impact of increased atmospheric $CO_2$ concentrations on all grasslands. The magnitude of the effect is to cause a maximum of 20% increase in plant production with a doubling of atmospheric $CO_2$ concentration. The effect of modified NUE and PET on plant production is a simple linear effect on these processes. In addition, changes in NUE affect litter quality so that higher NUE results in a slower decomposition of this material under a given temperature and moisture regime

## Climatology

Climate factors affect a number of key processes of an ecosystem that determine C fluxes and storage, such as decomposition rates, plant production, and respiration rates of soil and vegetation. Rainfall amounts tend to control the amount of plant production that occurs, whereas, temperature determines decomposition rates. Although, both rainfall and temperature interact in both processes to a certain degree. Climate also plays an important role in determining the frequency and the intensity of disturbances. Lightning, wind storms, and pest outbreaks are often associated with changes in climate factors.

Changes in climate variability may also have adverse impacts on ecosystem dynamics and on C fluxes from an ecosystem. Ecosystems are often sensitive to changes in extreme events, so that changes in low or high temperature extremes, or in drought occurrence can rapidly change the structure of an ecosystem, especially when these modifications in extremes coincide with a disturbance event that resets the ecosystem to a new state. In these instances, large fluxes of C and long-term changes in C storage may result.

## Model studies of ecosystem response

Ecosystem sensitivity to the temporal and regional resolution of climate change (changes in annual mean climate vs. seasonally varying changes) was evaluated by driving a regional ecosystem model with general circulation model (GCM) climate output. Using a grassland model developed for the North American Central Grasslands region (CENTURY, Parton et al. 1987, 1993), simulations of aboveground net primary productivity (NPP) and soil organic carbon (SOC) for 72 sites across the region were made. Simulated losses in soil fertility and soil organic matter indicated that 800-2000 gC/m$^2$ were lost from initial plow-out to present, a period of approximately 100y. The simulated losses from climate change over a 50-y period are relatively small compared to those resulting from cultivation.

In similar studies, the interaction of climatic effects on biogeochemical processes in grassland ecosystems indicated that despite increases in aboveground NPP decreases in soil carbon would counterbalance the apparent carbon gains (Schimel et al. 1990). Schimel et al. (1990) further observed that enhanced increase aboveground NPP was a transient phenomenon.

## Factors affecting carbon cycling in rangelands

We selected nine case studies ranging from cool dry regions in the semi-arid and arid areas of Asia to tropical regions in Africa. The mean annual rainfall of the sites ranged from 250mm to 800mm, and the mean annual temperatures ranged from -3 to 30 °C (Table 1). Soil texture and bulk density varied across these sites and represent an adequate range of soil properties that encompass the arid and semi-arid ecosystems around the world.

Table 1. Site Characteristics of Case Study Areas.

| Site | Latitude/ Longitude | Mean Annual Temperature (°C) | Mean Annual Rainfall (mm/yr) | Average Total NPP (gC/m$^2$/y) | Soil C (20 cm) (gC/m$^2$) |
|---|---|---|---|---|---|
| Tumentsogt Mongolia | 46N 113E | 1.4 | 271 | 102 | 4460 |
| Kalgoorlie Australia | 30.85S 121.5E | 18.3 | 276 | 130 | 4700 |
| Tuva Russia | 52.0N 94.0E | -3.5 | 292 | 115 | 3700 |
| Warra Australia | 26.4S 146.3E | 20.6 | 464 | 175 | 5000 |
| Niamey Niger | 13.5N 22E | 29.0 | 560 | 290 | 2410 |
| Alice Springs Australia | 23.42S 133.53E | 20.7 | 322 | 54.2 | 3003 |
| Xilingele China | 44N 117E | -0.05 | 358 | 58.0 | 4162 |
| Marondera Zimbabwe | 18S 31E | 17.95 | 803 | 257 | 5362 |
| Sheno Ethiopia | 9.2N 39.18E | 12.4 | 871 | 259 | 4250 |
| Yavelo Ethiopia | 6.0N 39E | 19.8 | 656 | 70 | 4000 |

In order to establish a baseline for simulations across these different sites, we used the available 25 years of meteorological data for each site. When data were not available from a researcher at the site, we relied on the "World Weather Disc" (Weather Disk Associates, Inc., National Climate Data Center) for a site. The 25 years of weather data were used to establish an "equilibrium" condition for each site based on a 2500 year CENTURY simulation. Grazing levels were established for each site that represent a light to moderate removal rate. In the Ethiopian case study we simulated a baseline "tractional" cropping system that implemented a 10 to 15 year fallow between 5 years of cropping of barley, with manure return during the cropping years. We then simulated the "current" cropping system which reduces the fallow period to 5 years and reduces the amount of organic matter return.

Our main focus is on analysing the effects of different grazing intensities at the case study sites. In the experimental simulations we simulated low and high grazing intensities at each site for 25 years during wet and dry years derived from each site. In the subsequent 25 years we continued the grazing intensity or implemented the alternate grazing pattern. There are thus four simulated grazing combinations - 50 years of high (H-H) or low (L-L) grazing intensity, and 25 year combinations (L-H and H-L); each of these combinations was simulated for a repeated sequence of three wet years and also for three dry years for each site, to help to separate the effects of climate compared to grazing. For the Alice Springs site, we also simulated the impact of erosion losses of whole soil from the ecosystem, assuming approximately 1 kg/m$^2$ of the soil was eroded annually from the sites in a single erosion event each year.

## RESULTS

### Baseline Dynamics

The simulated average aboveground plant productivity ranged from 60 gC/m$^2$/y for the Tumentsogt site in Mongolia to a high of 250 gC/m$^2$/y for the Maderona site in Zimbabwe (Figure 3). The range of plant productivity and soil C for a selection of these nine sites indicates the inherent climatic variability of the sites (Table 2), with the greatest range at the Alice Springs site reflecting its large rainfall variability. The simulated plant productivity over a 25 year period displayed a 200-800% range in productivity. The soil C levels ranged over a much smaller range of values, varying by 120-150 gC/m$^2$ during the 25 years, representing a 2-3% change in the mass of soil C in the top 20 cm. These changes reflect variation in organic matter inputs into the labile pools of SOM resulting from high and low productivity years.

Figure 3 Plant production relative to mean annual rainfall for the different case study sites.

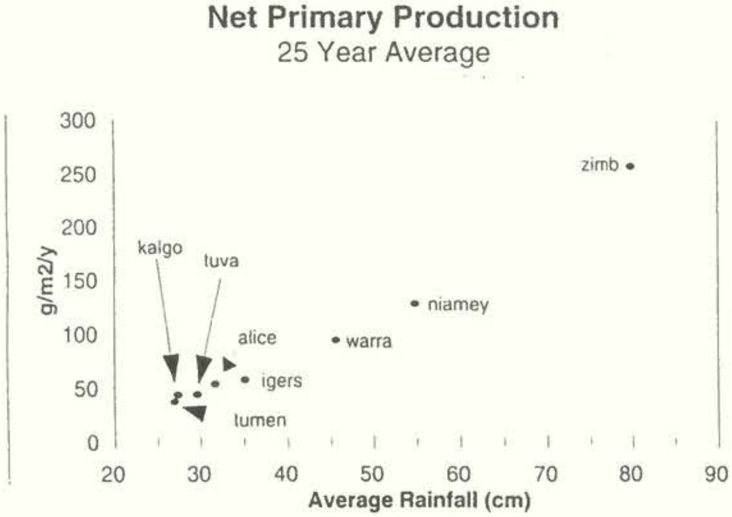


Table 2. Range of C dynamics.

| SITE | Soil C $gCm^{-2}$ | Aboveground NPP $gCm^{-2}y^{-1}$ |
|---|---|---|
| Tumentsgot, Mongolia | 4470 - 4590 | 30 - 70 |
| Alice Springs, Australia | 2930 - 3060 | 20 - 170 |
| Marondera, Zimbabwe | 5300 - 5440 | 160 - 320 |
| Kalgoorlie, Australia | 4085 - 4163 | 14 - 93 |
| Tuva, Russia | 3650 - 3750 | 20 - 70 |
| Warra, Australia | 5096 - 5210 | 26 - 165 |
| Niamey, Niger | 2595 - 2660 | 57 - 198 |
| Xilingele, China | 4100 - 4200 | 23 - 96 |
| Sheno, Ethiopia | 4000 - 4500 | 220 - 300 |
| Yavelo, Ethiopia | | 40 - 100 |

Grazing Intensity Assessment

The results of simulating changes from low to high or high to low grazing intensities indicate that the sites have a range of vulnerability to heavy utilization (Table 3). Fig.4 shows an example of how the soil C in the Mongolian site changes under the different grazing regimes in wet or dry year climate sequences. In all sites recovery capacity in a run of dry years was less than in wet years. This is most apparent in the warmer sites (e.g., Australian and African sites). The recovery of soil
C during wet years ranges from 400-600 $gC/m^2$ under low grazing intensity for 25 years after high grazing intensity (Table 3). The time taken to achieve most of the C gains with lighter grazing varied greatly between the sites, with the quickest recovery occurring in the dry simulations of Zimbabwe, which recovered most of its soil C in the first 5 years of improved grazing management. This result is due in part by the initial low levels of soil C of this site. Soil C recovery had not asymptoted on several sites even after 25 years. This indicates that the fullest possible sequestration of soil C under these conditions would take a long time.

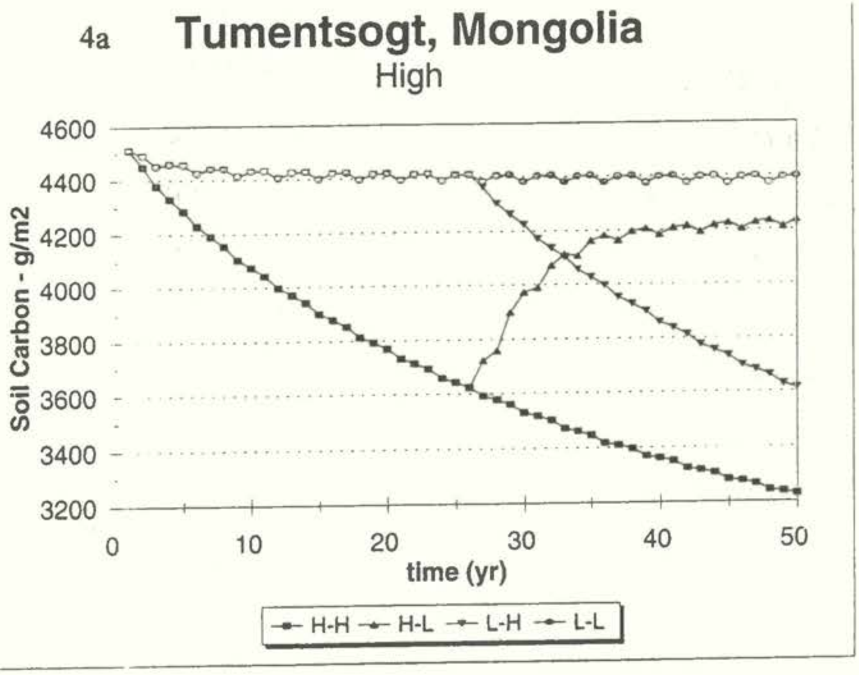
The simulation of low-to-high grazing intensity indicates that all of the sites have a sensitivity to increased grazing intensity. This implies that any sequestration gains could be lost by a subsequent increase in grazing pressure, thus indicating that any gains are always at risk of changed management.

Erosion Effects

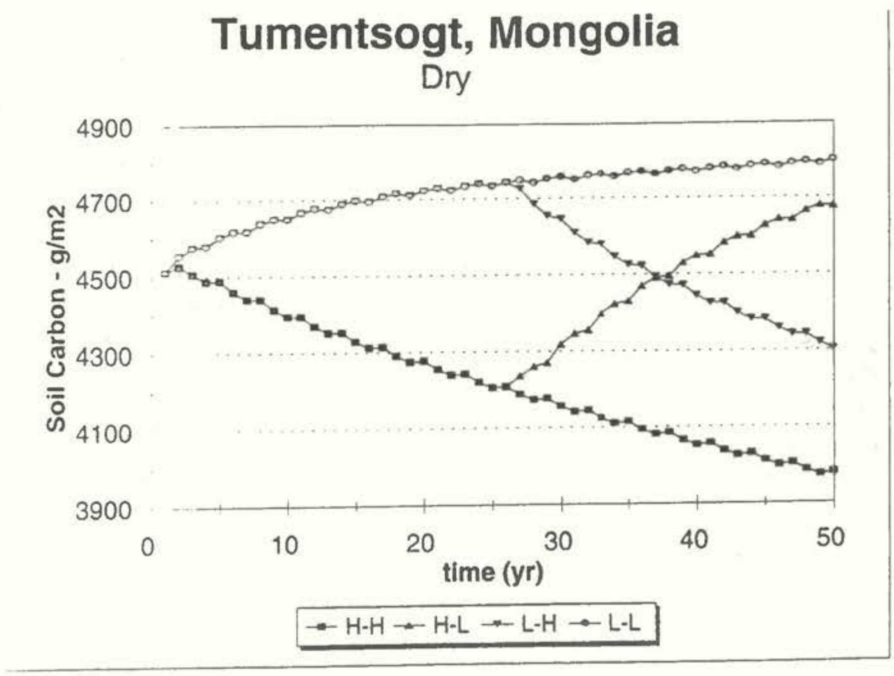
The results of the additional set of simulations of the impact of erosion losses following high grazing levels at the Alice Springs site are shown in Fig. 5.

Fig 4 Grazing impacts on soil organic matter levels in (a) high rainfall periods and (b) during a dry period at Tumentsogt, Mongolia

4a




4b



Soil erosion is assumed to remove whole soil particles and hence soil C from all fractions of SOM. The simulations indicate that erosion losses greatly reduce the capacity of an ecosystem to recover predisturbance levels of soil C.

Table 3. Soil C Recovery

| Site | Soil C ($gCm^{-2}$) | Time to 80% - 90% Recovery |
|---|---|---|
| Tumentsgot, Mongolia | | |
| Dry | 500 | >25y |
| Wet | 600 | ~10y |
| Alice Springs, Australia | | |
| Dry | 200 | ~15y |
| Wet | 600 | >25y |
| Marondera, Zimbabwe | | |
| Dry | 170 | ~5y |
| Wet | 400 | >25y |
| Kalgoorlie, Australia | | |
| Dry (med.) | 16.1 | 5y |
| Wet | 216.3 | >25y |
| Tuva, Russia | | |
| Dry | 449.1 | >25y |
| Wet | 1,106 | >25y |
| Warra, Australia | | |
| Dry | 248.7 | >25y |
| Wet | 302.6 | 10y,13y |
| Niamey, Niger | | |
| Dry (med.) | (915) | (>25y) |
| Wet | 924 | >25y |
| Xilingele, China | | |
| Dry | 10.6 | <1y |
| Wet | 431.9 | 20y,25y |

The recovery of soil C with erosion included was about 10% of that in simulations without erosion. This occurs because the whole soil losses remove passive and slow soil C components, which accumulate over decades to centuries. These components take much longer to recover than labile C.

Cropping and Organic Matter Inputs

The case study from the Ethiopian highlands illustrates the impact of cropping on soil C storage and modification of management practices to improve C storage. The important differences between the traditional and the current cropping system are that the fallow period is reduced to 5 years from an initial fallow period of 10 years, and that manure is no longer reapplied to the cropland, but is used for fuelwood. The simulations indicates that soil C declines under current cropping methods, dropping from 4600 $gC/m^2$ to less than 3500 $gC/m^2$. Returning to traditional cropping systems reverses the soil C losses, but the long fallow period is not feasible given the increased human pressures currently that exists. Further analysis indicates that soil C recovery is essentially dependent on manure inputs (Figure 6). Decreased grazing intensity improves soil C, but not as much as manure additions.

CONCLUSIONS FROM THE FIELD STUDIES

Current rates of degradation and recovery in the semi-arid and arid ecosystems are not well defined. What ecosystems are undergoing these land use and climate changes that have led to degradation of these systems? Knowledge of the human system in connection with the potential changes in the ecosystem properties need to be closely coupled with the factors leading to degradation and leading to recovery of these systems.

Natural variability in climate often mask the impact of land use changes relative to increasing or decreasing soil C in these ecosystems.

Soil C dynamics in strongly controlled by C inputs and losses from the ecosystems, and understanding the biological and human factors modifying organic matter inputs and losses is important to determine.

Soil erosion limits the capacity of the ecosystem to return to former state of soil C due to losses of passive soil C. The recovery of passive soil C takes hundred to thousand years of accumulation.

The fate of soil C from erosion is not well documented and how this redistributed soil C may enhance soil fertility within the landscape is an important consideration in computing net losses of C from these eroded ecosystems.

## Regional carbon budgets

In analysis and the computation of regional C budgets, a number of factors determine the C storage and fluxes. These include the state of the vegetation, the characteristic soil properties, the inherent climate regime, and the land use practices implemented in the region. The current C level is not only determined by the current state of these factors but is also dependent on the past conditions of these factors.

Components of the natural and semi-natural systems include forests, grasslands, croplands and range ecosystems. The dynamics of C in these ecosystems are greatly influenced by the structure and the stage of vegetation development of an ecosystem. Natural ecosystems may have large stores of C in soil or vegetative compartments (e.g., standing timber or dead wood). The net flux of C from natural ecosystems at steady-state conditions to the atmosphere are essentially neutral. However, when these ecosystems are disturbed through natural disturbances (e.g., fire, pest outbreaks, or storm damage) or converted to croplands or a semi-managed state by modifying land use practices large fluxes of C to the atmosphere may result. Release of C may results from soil compartments or from vegetative reservoirs.

Figure 5. Erosion effects on soil C storage interacting with different grazing levels at the Alice Springs, Australia site during a wet rainfall period.

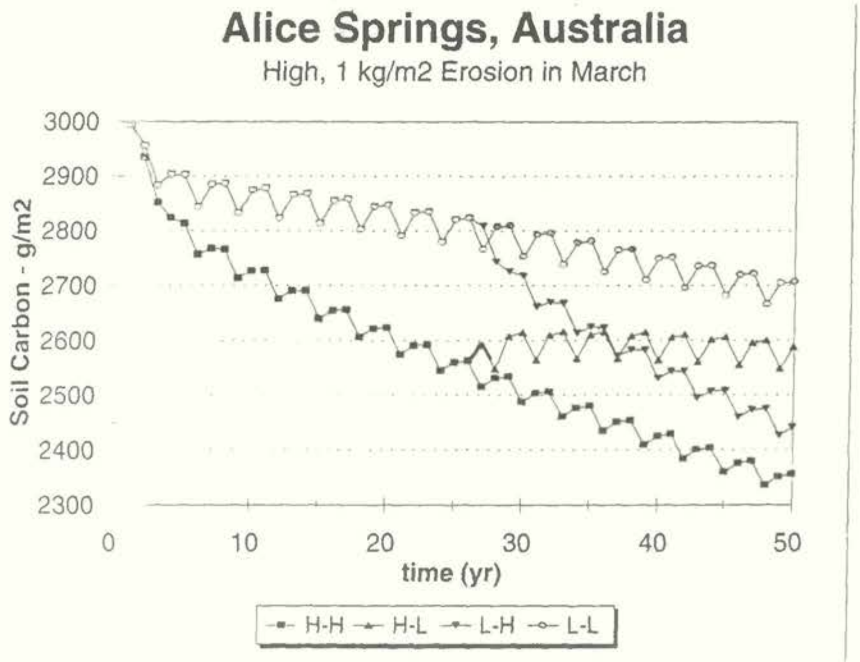
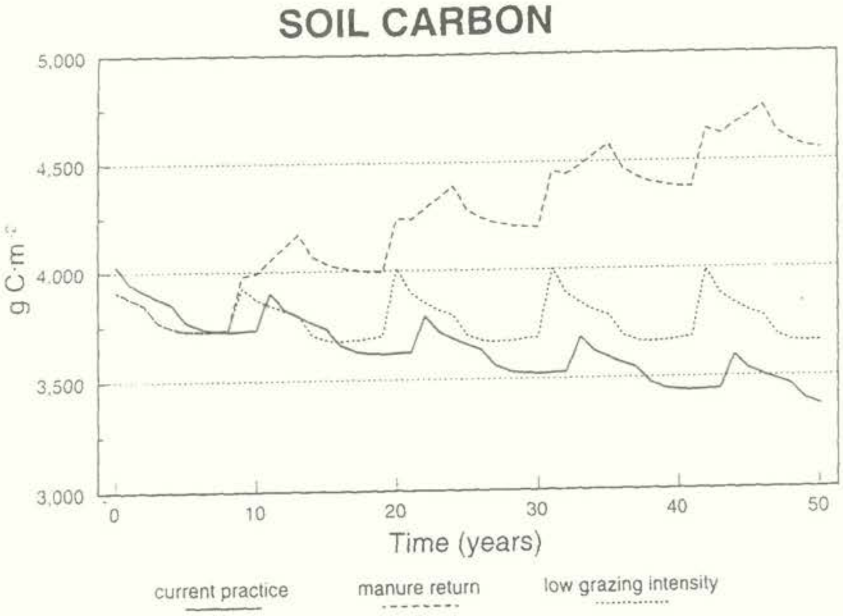



Figure 6 Organic matter inputs and grazing level effects on cropping systems of the Ethiopian highlands



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# Primary productivity in degraded drylands of Eurasia

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Synopsis
Climatic changes in the drylands of Eurasia over the past 100 years are analysed. Land use change is also analysed with special reference to changes in soil organic matter status (loss of humus). There has been complete anthropogenic transformation of 60-80% of the steppe region. The consequences of such a radical transformation are discussed in terms of desertification, loss of productivity and loss of biodiversity. Discussion focuses on the implications for global climate change.

Key points

1. Drylands cover about 30% of the land area in Europe and Asia. Deserts, semi-deserts and steppes characterise the Eurasian drylands.. These include steppic and semi-desert regions on both plains and mountains; and deserts. The latitudinal spread is quite wide from subtropical to subboreal. They are located on the drier margins of the agricultural land.

2. Productivity of the drylands has been monitored and is perceived to be falling. The current rate of accumulation of soil organic matter in drylands of Eurasia depends on the intensity of land use for any given climate zone. The below-ground component of production is high and comprises the main reserve of phytomass. In the Siberian steppes up to 95% of the phytomass is below ground.

3. There have been major changes in the drylands of northern Eurasia in response to the greater intensity of utilisation. These drylands had remained in a stable and largely unused state for many centuries. They were natural rangelands subjected to intermittent grazing by nomadic pastoralists and by the numerous wild ungulates. Industrialisation and modernisation of agriculture began there about 200-300 years ago but intensified over the past 70 years.

4. Many rangelands were converted to croplands . By the early part of the twentieth century steppes had all but disappeared from eastern Europe. Cultivated land now makes up 50-70% of the total. Productivity has declined dramatically as soil organic matter has been depleted. Soil degradation is common with soil erosion a critical problem in many areas. Areas with moderate to severe erosion make up 60-80% of all agricultural land in the region.

5. Because of their location immediately adjacent to the agricultural areas and, in some cases, urban and industrial land, the surviving rangelands serve as refugia for biota. Overstocking and prolonged overgrazing by domestic livestock are putting pressure on these remaining drylands. Productivity has declined and land degradation has begun. Soil organic matter levels are dangerously low.

**Key words** land degradation, biodiversity, soil erosion, agriculture, biota, refugia, phytomass, climate change, pastoralism, land use change

## Introduction

The drylands of Eurasia have been classified into various steppes, semi-desert and desert ecosystems (Table 1). They cover a vast area which has a wide latitudinal spread from subtropical to subboreal. About 30% of the plains and 10% of the mountains are classified as drylands (Babaev et al 1986).
Productivity of the various ecosystems has been assessed from an analysis of

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data collected from over 600 sites throughout the region. There is a good relationship between mean annual precipitation and productivity at a site with above-ground biomass ranging from 500 kg/ha on subboreal deserts in Central Asia to over 2500 kg/ha on riparian areas in the Central Russian Plain. Below-ground biomass was, in each case, considerably higher (Chibilev 1992; Tishkov 1994). The proportion of below-ground biomass was inversely proportional to mean annual rainfall. The tendency toward a continental climate can also influence the proportion of below-ground biomass. In the Siberian steppes up to 95% of the total biomass is below ground (Bazilevich and Tishkov 1986)

Dead material accumulates in drylands at a rate that depends on land use and climate. In eastern Siberia (Yakutiya, Zabaikal'e) and in Mongolia the processes of destruction of organic material are very slow. Steppic ecosystems there can accumulate 20-40 t/ha of dead material. In desert ecosystems the rates of accumulation are much lower and are influenced by biotic destruction (grazing, haymaking and fuel gathering) and transport by wind.

Table 1 A schematic classification of dryland ecosystems in Eurasia

| | **Major vegetation type** | | |
|---|---|---|---|
| | ***Steppe*** | ***Semidesert*** | ***Desert*** |
| **Plains -** subboreal | | | |
| | -meadow | -Typical | -zonal(clay) |
| | -moderately dry | -sandy | -sandy |
| | typical dry | -salt | -salt |
| | dry and desertified | | |
| - subtropical | -meadow | -typical | zonal(clay) |
| | -typical | -sandy | -sandy |
| | -dry and desertified | -salt | -salt |
| **Mountains** -subboreal | -meadow | # | # |
| | -moderate and typical | n.a. | n.a. |
| | -dry and desertified | n.a. | n.a. |
| - subtropical | -meadow | # | # |
| | -typical | | |
| | -dry and desertified | | |

1 Deserts of the plains are further subdivided into subboreal(northern) subboreal (typical), subtropical (northern) and subtropical (southern)

\# Represented here

n.a. Not applicable

## Changes in the drylands of Northern Eurasia

For many centuries the steppic regions of northern Eurasia were undeveloped. They were called the "wild" region and were the domain of wildlife and nomadic herders. They represented the border between the settled population of Russia and the nomads from the south.

Later, as industrialisation was promoted and mechanised agriculture expanded, large areas were plowed up. The first targets were meadow lands but as time went by the drier steppes were also converted to cropland and pasturelands. There were fundamental changes in vegetation type and cover. Animal populations diminished and soil degradation became widespread.. Plowed up areas currently represent 50-70% of all land (Table 2) but the soils are badly degraded with soil loss in the range of 15-30 tons/ha (Karavaeva et al 1989). Productivity of the soil has declined as humus content was reduced.

Table 2 Ecological conditions, soil type and area of plowed land in steppes of Eurasia and European Russia

| **Region** | **Agro-climatological data** | | **Area of Steppic soil** | |
|---|---|---|---|---|
| | Sum of T>10°C | Precipitation | Total (ha) | % |
| All zone in Eurasia | 1500-3500 | 200-600 mm | 118.9 mill | 57 |
| European Russia incl.N. Caucasus | 3000-3500 | 400-600 | 14.8 | 62 |
| Chernozem Centre | 2600-3200 | 350-550 | 5.5 | 83 |
| Middle Volga | 2200-2800 | 300-400 | 12.9 | 50 |
| Southern Urals | 2000-2900 | 300-400 | 13.5 | 60 |

## Climatic change and land degradation

An analysis of the meteorological records throughout Eurasia over the past 100 years reveals apparent changes in the climate. The warming trend seems to begin about the end of the nineteenth century and peaked about the 1940s. Evidence is that in the period 1940 to 1970, there has been a slight cooling (temperatures have fallen by 0.2° C.). However on balance the calculated rate of warming is set at 0.03 °C per decade over the period 1885 to 1985. There are regional differences with values ranging from 0.006 °C/yr for the meadow steppes and agricultural regions in the European section of Russia to 0.008 °C/yr in the meadow-moderate to dry steppes of central Chernozem region of Russia

Two broad belts have been identified where there has been an apparent warming. One of these belts covers the steppic zone - 60-80% of which are now plowed (Table 3/. Anthropogenic transformation of the steppes has led to desertification on the southern margins of the steppes, degradation of forest "islands" in the steppes, loss of productivity and loss of biodiversity (Table 4).

Table 3. Changes in humus reserves and concentration in chernozem soils from 1881 to 1991 in regions of European Russia #

| **Regions** | **Reserves and concentrations** | | | | **Loss of humus** | |
|---|---|---|---|---|---|---|
| | 1881 | | 1981 | | from 1881 to 1981 | |
| | t/ha | % | t/ha | % | t/ha | % |
| N.Caucasus | 221-315 | 7-10 | 150-263 | 4-7 | 67-81 | 20-34 |
| Chernozem | 300-330 | 10-13 | 210-330 | 7- | 90 | 23-34 |
| Centre | 10 | | | | | |
| Middle Volga | 390-480 | 13-16 | 120-210 | 4-7 | 270 | 56-69 |
| S.Urals | 270-330 | 9-11 | 180-240 | 6-8 | 90 | 27-33 |

# 0-30 cm layer

Table 4 Potential climate changes in various dryland regions of European Russia

| **Regions** | **Climate changes (projected)** | **Potential change in productivity** |
|---|---|---|
| Steppes of middle Volga | Reduced length of growing season. Reduced precipitation, warming | Decrease in standing biomass and annual production, change of community structure |
| Steppes of center of Russian Plain | Reduced length of growing season, warming, more arid conditions | increase in below-ground component of biomass, change in community structure |
| Dry-steppes.semi-deserts and N subboreal deserts of SE European Russia | Longer growing season, less rainfall, warmer | Reduced standing biomass and lower annual production, faster rate of rangeland degradation |

## Conclusions

The vast Eurasian region is an analogue for the comparable latitudinal belt in western Europe and North America. Data are now becoming available from the former USSR which can add to the results already published from North America to give a better appreciation of the impacts of anthropogenic transformation of grasslands and steppes on global climate change. As the net primary productivity of these regions declines, through loss of soil organic matter and more intensive utilization, the prospects for carbon sequestration will diminish.

These vast areas will be a net emitter of carbon dioxide rather than a sink. However, if the productivity can be restored the potential is there to reverse the trend and add this important region to the list of carbon sinks. The challenge for scientists is to devise methods to increase output of food crops and forage in the face of increasing populations of people and livestock . Preservation of biodiversity is also a priority.

## Acknowledgments

The author is grateful to his colleagues within the Russian Academy for access to their data and to Dr. V. Squires for editing the manuscript. UNEP provided travel funds to attend the Nairobi Workshop.

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# The role of woody plants in carbon and nitrogen budgets of arid lands

**R. J. Scholes**

Synopsis

This paper presents the hypothesis that water limitation in arid ecosystems operates primarily via the nitrogen cycle. It addresses four key questions with respect to woody plants (trees and shrubs) in drylands. Does the inclusion of woody planted in the ecosystem increase net primary productivity? Do woody plants confer greater interannual stability of primary production? Do woody plants confer greater resilience to the ecosystem? Do woody plants offer a pathway for recovery from degradation? It concludes that trees can form part of a rehabilitation strategy for degraded arid lands.

Key Points

1. In many parts of the world the increasing density of 'woody weeds' has been identified as a symptom of degradation. Paradoxically, in other places, the loss of trees is an indicator of degradation. In some circles there is an almost mystical faith in the power of trees, and other woody plants, to restore degraded landscapes. In others ecosystems, woody plants are seen as competitors with grasses for scarce resources.

2. Woody plants differ in many respects from non-trees in the way they obtain and use nitrogen. A consequence is that the inclusion of woody plants in arid, nitrogen-limited systems increases net primary production. The longevity of woody plants, and their capacity to carry resources over between seasons, reduces interannual variability on primary production and confers resilience to certain types of perturbation.

3. The presence of trees in the ecosystem helps to reduce the rate of loss of nutrients from the system. Particularly when it is subject to perturbation by drought, grazing or cultivation. Holding a significant portion of the ecosystem nitrogen pool in a medium-term turnover pool such as tree biomass buffers the system against short-term losses. The stability of the tree cover affords protection to the soil surface when the grass cover is not present. This reduces soil erosion, which is frequently the major avenue of nutrient loss. The deep roots and extended growing season of the trees offers an opportunity to capture nitrate which may have leached below the grass root zone. The rapid uptake of inorganic nitrogen reduces the chance of denitrification. Since only a small fraction of the woody plant biomass is consumed by herbivores, only part of the plant nitrogen is prone to volatilisation from urine and faeces. The nitrogen in trunks and leaves (especially in larger trees) may be largely protected from pyrodenitrification during low intensity dryland fires.

4. The inclusion of woody plants in nitrogen-limited ecosystems, which includes most dryland ecosystems, can increase net primary production. There may be an upper limit to the tree cover for optimal net primary production (NPP) and there are climate limits below which the tree life strategy is unviable. The presence of trees also

reduces the interannual variability in NPP. This is due to a carryover of resources between years (either within the woody plants, or in the ecosystem pools inaccessible to other plants) and to the inertia inherent in long-lived species. Woody plants increase the resilience of dryland ecosystems to drought and episodes of intense grazing, Due mostly to their partial decoupling from recent rainfall and their inaccessibility or unpalatability to grazers/browers respectively.

5. Woody plants have a role in the rehabilitation of drylands, as part of an overall strategy including grasses and amelioration of degradation-causing processes. The re-establishment of woody in landscapes from which they have disappeared requires a significant input of effort. So too does the reduction of tree density in landscapes where they have become overabundant. Trees are not a panacea for degraded ecosystem, but they can make a valuable contribution in some cases. In particular, where it is necessary to reduce the rate of nutrient loss from the ecosystem due to leaching or erosion, trees in combination with grasses. Woody plants offer more complete and stable protection than grass alone. Trees and shrubs, especially thorny shrubs, provide refugia in which regeneration of other species is possible. Nitrogen-fixing species could help to raise the nitrogen status of some degraded lands, not all are nitrogen-deficient.

*Keywords*: carbon, nitrogen, water relations, nutrient recycling, degradation, woody plants, Africa

## Introduction

In many parts of the world the increasing density of 'woody weeds' has been identified as a symptom of land degradation; paradoxically, in other places the loss of trees is an indicator of degradation. In some circles there is an almost mystical faith in the power of trees to restore degraded landscapes; in others, trees are seen as competitors with grasses for scarce resources. What is their niche in arid lands?

This paper addresses four key questions with respect to trees in arid lands:

- does the inclusion of trees in the ecosystem increase net primary production?
- do trees confer greater interannual stability of primary production?
- do trees confer greater resilience to the ecosystem?
- do trees offer a pathway for recovery from degradation?

These questions ultimately translate into questions about differences in the resource access and use efficiency between trees and non-trees. The first section that follows will address the issue of resource limitation in arid ecosystems (which is the key resource that must be efficiently used?); the second section examines the way trees access and use resources.

For the purposes of this paper, woody plants are taken to include all long-lived plants (>10 years) in which a large proportion (>75%) of the biomass at maturity consists of secondary lignification (wood). This definition includes both trees, which are typically tall and single-stemmed, and shrubs, which are characteristically short and multi-stemmed. Trees and shrubs form a structural continuum and share many functional features. For brevity they will both be referred to as 'trees' in the paper.

### How does aridity control primary production?

It is axiomatic that water is an important constraint in arid lands, and it has been known for many years that net primary production in arid lands is highly correlated with water availability (Walter 1939, Rutherford 1980, Le Houerou & Hoste 1977, Scholes 1994). In the semi-arid tropics, the relationship between grass production and rainfall is essentially linear up to about 900 mm per annum of rainfall; above this it may begin to level off (see Ojima et al, Chapter 5). This is interpreted as indicating that water is the single overriding plant growth-limiting factor in arid lands, and that only once the water supply is sufficient does nutrient limitation become apparent.

The physiological mechanism by which water limitation operates is seldom discussed. It is assumed to be self-obvious that primary production is essentially to do with the carbon cycle, given that about half of plant biomass is carbon, and most of the rest is hydrogen and oxygen. The implicitly assumed mechanism is that a lack of soil water causes the plant water potential to become more negative and this ultimately causes the stomata to close. The decrease in stomatal conductivity prevents the diffusion of carbon dioxide into the leaf, causing carbon assimilation to slow down or stop. All the steps in this mechanism undoubtably do operate, and are responsible for the control of NPP at the time-scale of hours to days. Other mechanisms, however, may provide a more satisfactory explanation of the control of NPP at the monthly to annual timescale.

The main weakness of the carbon cycle mechanism for water limitation is that plants seem to assimilate far more carbon than appears in their tissues as biomass increment. The difference is lost through respiration, exudation from the roots, and volatilisation of hydrocarbons from the leaves; together these can make up more than the biomass increment. Even plants stressed by herbivory can invest large amounts of carbon in structural and chemical defences. In natural ecosystems the use of nutrients, by contrast, is extremely parsimonious.

Nutrient fluxes from undisturbed natural ecosystems are usually close to the threshold of analytical detection. When a tree or perennial grass sheds its leaves, most of the nutrients not physically bound into the cell structure have already been retranslocated into new growth or storage. Nitrogen-based herbivore defences (such as alkaloids) are usually only found in ephemeral plants, or for brief periods in the lifespan of perennial plants (Rhoades & Cates 1976).

If the mechanism of limitation is via the carbon cycle, why do fertilised plots (or naturally fertile soils) in arid lands produce more for a given amount of rainfall than unfertilised plots (eg, Donaldson, Rootman & Grossman 1984)? One explanation is that the maximum rate of carbon assimilation is strongly dependent on the leaf concentration of the primary assimilatory enzyme (ribulose-bisphosphate-carboxylase oxygenase), which makes up most of the nitrogen in the leaf (Mooney & Field 1989). A shortage of nitrogen could therefore affect the rate of carbon assimilation. Similar mechanisms are proposed for the role of phosphorus in adenosine triphosphate, the main energy carrier in the leaf. The relationships between carbon assimilation rate and leaf nutrient content offered in support of this mechanism hold across plant species and environments, but within a species and environment they are rather weak (Schulze et al 1994).

An alternate mechanism for the aridity constraint on primary production is based on the nitrogen cycle. Nitrogen is singled out from the other essential nutrients because it appears to be the most generally limiting nutrient of both primary and secondary production in arid lands, as evidenced by responses to added nitrogen. In subhumid systems, phosphorus may be equally important, and a similar set of arguments can be proposed. I speculate that the prevalence of nitrogen limitation in arid tropical ecosystems is because its cycle includes several major 'leaks' to the atmosphere; the most important in the context of dry lands are 'pyrodenitrification' and aerobic denitrification. While in the gaseous phase nitrogen can be transported away from the site.

The rate-limiting step for the assimilation of nitrogen by plants in natural ecosystems is seldom the capacity to take up inorganic nitrogen at the root surface. If this were the most limiting step, there would be a large pool of inorganic nitrogen in the soil -- in reality, it is generally very small in undisturbed natural ecosystems. The largest soil pool (indeed, the largest ecosystem pool) is non-living organic nitrogen, in the form of litter and humified substances. This implies that the rate-limiting step is the conversion of organic nitrogen to ammonium, through the process of nitrogen mineralisation.

The controls on this process are water availability, temperature, soil clay content and mineralogy and the amount and chemical composition of the organic substrate. In the semi-arid tropics, soil temperature is generally not limiting. Clay content and mineralogy are givens in the medium term and serve as the major constraints on the amount of organic matter that can accumulate in the soil. The soil fertility has a strong influence on the quality of litter, but not ultimately on the mineralisation rate, since the low-quality litter simply accumulates until its collective mineralisation rate equals the litterfall rate. Thus, in the short- and medium-term, water availability is the dominant control on nitrogen mineralisation rate, which in turn controls the nitrogen uptake rate. Under this hypothesis, nitrogen uptake controls NPP both through the Rubisco mechanism discussed above and through the tight coupling of the carbon and nitrogen cycles imposed by the stoichiometry of organic molecules (Nadelhoffer et al 1994).

Most biological molecules have a fixed proportion of nitrogen. The nitrogen content of a plant tissue is more variable, since the mix of molecules can alter within narrow bounds. Similarly, the proportions of different tissue types in the plant conveys a further degree of flexibility. Nevertheless, the rate of synthesis of new biomass can in principal be constrained by the availability of any one of its essential elemental building blocks, and nitrogen is frequently the one in shortest supply.

If there is excess carbon relative to another nutrient, the carbon would have to be respired away, or invested in low-nutrient structures, or in roots or root exudate, where it would help to stimulate nutrient availability. This is in fact what is commonly observed. On the other hand, if nitrogen were limiting, an increase in nitrogen availability, would lead to an increase in biomass production, which is commonly observed.

I proceed, therefore, on the tentative hypothesis that nitrogen limitation ultimately controls primary production in arid lands, and that water limitation operates principally through its effects on the nitrogen cycle.

## The ecological effects of trees in arid lands

### *Trees as deep rooters*

The classical analysis of the competitive interaction of trees and non-trees (principally grasses, but also including forbs) in arid lands is based on differences in rooting depth (Walter 1971, Walker & Noy-Meir 1982). It proposes a stable coexistence since grasses have superior access to water in the surface horizons, while trees have superior (or exclusive) access in the deeper horizons. This is generally true, but the degree of observed separation is small and inadequate to explain the patterns of resource use (Scholes & Walker 1993).

Both trees and grasses have most of their fine roots in the 0-30 cm zone, which is not surprising, since this is where the bulk of the water and nutrients (especially nitrogen) is located. Grass roots outnumber tree roots in absolute terms to a depth of a metre or more. Deep rooting by trees should be interpreted as a survival strategy, not a competition-avoidance mechanism. Access to deep water permits trees to survive during intra- and interannual droughts; for instance, trees suffered 5% mortality in the severe 1982-4 drought in southern Africa, while perennial grasses suffered 95% (Scholes 1985). Access to small amounts of water also allows trees to leaf-out earlier in the season than grasses, often before the first rains.

Recharge of the deep water resources constitutes only a few percent of the annual water budget, so in general, reliance on this resource is not a viable strategy. There are exceptions, such as the phreatophyte *Faidherbia albida*, but non-riparian habitats could only accommodate a small number of exceptions.

Deep rooted-ness allows trees to explore a greater soil volume for nutrients; this may be an advantage for mobile ions such as potassium and nitrate. It is less of an advantage for phosphorus, which is highly immobile, and little advantage for ammonium, which is located near the surface.

*Trees as temporal stabilisers*

The main axis of niche separation between trees and grasses is in time. The large storage volume and longevity of trees allows them to carry over substantial resources between seasons. Both root extension and leaf expansion take place in the first weeks of the growing season, largely using carbohydrates and nutrients assimilated in the previous season (Rutherford & Panagos 1982).

Grasses, on the other hand, have only enough storage products to initiate growth -- the bulk of the leaf area must be grown using the current season's assimilates. This allows trees to capture most of the resources available in first month or two of the growing season -- a time when nitrogen, in particular, is most available due to the mineralisation of the readily-decomposable litter from the previous season.

The carry-over of resources between seasons stabilises NPP by trees. This can be interpreted from the relationships between primary production versus rainfall for trees and grasses (figure 1). For grasses, this relationship has a negative y-intercept; as a result, the coefficient of variation (CV) of grass annual production is higher than the CV for annual rainfall.

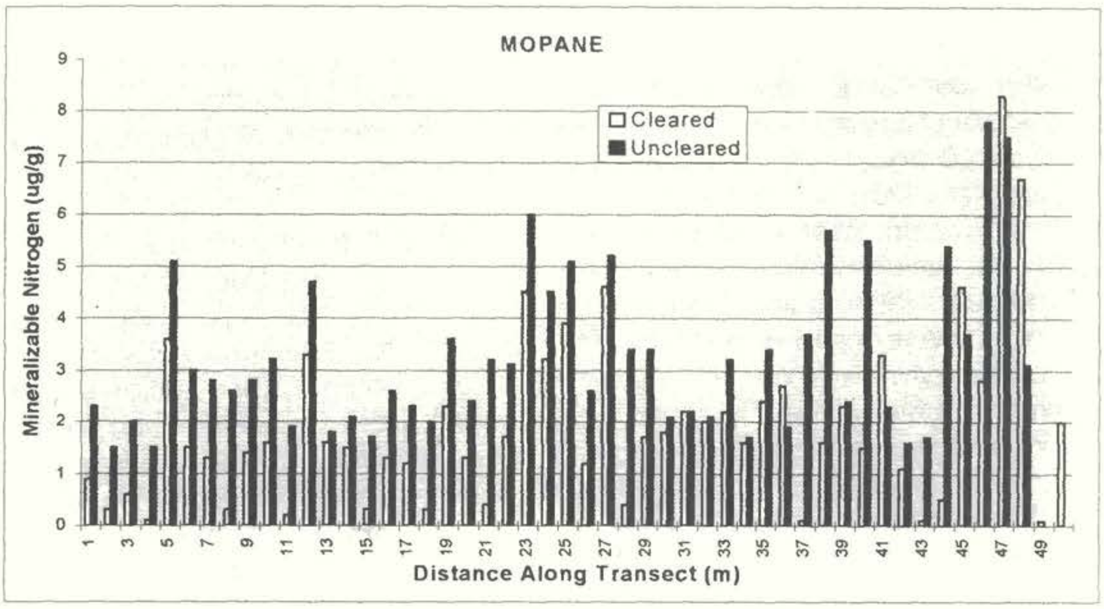


Figure 1. The readily mineralizable nitrogen content of the topsoil along a transect in *Colophospermum mopane* savanna and an adjacent area from which the trees were removed 15 years previously. The spikes represent areas below tree canopies. The patchiness created by the trees is still visible after clearing, but has become greatly reduced. (Gage 1995)

In trees growing in subhumid environments and on deep, sandy soils in semiarid environments, the intercept is positive and the slope is small; resulting in a CV smaller than interannual rainfall variation. The best predictor of stem increment and leaf litterfall in semi-arid savannas includes both the current season's rainfall and the previous season's rainfall. In very arid environments the relationship for trees is very similar to that for grasses - both are dependent on rainfall in the current season, and both have a negative y-intercept (de Ridder et al 1982)

*Trees as spatial organisers*

It is frequently stated that trees operate as 'nutrient pumps'; usually meaning that they bring nutrients from deep in the profile and deposit them at the surface. A quantitatively more important pumping function, particularly for nitrogen, operates laterally. The lateral extent of tree roots in arid lands is many times the lateral extent of the canopy (Groot & Soumare 1995 ).

The nitrogen is returned to the soil mainly via leaf litterfall, which largely occurs below the canopy. The result is a strong concentration of organic matter and nitrogen below the tree canopy, leading in many savannas to differences in both the primary production and species composition of the herbaceous layer. These concentrations persist for a number of years after tree removal, but gradually dissipate (Figure 1).

There is a difference of opinion in the literature regarding the ecosystem-level consequences of nitrogen patchiness. The North American 'Jornada school' see it as a mechanism of degradation, since it impoverishes the between-patch areas, whereas the Australian 'Mulga school' see it as a valuable enrichment of the landscape. The concentration of resources is beneficial where the y-intercept of the production versus resource availability relationship is negative.

This is certainly the case for secondary production; a minimum of about 1% nitrogen is required in forage before ruminants can digest it. For primary production this may not be true, and in the Jornada example the nitrogen may be over-concentrated.

*Trees as nitrogen fixers*

The tree flora of arid lands is often dominated by legumes. However, not all legumes have the capacity to form symbiotic associations with bacteria capable of converting atmospheric dinitrogen to ammonia; and those that have the capacity frequently do not express it in the wild.

Of the three families which make up the legumes, the capacity to form N-fixing nodules is concentrated in the Papilionaceae and Mimosaceae, and generally absent in the Caesalpinaceae (which dominate the subhumid zone in Africa). The Papilionaceae seldom dominate as trees, but are important in the herbaceous layer. The typical rates of N fixation are 20-60 kg N/ha/yr, although rates of 200 kgN/ha/yr are possible (Nair 1984, Dommergues 1987).

Why do the Mimosaceae, which includes *Acacia* and *Prosopis* and dominate in arid areas, not always fix nitrogen? Possible explanations include a shortage of necessary cofactors, such as phosphorus or molybdenum, or that it is not energetically efficient for them to do so. The support of nitrogen-fixing bacteria can consume up to half of the carbon assimilated by the tree. If the availability of mineralised nitrogen is relatively high, it may be more efficient for the tree to rather invest in more roots.

Whether or not they obtain their nitrogen by fixation, members of the Mimosaceae have a higher leaf N content than members of the Caesalpinaceae (Table 1), which in turn have a higher leaf N than non-legume trees, which are higher than grasses.

Table 1. The leaf nitrogen content (mg/g) of legume trees, non-legume trees and grasses. Sources: Scholes & Walker (1993), Schulze et al (1994).

| Trees from legume families | | Non-legume trees | Grasses |
|---|---|---|---|
| Mimosaceae and Papilionaceae | Ceasalpinaceae | | |
| 27.1 | 19.1 | 24.5 | 10.7 |

*Trees as nutrient retainers*

The presence of trees in the ecosystem helps to reduce the rate of loss of nutrients from the system, particularly when it is subject to perturbation by drought, grazing or cultivation. Holding a significant portion of the ecosystem nitrogen pool in a medium-term turnover pool such as tree biomass buffers the system against short-term losses.

The stability of the tree cover affords protection to the soil surface when the grass cover is not present, reducing soil erosion, which is frequently the major avenue of nutrient loss. The deep roots and extended growing season of the trees offers an opportunity to capture nitrate which may have leached below the grass root zone, and the rapid uptake of inorganic nitrogen reduces the chance of denitrification. Since only a small fraction of the tree biomass is consumed by herbivores, only part of the plant N is prone to volatilisation from urine and faeces. The N in tree trunks and leaves is largely protected from pyrodenitrification during low-intensity arid land fires.

*The nitrogen use efficiency of trees*

'Resource use efficiency' can have many meanings. In one sense it is the amount of dry matter accumulated per unit of the resource used; in this case gDM/gN. In this sense trees have a higher nitrogen use efficiency than non-trees simply because wood has an extremely low nitrogen content (0.25%), whereas grass leaves are in the region of 1-2%N. In other words, one gram of nitrogen could result in 400 g of wood production, but only 50-100 g of grass production. Since the C:N ratio of soil organic matter (including soil litter) is usually in the region of 20, the carbon stored in a nitrogen-limited ecosystem can actually be increased by allowing some of the SOM to be mineralised and taken up by trees.

Another definition of resource use efficiency is the ratio of losses to turnover in the system. Where the 'system' is defined as an individual tree, the loss of N through litterfall is typically about 40% of the N needed for new growth - in other words, about 60% is internally recycled (Scholes & Walker 1993). The fraction in grasses is lower, mainly because they do not have organs for storage of large amounts of N. At the whole ecosystem scale, it is likely that treed ecosystems have smaller losses relative to turnover than non-treed systems (particularly in the face of disturbance), for the reasons outlined in the previous subsection. I know of no arid-land catchment-scale nutrient balance studies to validate this hypothesis.

*Trees as agents of ecosystem rehabilitation*

Trees are not a panacea for degraded ecosystems, but they can make a valuable contribution in some cases. In particular, where it is necessary to reduce the rate of nutrient loss from the ecosystem due to leaching or erosion, trees in combination with grasses offer more complete and stable protection than grasses alone. Trees (especially thorny shrubs) provide refugia in which regeneration of other species is possible. Nitrogen-fixing trees could help to raise the nitrogen status of some degraded lands; however, not all are nitrogen-deficient.

Tree seeds are frequently very long-lived, and so will still be present after long periods of no seed production. The tree establishment phase, however, is extremely vulnerable to drought and herbivory. This is the primary constraint to the role of trees in rehabilitation.

Bush encroachment is a self-limiting process once it has reached its terminal state. The soil is protected, the grazier who initiated the process is bankrupt, and the essential ecosystem processes are conserved. Recovery to a sparsely-treed state will not occur spontaneously in the short term, but is likely to occur gradually over a period of several tree lifespans.

## Discussion

The removal of trees from savannas is a widespread practice, justified in terms of improving the ecosystem productivity. This is based on a narrow view of productivity, in which the only useful production is that which can be fed to cattle. Total ecosystem NPP is almost certainly higher in treed arid lands than in savannas from which the trees have been removed, despite the fact that the peak instantaneous rates of carbon assimilation by grasses are usually slightly higher than for trees, and the carbon assimilated per unit water used or per mole of photons intercepted is also higher for $C_4$ grasses than $C_3$ trees. The failure of these predictors is due to their being denominated in carbon assimilation terms, which is not synonymous with biomass production.

Is higher NPP necessarily a good thing? The answer obviously depends on the management objective. As previously mentioned, higher NPP is not desirable for cattle farmers if it comes at the expense of forage. Subsistence pastoralists tend to use a broader spectrum of ecosystem products than commercial ranchers do, including wood for fuel, construction, crafts and medicine, browse for goats and fruit, insects and honey for food. Consequently,'bush encroachment' is seldom a problem in areas of subsistence pastoralism; tree harvest in excess of the replacement rate often is a problem (sometimes called 'deforestation' despite the fact that arid savannas are not forests).

If the objective is ecosystem conservation, then NPP is irrelevant unless the system has become degraded. The process of recovery from degradation involves rebuilding nutrient, energy and carbon pools and fluxes; higher NPP both promotes this process and indexes its progress. Finally, if the objective is carbon sequestration, higher NPP is an objective, along with the greater interannual stability and carbon pool size and security offered by woody plants.

If trees are good, are more trees better? Not necessarily, again depending on the management objective. It is an open question whether there is an optimum tree cover (less than the maximum possible cover) with respect to maximising NPP in savannas. I speculate that in arid and semiarid savannas (<900 mm/y) there is, and the ecosystem will trend towards it if permitted to do so. It will occur where tree-on-tree competition limits tree production.

Grasses and forbs will then grow to exploit whatever resources remain in the system. The access which trees have to resources beyond the rooting depth of grasses can cause problems; for instance, the planting of deep-rooted, fast-growing eucalypts in southern Africa causes a reduction in streamflows. Conversion of grasslands or savannas to thickets or woodlands will reduce the habitat available for grass-dependent animals; for biodiversity maintenance purposes there is likely to be an optimum tree density.

Finally, at the arid extreme, a multi-year drought-tolerator growth stategy is only viable within landscape niches which receive a supplementary water supply - typically in valley bottoms. attempts to establish trees elsewhere will fail.

## Conclusions

The inclusion of trees in nitrogen-limited ecosystems, which includes most arid-land ecosystems, can increase net primary production. There may be an upper limit to the tree cover for optimal NPP and there are climate limits below which the tree life strategy is unviable.

The presence of trees reduces the interannual variability in NPP, due to carry-over of resources between years (either within the tree, or in ecosystem pools inaccessible to other plants) and the inertia inherent in long-lived species.

Trees increase the resilience of arid ecosystems to drought and episodes of intense grazing, due to their partial decoupling from recent rainfall and their inaccessibility or unpalatability to grazers respectively.

Trees have a role in the rehabilitation of arid lands, as part of an overall strategy including grasses and amelioration of the degradation-causing processes. The re-establishment of trees in landscapes from which they have disappeared requires a significant input of effort; as does the reduction of tree density in landscapes where they have become overabundant.

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# Dynamics of carbon storage in degraded arid land environments: a case study from the Jornada experimental range, New Mexico, (USA)

Sean L. Connin, Ross A. Virginia, Page Chamberlain, Laura Huenneke, Kevin Harrison, and William H. Schlesinger

Synopsis

This paper *reviews* spatial and temporal dynamics of carbon storage related to Holocene climatic change and recent shrub invasion (due to desertification) in the Jornada Basin, New Mexico (USA) It *evaluates* the role of regional climate history, vegetation distribution, and soil characteristics as controls on carbon cycling in this system. Ecosystem-level changes in above- and below-ground carbon pools in grassland and desertified shrub communities are compared to quantify changes in the net carbon balance associated with desertification

An assessment is made, in the context of the Jornada desertification model, of the potential opportunities to sequester carbon in drylands and to identify current research needs.

*Key Points*

1. At present we know little about the responses of desert climates and carbon storage to rising atmospheric $CO_2$ concentrations and coincident global warming. but it is likely that soil management strategies in dryland regions will have a greater impact on soil carbon storage than will climate change or $CO_2$ patterns in southwestern USA indicates long-term-term climatic warming accompanied by more recent oscillations.. During the next century, climate warming may promote a transient period of increasing continental drought. Such non equilibrium responses to climatic warming will likely shape future policy directives to combat rising atmospheric $CO_2$ and to enhance terrestrial C storage.
2. Models of climate change impacts on vegetation distribution and carbon storage provide a first approximation of shifts in the magnitude and rate of terrestrial carbon fluxes and offer an opportunity to evaluate mitigation strategies Unfortunately, global terrestrial carbon models are limited by an inadequate understanding of ecosystems and their response to disturbance. The response of arid land carbon pools to future climatic change is particularly unconstrained. These inadequacies highlight a need for direct quantification of carbon dynamics in dryland regions, particularly in response to desertification.
3. Over the past century, degradation of semiarid and arid lands apparently reduced carbon storage in these ecosystems worldwide. In the southwestern USA, intensive grazing by livestock and coincident periods of drought interacted to promote desertification and the replacement of native grasslands by invasive shrubs. Shrub invasion increases the spatial and temporal heterogeneity of soil resources and plant biomass. modifying the fundamental structure and function of arid land ecosystems. These impacts alter biogeochemical cycling at landscape and global scales.

4. Modern dryland ecosystems comprise a relatively minor component (c. 7%) of the pool of organic carbon in soils. which totals c. 1400-1555 Pg . In arid lands an additional 800-1738 Pg carbon is incorporated in pedogenic carbonates. However, slow turnover times relegate carbonates to a minor role in the global carbon cycle. Over geologic time, moisture/temperature gradients determine the capacity of soils to store organic carbon. Desertification processes disrupt climatic control on detrital production and decomposition, accelerating the release of $CO_2$ from soils. resultant changes in carbon pools reflect a balance between the fluxes in the above-ground and below-ground biomass and soil organic carbon.
5. In the southwestern USA, desertification (i.e. shrub invasion) alters dryland carbon cycling via changes in plant biomass distribution and soil disturbances. Potential influence of post-industrial $CO_2$ fertilisation on soil carbon storage appears to be negligible. The transition from grassland to shrubland may be analogous to proposals to produce halophyte biomass in arid and semi-arid ecosystems to increase carbon sequestration in vegetation and soil. Impacts of shrub invasion (on carbon storage) indicate the potential carbon sinks produced by halophyte biomas production or other afforestation/reforestation projects may be modest and must be assessed in the context of agricultural practices - which may produce "collateral" reduction in soil carbon pools and inadvertent release of $CO_2$ from other activities.

Keywords: carbon stores, soil carbonates, New Mexico, rangeland, Holocene, Pleistocene, nutrient cycling, vegetation dynamics, Carbon cycling, dryland, desertification, shrub invasion, soil organic matter.

## Introduction

Attempts to model global dynamics of terrestrial carbon pools are limited by inadequate understanding of ecosystems and their response to disturbance. The response of arid land carbon pools to future climatic changes is particularly unconstrained. Desertification often increases the spatial and temporal heterogeneity of soil resources and plant biomass, modifying the fundamental structure and function of arid land ecosystems (Schlesinger et al. 1990). These impacts alter biogeochemical cycling at landscape and global scales (Schlesinger et al. 1990); however, relationships between desertification and carbon cycling are poorly understood.

Over the past century, degradation of semiarid and arid lands apparently reduced C storage in these ecosystems worldwide. Ojima et al. (1993) suggest that the levels of organic carbon in grassland/dryland regions of the world decreased 13.2 - 25.5 Pg between 1800-1990, primarily because of cropland conversion and cultivation. However, these estimates do not specifically address impacts of desertification on C storage. Approximately, 3.6 billion ha, or 70% of the world's non-forested drylands (29% of the world's land surface) have been desertified, resulting in lost net primary productivity (NPP) and $CO_2$ release to the atmosphere (Daily 1995; Dregne et al. 1991). An additional ~9 to 11 million ha drop out of economic production each year due to land degradation (Daily 1995).

Modern dryland ecosystems comprise a relatively minor component (~

7%) of the pool of organic C in soils, which totals ~ 1400-1555 Pg C in various fractions of soil organic matter and surface detritus (Eswaran et al. 1993, 1995; Post et al. 1982; Schlesinger 1977). An additional 800-1738 Pg C is incorporated in pedogenic carbonates ($CaCO_3$) forming in dryland soils of the world (Eswaran et al. 1995; Schlesinger 1982, 1985). At present, we know little about the response of desert climates and C storage to rising atmospheric $CO_2$ concentrations and coincident global warming. For example, Emanuel et al. (1985) predicted a 6-17% increase in the world's desert area under doubled atmospheric $CO_2$, which would result in a net release of C from desertified regions. In contrast, Prentice and Fung (1990) predicted a 62% decrease in the area of desert land; however, their model over-predicted moisture during the last glacial when deserts were widespread.

Smith et al. (1992) employed several general circulation models (GCMs) to compare shifts in ecosystem complexes and associated carbon storage under four climatic change scenarios. They reported a significant reduction of desert area, particularly from replacement of deserts by temperate grasslands. Similarly, model projections of Ojima et al. (1993) indicate an expansion of warm grasslands (coupled with increased soil organic C) under doubled atmospheric $CO_2$, creating a net C sink of 5.6-27.4 Pg. Greatest C storage was observed under conditions of optimal soil management (e.g. grazing intensity and fire frequency). The authors suggested that soil management strategies in grassland/dryland regions will have a greater impact on carbon storage than will climate change or $CO_2$ enhancement.

During the next century, climatic warming may promote a transient period of increasing continental drought. In model demonstrations, Rind et al. (1990) found that increased potential evapotranspiration (relative to precipitation) can decrease soil moisture and/or increase vegetative stress. They predict that a 4° C warming due to doubled atmospheric $CO_2$ would increase the frequency of severe drought in the United States from 5% today to 50% by the 2050s. Identifiable effects of increased drought could appear as early as the 1990s. Such non-equilibrium ecosystem responses to climatic warming will likely shape future policy directives to combat rising atmospheric $CO_2$ and to enhance terrestrial C storage.

Models of global change impacts on vegetation distribution and carbon storage provide a first approximation of shifts in the magnitude and velocity of terrestrial C fluxes and offer an opportunity to evaluate mitigation strategies. Unfortunately, many global terrestrial carbon models exclude hyperarid regions (P/PET $< 0.05$), pools and fluxes of pedogenic carbonate, and effects of shrub invasion resulting from desertification - - largely due to lack of representative data (e.g. Ojima et al. 1993). In addition, these models are also limited to equilibrium solutions in which forcing variables (e.g. climate, $CO_2$ concentration) and response variables (e.g. ecosystem type, carbon pools) reflect steady-state conditions. In reality, soils may require at least 3000 yrs. to achieve steady state concentrations of organic matter in response to changing climate or vegetation (Birkland 1984; Schlesinger 1990). These inadequacies highlight a need for direct quantification of carbon dynamics in dryland regions, particularly in response to desertification.

While drylands contain a small proportion of the pool of C in soils, they may provide extensive areas for reforestation and afforestation projects designed to sequester additional C through increased areal NPP (UNEP 1991). These possibilities are particularly attractive in light of research indicating the potential importance of terrestrial vegetation to serve as a $CO_2$ sink (Dixon et al. 1994).

This paper reviews spatial and temporal dynamics of carbon storage related to: Holocene climatic change and recent shrub invasion (due to desertification) in the Jornada Basin, New Mexico (USA). We also evaluate the role of regional climatic history, vegetation distribution, and soil characteristics as controls on C cycling in this system. Results will be discussed in the context of the Jornada desertification model (Schlesinger et al. 1990) and used to assess potential opportunities to sequester C in dryland environments and to identify current research needs.

## The Jornada Experimental Range

This analysis was compiled from data gathered at the Jornada Experimental Range approximately 40 km NNE of Las Cruces, Dona Ana County, NM, USA. The area encompasses approximately 58,470 ha of the Jornada Del Muerto basin in the northern Chihuahuan Desert (Hennessy et al. 1983) ranging in elevation from 1,260 m on basin plains to 2,833 m at the crest of the San Andreas Mountains (Gibbens et al. 1983). Overall, the climate is arid; mean annual precipitation (MAP) is 211 mm (Houghton 1972) with 52% falling between July and September. Transition to semiarid conditions occurs above 1,524 m due to increased precipitation, MAP 250-300 mm (Gile and Grossman 1979). Ambient temperatures occasionally fall below freezing during winter months and may exceed 40í C in the summer.

Over the past century, overgrazing and drought in the Jornada Basin promoted replacement of native grasslands by shrub communities (Buffington and Herbel 1965; Gibbens et al. 1983; Hennessy et al. 1983). Invasion of shrubs, primarily the woody legume mesquite (*Prosopis glandulosa*), into native black grama (*Bouteloua eriopoda*) and poverty threeawn (*Aristida divariacata*) grass communities has been followed by loss of grass cover, soil loss and redistribution, and the formation of extensive areas of mesquite coppice dunes.

At present, mesquite shrubs dominate ~ 63% of the experimental range, increasing from 15,500 ha in 1858 to 37,800 ha in 1963 (Buffington and Herbel 1965). Extensive efforts to control, exclude, or eradicate mesquite within valuable forage grasslands have met limited success (Cable and Martin 1973, Westoby et al. 1989) and despite such efforts, mesquite cover often continues to increase (Brown 1950; Fisher 1977; Wright 1982). It appears that reversing the impacts of shrub invasion naturally or through human intervention is difficult at best (Conley et al. 1992).

## Pools of Soil Carbon in the Jornada Basin

Grossman et al. (1995) recently tabulated estimates of organic and inorganic (carbonate) soil C based on divisions in aridity and soil texture for 90,000 ha of the Jornada Basin. The weighted average organic C content for arid portions of the study area is 3 kg/m2, exceeding estimates for desert shrublands in Arizona (2.2 kg/m2) (Schlesinger 1982) and for the world's warm deserts (1.2 kg/m2) (Post et al. 1982) - - the life zone classifications that incorporate the Jornada Basin region. However, this figure is less than estimated soil C in desert scrublands globally (5.6 kg/m2) (Schlesinger 1991).

Kimble et al. (1990) reported a global estimate of organic C content (4.2 kg/m2) in Aridisols and emphasized that high intrinsic variability is associated with such samples (CV=60%). Globally, Aridisols contain ~ 2.2% of organic C stored in soils (Buringh 1982).

Grossman et al. (1995) report that in the Jornada Basin, concentrations of organic C are inversely related to abundance of >2mm rock fragments. In arid portions of the study area, skeletal soils (>34% vol. rocks) contain the lowest organic C contents (< 2 kg/m$^2$). Semiarid zones contained greater relative amounts of organic C (> 6 kg/m$^2$). Across the basin, highest organic C concentrations are associated with fine-silt soils derived from limestone (~10 kg/m$^2$). In addition, high surface concentrations of organic C occur in coarse-textured coppice dunes due to strong biological activity. The positive relationship between $CaCO_3$ and organic C content is poorly understood (Grossman et al. 1995) and may not be found in arid land soils globally (Schlesinger 1982). However it is interesting to speculate that moisture retention by carbonate (Hennessy et al. 1983) and/or formation of Ca-humate complexes (Oades 1988) may contribute to these patterns.

Within Jornada soils, carbonate C exceeds organic C pools by ~6-fold (Grossman et al. 1995). In comparison, Schlesinger (1982) determined that carbonate C exceeds organic C contents in Arizona soils by 4 to 5.7 times. Carbonate contents in arid and semi-arid portions of the basin (in noncalcareous parent material) are 29 kg m$^{-2}$ and 13 kg m$^{-2}$, respectively; producing an area weighted mean of 24 kg m$^{-2}$. Carbonate contents in limestone-derived soils are much greater: 43 kg m$^{-2}$ and 37 kg m$^{-2}$, respectively.

Grossman et al. (1995) indicate that carbonate accumulation rates in the Jornada Basin (0.1-1.4 g m-2 yr$^{-1}$) are similar for soils ranging from mid-Pleistocene age (ca. 250-900 k yr BP) to those formed during the Holocene. In contrast, Machette (1985) estimated that carbonate accumulated in Holocene soils at twice the rate as during Pleistocene pluvial episodes in the southwestern United States. The reported increase may have resulted from decreased vegetation cover during the Holocene, promoting erosion and increased concentrations of airborne $Ca_2$+ and $CaCO_3$ . In contrast, Schlesinger (1985) calculated Pleistocene carbonate deposition rates of 1.6 to 3.5 g m-2 yr$^{-1}$ in the Mojave Desert. He suggests that greater rainfall during the Wisconsin glaciation provided Ca-rich solutions to soil profiles, promoting carbonate deposition.

Soils may derive carbonate from calcareous parent material, from chemical weathering of Ca-bearing minerals in soils or from an influx of $Ca_2$+ in aerosols and precipitation (Marion 1989; Schlesinger 1985). Only carbonate formed from silicate mineral weathering represents a net transfer of atmospheric $CO_2$ to the soil C pools (Schlesinger 1991). Chadwick and Capo (1993) report that ~ 95% of $Ca_2$+ in calcretes sampled in the Jornada Basin originates from dust and precipitation influx. This suggests that long-term dynamics of pedogenic carbonate do not determine net carbon balance in this ecosystem. On the whole, slow turnover times relegate desert carbonates to a minor role in the global C cycle (< 0.023 Pg C yr$^{-1}$) (Schlesinger 1990).

## Southwestern Paleoenvironments and Their Vegetation

In order to understand spatiotemporal dynamics of carbon storage in the Jornada Basin, both site prehistory and recent desertification are considered to elucidate controls on carbon flux. Reconstruction of Pleistocene to Holocene vegetation patterns in the southwestern USA indicates long-term climatic warming accompanied by more recent short-term oscillations. Van Devender and Spauding (1979) analyzed fossil plant assemblages in packrat middens in the Southwest to infer climatic changes during the past 22,000 yrs. Middens dating from the late Wisconsinan glacial maximum (22,000 to 17,000 yrs. BP) indicated presence of pinyon-juniper woodlands accompanied by a perennial $C_4$ grass understory at elevations (1,525 to 2,200 m) now occupied by desert scrub communities. Brakenridge (1978) suggests that temperatures in this pluvial environment were 7-8° C cooler and precipitation somewhat less (and more evenly distributed between winter and summer seasons) than at present. Other studies indicate cooler, wetter winters during the late Pleistocene (Marion et al. 1985) in locations proximal to the Jornada Basin (Wells 1966).

During the mid-Holocene (~8,000 yrs. BP) woodland plants were displaced northward and to higher elevations by present-day desert communities. In the Chihuahuan desert, expansion of grasslands appears to have coincided with warming temperatures, reduced winter precipitation and expansion of the summer monsoon. Summer rainfall was probably greater than at present. Freeman (1972) analyzed fossil pollen of radiocarbon-dated alluvial deposits in the Jornada Basin and discovered a shrub -to- grassland transition between 5000 - < 4000 BP. Late Holocene ("little ice age") cooling resulted in expansion of desert shrubs producing a mosaic of shrub- and grass-dominated communities in the Jornada Basin (Neilson 1986). Differences in atmospheric flow patterns that dictate seasonal precipitation patterns may have influenced these vegetation dynamics (Neilson 1986).

Carbon isotope analyses of light and heavy fractions of organic C (Stevenson and Elliot 1989), reflecting active and stable soil organic carbon pools (Hsieh 1992), respectively, in a Jornada Basin soil profile provide additional evidence of a prehistoric $C_3$ to $C_4$ (i.e. shrub -to- grass) shift in plant species (Figure 1). A gradual d13C depletion of soil organic matter (with depth) suggests increased importance of $C_3$ species in prehistory (Ludlow et al. 1976). This site is presently dominated by black grama, a $C_4$ grass. In terrestrial ecosystems, relative $C_3$/ $C_4$ biomass reflects characteristics of local moisture and temperature (Teeri and Stowe 1976). Conditions of moisture stress and high light intensity favor $C_4$ grass species. Thus, the d13C analysis of the soil profile supports a scenario of climatic warming through the Holocene.

Importantly, the d13C profile also suggests soil organic C fractions are in disequilibrium relative to the current plant community. Approximately, 93% and 100% of heavy and light fractions at the soil surface are derived from $C_4$ sources, respectively. $C_4$ dominance declines to 51% at 100 cm in the heavy fraction and to 72 % at 90 cm the light fraction. The d13C lag between heavy and light fractions represents differences in their respective turnover times. Light organic fractions apparently consist of labile material, actively contributing to soil C flux. Heavy organic fractions represent more recalcitrant material linked to long term soil C storage (Spycher et al. 1983). To assess impacts of climatic changes on soil C storage, it is necessary to differentiate between bioreactive forms which may respond to short-term impacts and stable forms which reflect long-term trends in the environment (Cheng and Molina 1995). Assumptions of soil organic C (steady-state) response to environmental change may be inappropriate within regions impacted by shrub invasion.

## Implications for Carbon Storage Through the Holocene

Climate changes influence vegetation distributions, primary production, and detrital decomposition. Following late-Pleistocene warming and glacial retreat, terrestrial ecosystems appear to have been a sink for C due to expansion of exposed land area (Adams et al. 1990). In the Jornada Basin mean annual precipitation has not varied significantly since mid-Pleistocene. However, a shift to monsoonal distribution patterns coupled with rising ambient temperatures may have increased evaporative demand, promoting decreased primary productivity and increased decomposition rates. As a result, southwestern soils may have been a net source of atmospheric $CO_2$.

Climatic warming may reduce organic C storage in soils if organic inputs decline while decomposition rates remain steady or increase (Schlesinger 1986). At present, predicted global warming is likely to increase soil respiration worldwide (Raich and Schlesinger 1992; Schleser 1982). Post-glacial climatic warming in the southwestern USA provides an intriguing backdrop on which to evaluate effects of climatic change on soils. Unfortunately we have no concise climatic records or soil C estimates throughout this period. However, analysis of soils developing under similar vegetation along altitudinal (i.e. precipitation/temperature) gradients may elucidate relationships between environmental factors and potential soil C storage.

Gile (1977) analyzed effects of increasing precipitation on soil C along an arid-semiarid transition on piedmont slopes in the Jornada Basin. On both monzonite and rhyolite parent material, he noted an increase in organic C storage and thickness of the noncalcareous zone with a mountainward increase in precipitation. Between 1,280 and 1,768 m, precipitation increased from ~ 22 cm $yr^{-1}$ up to 35 cm $yr^{-1}$ and organic soil C increased almost four fold. Soil textural properties did not play a significant role in these trends. Percent organic C was weakly related to % silt ($R^2$=.4094, P=.0002) and % sand ($R^2$=.2870, P=.0023). Curiously, no significant relationship existed between % organic C and % clay, perhaps due to the small sample number (n=28) and/or low clay contents of the sample pedons.

In arid land soils, organic C contents are positively correlated with annual precipitation (Gile 1977; Schlesinger 1982). As a result, organic C in soils at higher elevations in the Jornada Basin (ie. 10.8 kg $m^{-2}$) may reasonably estimate average soil C in a cool/wet late-Pleistocene climate. Assuming this is true, C stored in arid portions of the basin (i.e. lower elevations) decreased ~ 3-fold during the Holocene. While these calculations neglect patterns in soil C storage due to chronosequence accumulation (Jenny 1980), three Jornada soil profiles of different age, at similar elevations, and with identical parent substrate and texture contain similar amounts of C, indicating that climate ultimately determines C storage (Table 2).

Modern Desertification And Shrub Invasion In The Jornada Basin

During the last century large areas of semiarid grassland have been lost in favor of desert shrubland in southern New Mexico (Buffington and Herbel 1965; Gibbens et al. 1983; Hennessy et al. 1983). A variety of mechanisms have been proposed to explain this transition including overgrazing, fire suppression, drought, and rising atmospheric $CO_2$ (Grover and Musick 1990; Humphrey 1958; Johnson et al. 1993; Schlesinger et al. 1990). In a recent review, Grover and Musick (1990) concluded that shrub expansion resulted primarily from livestock overgrazing at the end of the nineteenth century, coupled with coincident periods of drought.

Displacement of grasses increases spatial/temporal heterogeneity of soil resources and standing plant biomass across the landscape (Bach et al. 1986; Nelson 1934; Pluhar et al. 1987), perpetuating positive feedbacks which favor shrub establishment (Schlesinger et al. 1990). Following shrub invasion and associated soil redistribution, detrital production, and biotic activity consolidate beneath shrub canopies forming "islands of fertility" (Schlesinger et al. 1990; Virginia and Jarrell 1983). These variations disrupt historical patterns of detrital production and decomposition (Whitford 1992).

## Changes in Organic C Storage

Grass to shrub species transition shifts below-ground litter input from fine root biomass (distributed in surface horizons) to deeply rooted, structurally resistant woody material. These structural changes may enhance soil C storage lower in the soil profile. In some Amazonian forests, soil organic C beneath 1 m exceeds that of aboveground biomass (18 kg m-2), due to C allocation to deep roots by woody plants (Nepstad et al. 1994). Deep rooted grasses in American savannas also sequester significant amounts of carbon within soil profiles (Fisher et al. 1994). In drylands, higher mass loss rates associated with fine root biomass relative to coarse roots (Fisher et al. 1990) and decreased microbial biomass with soil depth (Jenkins et al. 1988; Virginia et al. 1992) suggests that shrub invasion may result in greater long-term C storage in soils. However soil erosion and associated loss of soil C may balance these inputs (Gibbens et al. 1983).

To determine landscape impacts of shrub invasion on carbon cycling, we calculated effects of community shift on above- and belowground C pools for 58,400 ha of land contained within the Jornada Experimental Range. The transition in vegetation has caused some significant changes in C storage (Table 3). Among invasive shrublands, only mesquite communities contain more biomass C (g $m^{-2}$) than grasslands they have replaced, largely as a result of their extensive root networks. However, mesquite and creosote-bush (Larrea tridentata) communities contain lower amounts of soil organic C than the former grasslands. Assuming similar soil C contents (across sites) prior to shrub expansion, mesquite communities maintain total C at historical levels (by balancing soil C loss and biomass gain) while creosote-bush communities lose C.

Tarbush (Flourensia cernua) communities exhibit the lowest amounts of biomass C and yet the highest soil C contents, double that of grasslands. As a result, they contain the greatest total ecosystem C among sites; however, it is likely that these sites contained greater soil C prior to shrub establishment. Apparent C sequestration in tarbush communities may reflect their proliferation in lowlands likely to receive organic C (due to soil transport) and presence of heavy clay-rich soils that retain organic C (Oades 1988).

Net changes in the C pool during desertification reflect a balance between fluxes in above- and below-ground biomass and soil organic C. We calculated resultant shifts in C using a weighted average of biomass on the landscape as a function of bare and vegetated spots. Except in tarbush sites, shrub invasion results in large soil C losses relative to grass communities. However if tarbush sites are included, shrubland succession results in a slight increase in soil C storage of ~ 40,000 mt or 0.7 g m-2 yr-1 between 1853 and 1963.

These calculations attribute differences in standing biomass only to changing plant types. Given an increase of atmospheric $CO_2$ concentrations (~ 27%) since the early 1800's (Neftel et al. 1985), it is also possible that $CO_2$ fertilization influenced plant competition and biomass C storage during desertification (Johnson et al. 1993). For example, BassiriRad et al. (1995), noted greater biomass accumulation in two $C_3$ shrubs (mesquite and creosote-bush) compared to a $C_4$ grass (black grama) grown under doubled atmospheric $CO_2$ in controlled conditions. However, the grass demonstrated greater capacity for nutrient uptake, which may provide it competitive advantage during long-term exposure to elevated $CO_2$.

In this study we use Harrison et al.'s (1993) C storage model to estimate the maximum possible influence of $CO_2$ fertilization on soil C storage during shrub invasion. Assuming steady-state conditions in which all plant productivity enters soil C pools, we calculate a 6-yr turnover time for active soil C in grass communities and a 5-35 yr turnover in shrub communities (Table 4). These values are greater than those usually reported for temperate grasslands (20-50 yrs) (Parton et al. 1987), resulting in a greater sensitivity of soil C storage to CO2 fertilization. Assuming a CO2 enrichment factor of 0.35 and that the soil profile contains 50% active and 50% passive C (t = 4,700 yrs) (Harrison et al. 1993), maximum C sequestration due to fertilization is only 3-4 % of the total modern inventory. As a result, we conclude that potential effects of CO2 fertilization on soil C pools have been negligible.

Shrub ecosystems may ultimately accumulate soil C (relative to grasses) due to differences in plant growth. Cebrian and Duarte (1995) reported that soil detrital C mass is inversely related to plant turnover rate (primary productivity/plant biomass) rather than to NPP or detrital flux, suggesting that slow growing plants create C sinks. They suggest that differences in litter quality between growth types (i.e., slow turnover = nutrient poor) determine the relative lability of soil detrital C, and ultimately soil C storage capacity. If this is true, then invasive shrubs (low biomass turnover) may enhance C storage relative to grass communities (high biomass turnover). Since community change in the Jornada Basin began recently (~ 100 yrs. ago), the recent loss of organic soil C may precede future steady-state increases. However, litter quality may not control decomposition in desert environments (Schaefer et al. 1985) and, subsequently, plant life strategy may not allow prediction of soil C storage.

The transition from grassland to shrubland in New Mexico is analogous to proposals to plant woody vegetation over large areas of arid and semiarid ecosystems to increase carbon sequestration in vegetation and soils (Glenn et al. 1993). Without considering the carbon costs (i.e., $CO_2$ emissions associated with planting and irrigation), our opinion is that such land management is not likely to have dramatic effects on overall carbon storage. It is also interesting to note that such proposals reflect a shift in valuation of woody plants in drylands, which have historically been considered economically detrimental to livestock production (Branscomb 1956).

## Changes in Inorganic C Storage

In the previous calculations we assume no change in pools of soil carbonate over the last 100 years. However, estimates of total C storage must incorporate potential changes in inorganic soil C pools. Net flux of soil carbonate in mesquite dune systems reflects a balance between erosional loss in interdunal soils and redeposition within accumulating dunal sands. A rough calculation of inorganic C loss can be derived from soil erosion estimates in the Jornada Basin. Gibbens et al. (1983) report a 3.4-cm loss of soil along a historical grassland/shrub land ecotone over a 59-year period. Initial calculations using reported soil parameters (Gile and Grossman 1979; Virginia and Jarrell 1983) suggest that soil carbonate decreased by 194 g C $m^{-2}$ along the ecotone. Assuming that pedogenic carbonate forms at a mean rate of 0.95 g C $m^{-2}$ $yr^{-1}$ over the same period (Gile and Grossman 1979), approximately 56 g C $m^{-2}$ $yr^{-1}$ accumulated for a total loss of 138 g C $m^{-2}$ $yr^{-1}$. While this estimate suggests total soil inorganic C decreased by erosion during desertification, it fails to define internal redistribution of soil C within the basin area.

Potential For Carbon Sequestration In Drylands

Dryland ecosystems typically contain small amounts of soil organic matter (Schlesinger 1982; Post et al. 1982), reflecting limited plant production in arid climates and rapid rates of decomposition that are experienced during short periods of moisture availability. Schlesinger (1990) estimated that mean global storage of recalcitrant humic substances in soils is ~ 0.4 Pg C $yr^{-1}$ , less than 1% of terrestrial NPP. Long-term storage of soil organic C is largely attributed to this recalcitrant fraction (Spycher et al. 1983). As a result, Schlesinger (1990) concluded ". if the terrestrial biosphere is indeed to act as a carbon sink under future elevated levels of carbon dioxide, this would more likely be the result of changes in the distribution and biomass of terrestrial vegetation than of changes in the accumulation of soil organic matter."

Global considerations of the changing C budget of drylands must focus on the area of these ecosystems that is most subject to change. For example, the Sahara desert makes a large contribution to any global compilation of the area of terrestrial ecosystems, but the vast majority of its extent is not likely to show changes in C storage in any likely scenarios of climate change or human land use over the next century. Thus, we should focus on the areas at the margins of deserts, where human use or abuse of the land may increase or decrease the storage of carbon in dryland ecosystems.

Stewart (1995) recently concluded that "the potential for sequestering carbon decreases as annual precipitation amounts decrease, and generally as mean temperatures increase." One might expect soil organic contents to increase with irrigated cultivation, similar to natural gradients in precipitation; however, most studies indicate otherwise (Table 5). Apparently, cultivation promotes erosional soil loss and reduces annual litter production, while stimulating decomposition due to increased soil temperature, aeration, and moisture (Schlesinger 1986).

Soil organic matter typically accumulates at rates up to 100 g C-$m^2yr^{-1}$ when vegetation first invades areas of bare mineral soil (Schlesinger 1990); typically, this rate slows with time. If an initial rate of 100 g C-$m^2yr^{-1}$ could be established over the 330 x $10^6$ ha of desertified drylands that are thought to have the potential for reforestation (Williams 1994), 0.33 x $101^5$ g C yr-1 might be sequestered from the atmosphere -- 5.5% of the annual worldwide fossil fuel emission. Thus, it would seem that management of soil organic matter in dryland ecosystems is of rather limited potential as a sink for atmospheric $CO^2$

.

Carbonates are a large but sluggish pool in the global carbon cycle (Schlesinger 1995) and humans are more likely to increase or decrease the carbon budget of drylands through their effects on vegetation and soil organic matter. Several studies have addressed soil carbonate flux resulting from irrigated cultivation. Magaritz and Amiel (1981) reported a 6 kg C $m^{-2}$ loss from dissolution of carbonate in calcareous soils irrigated over a 40-year period. In noncalcareous soils, application of salt-rich irrigation waters may favor carbonate precipitation (Schlesinger 1986). For example, Schlesinger (1986) found an increase in carbonate C of ~ 20-30 kg C $m^{-2}$ in cultivated soils derived from noncalcareous parent material (relative to undisturbed profiles). The increase in soil carbonate that sometimes accompanies dryland cultivation stems from high $Ca_2$+ contents of irrigation water, and is associated with release of $CO_2$ to the atmosphere, viz. $Ca_2$+ +2$HCO_3$- « $CaCO_3$ + $H_2O$ + $CO_2$

Thus it is apparent that irrigation alone cannot be expected to increase carbon storage in soils of dryland ecosystems.

## Conclusion

In the Sahel, increases in the area of deserts that can be linked to recent monotonic changes in climate appear small (Tucker et al. 1991, 1994). Desertification appears to be primarily due to direct human land uses. Similarly, in south-central New Mexico (USA), losses of productive grasslands in favor of desert shrublands appear unrelated to climatic fluctuations during the last century (Conley et al. 1992). Regional semiarid/arid land ecotones are particularly sensitive to anthropogenic disturbances (Schlesinger et al. 1990; Risser 1995), and global climate change may severely alter their ability to store C (OIES 1991). Thus, while remaining alert to the potential for global warming to increase the area of deserts (Rind et al. 1990), we should focus on lands that are dominated by human management (UNEP 1992).

The Jornada desertification model defines dryland degradation in terms of increasing spatial and temporal heterogeneity of both soil resources and biomass resulting from biological feedbacks associated with climate and human activities (Schlesinger et al. 1990). Redistribution of these properties perpetuates shrub establishment and the desertification process. The overall result is a permanent change in the productive capacity of the landscape.
Although productivity is closely linked to fluxes in organic C, few studies address changes in soil C resulting from desertification. Long-term studies at the Jornada Basin suggest that while mesquite communities gain biomass C, mesquite and creosote-bush communities lose soil organic C (due to erosion) relative to grasses. Tarbush communities appear to accumulate soil C; however, these sites probably contained more C (prior to shrub establishment) than modern grasslands. As a result, there has been little overall change in total C within the basin. Unique quantification of inorganic C flux during desertification has not yet been determined.

Analysis of soils across different geomorphic surfaces in the Jornada Basin indicates that moisture/temperature gradients are the primary determinants of C storage capacity in this system. These results suggest that agricultural techniques such as irrigation may enhance C sequestration in soils. However, the Jornada desertification model illustrates that C gains will only be realized if losses due to soil erosion are prevented and C costs related to energy use remain negligible. Glenn et al. (1993) recently demonstrated the economic efficacy of producing halophyte biomass to sequester C. However, such agricultural activities may accelerate soil erosion, promoting losses of C from surface litter and soil organic matter. The net global C sink produced by halophyte biomass production or other afforestation/reforestation programs comprises only a trivial fraction of annual fossil C emissions. As a result, we believe little benefit will be provided by efforts specifically directed at mitigating rising atmospheric $CO_2$ concentration through biomass production.
The economic costs of desertification are substantial. UNEP (1991) estimated that annual income lost to degraded rangelands is approximately 23x109 ($USA) $yr^{-1}$. While the cost-benefit ratio of restoring degraded drylands is 1:3.5 on a global basis, these costs increase with the degree of degradation (UNEP 1991). Measures to prevent desertification are less costly than corrective practices or rehabilitation projects. While research to mitigate rising atmospheric $CO_2$ concentrations through manipulation of biomass is warranted, we believe greater effort should be allocated to promoting sustainable use of these ecosystems.

**Acknowledgments.** This work was supported by the U.S. National Science Foundation and is a contribution to the Jornada Long-Term Ecological Research Program.

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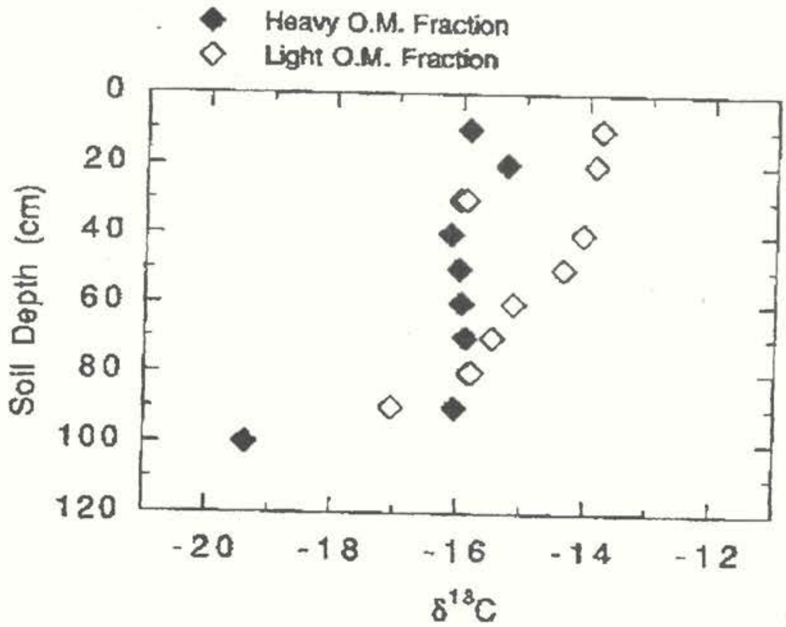
Figure 1. d13C profiles for two fractions of soil organic matter in a grassland soil at the Jornada Experimental Range (from previously unpublished data). Open and shaded diamonds represent light and heavy fractions, respectively.



# Carbon Sequestration in Some Natural, Managed and Degraded Ecosystems of Africa

Paul L. Woomer

Synopsis

This paper documents the carbon storage and dynamics as selected African woodland ecosystems are converted to smallholder agriculture. A Case study approach is used to examine smallholder cropping practices as either contributing toward or ameliorating that decline in system carbon. A smallholder scenario is presented in which farm practices that simultaneously contribute toward farm livelihood, farm resource conservation and system carbon storage are introduced to a degraded farming system.

This approach is applied to three land conversion processes in sub-saharan Africa; coastal dune forests in southern Mozambique converted to short-lived mixed cropping systems by shifting cultivators, maize-based cropping systems derived from miombo woodland by communal farmers in Zimbabwe and the slash-and-burn agriculture practised in the humid forests of southern Cameroon.

Key Points

1. Total carbon storage is reduced as natural habitats become converted to agriculture. For a coastal dune forest in Mozambique, that was converted to mixed agriculture by slash and burn, the loss of carbon from the system was estimated to be 33 T C/ha after three years (from 58 to 15 T C/ha, respectively). When a miombo woodland was compared to an adjacent maize field, in Zimbabwe, the total system carbon was reduced by 50% (from 56 to 28 T C/ha), most of which resulted from biomass removal. In a land use chronosequence in Southern Cameroon, total system C was reduced from 282 T C/ha in the original forest to 57 T C/ha following slash and burn but these lands recovered quickly with 67% of the original C level being reaccumulated in a 20-yr old fallow.
2. When new lands are cleared for agriculture, some disruption of the natural ecosystem is inevitable. Most field and cash crops require full sunlight and cannot be productively cultivated under closed tree canopies. After trees are felled by farmers, the sheer total amount of litter lends burning as an attractive option both for improved of field access and nutrient recycling.
3. Declines in carbon resulting from smallholder farming results in part from a conflict between the basic farmers'objectives of securing a livelihood and the environmental protection concerns of society at large. At the same time, other farm improvements such as organic resource management, agroforestry and conservation tillage, are compatible with both farmer short-term objectives, farm resource conservation and longer- term C sequestration.
4. It is possible to develop farming systems (a stepwise sequence of planned farm changes) which simultaneously sequester C and offer economic return to a smallhold farmer, C stocks may be increased by 50% due largely to benefits derived from increased tree biomass and soil erosion control.

Key words: slash and burn agriculture, Cameroon, Mozambique, Zimbabwe, smallholder , tree canopies, cash crops, nutrient cycling, sub-Saharan

## Introduction

Total system carbon is most usually lost from natural ecosystems as humans convert tropical humid forest (Ayanaba, 1976; Bouwman, 1989), dryland forest (Araki, 1993) and savannas (Tiesen *et al.*, 1992) to agriculture. The loss of carbon is primarily to the atmosphere through burning or accelerated decomposition and is one of the driving forces in global atmospheric change, particularly the increase in $CO_2$ (Palm et al, 1986; Hall, 1989) second only to that emanated from the oxidation of fossil fuels (Detwiler, 1986). The reduction in system carbon results from a series of farmer practices. First, natural vegetation is removed, often by felling and or burning (Nye and Greenland, 1960). The crops that are established seldom contain biomass equal to that of the original vegetation and in the case of annual field crops the land may be devoid of vegetation cover during parts of the year. Tillage operations accelerate the decomposition of soil organic matter (Tisdale and Oades, 1982). Reduced litter and root inputs to the soil are unable to offset organic carbon losses. The paucity of vegetation and litter cover predispose soil to accelerated erosion causing additional organic matter to be lost from the system (Dulal *et al.*, 1986; Estwaran, 1994). Pessimism concerning the effects of land management on carbon sequestration becomes too frequently justified as one system after another is studied in sub-saharan Africa (Woomer, 1993) and elsewhere in the tropics.

While the trend toward reduced carbon storage in managed ecosystems is near universal, the magnitude of that loss is largely under the control of farmers and other land managers. In this paper, I will document the carbon storage and dynamics as selected ecosystems are converted to smallholder agriculture. Then I will examine smallholder cropping practices as either contributing toward or ameliorating that decline in system carbon. Lastly, a smallhold scenario is presented in which farm practices that simultaneously contribute toward farm livelihood, farm resource conservation and system carbon storage are introduced to a degraded farming system.

## Carbon Sequestration Case Studies

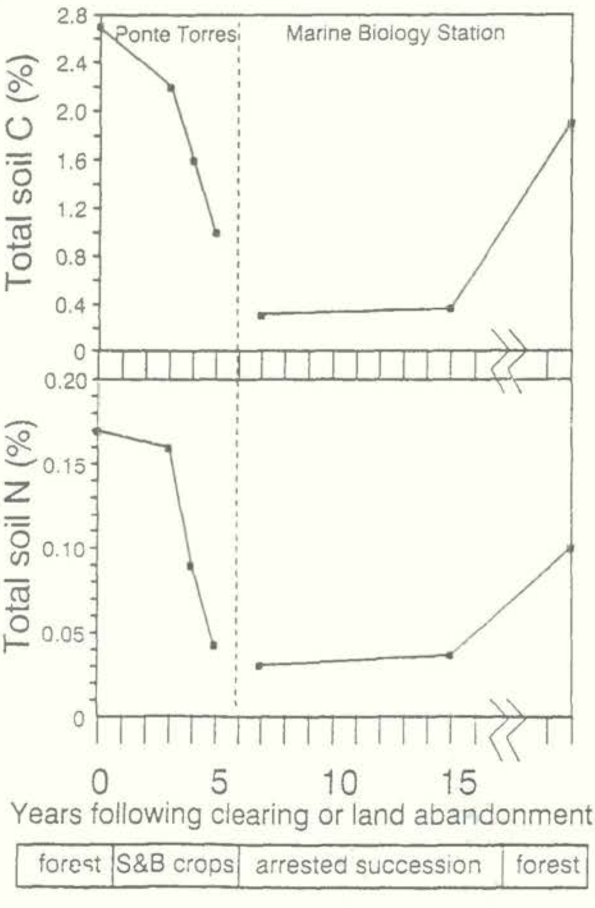
The Tropical Soil Biology and Fertility Programme (TSBF) is a voluntary, participatory research organisation with membership based upon an interest in examining the role of biological processes in soil fertility (Woomer and Swift, 1995) and a commitment to undertake a standardised site characterisation (Swift, 1986) using common methods (Anderson and Ingram, 1993). One approach to understanding carbon and nutrient dynamics is to compare adjacent, contrasting land uses on similar soils. This approach is essentially a type I study identified by Sanchez *et al.* (1985) where space is substituted for time in order to characterise chronosequential events.

In the following discussion, I will apply this approach to three land conversion processes in sub-saharan Africa; coastal dune forests in southern Mozambique converted to short-lived mixed cropping systems by shifting cultivators, maize-based cropping systems derived from miombo woodland by communal farmers in Zimbabwe and the slash-and-burn agriculture practised in the humid forests of southern Cameroon.

*Coastal Mozambique.* Slash and burn farmers, many of them internal refugees from civil unrest in the recent past, are deforesting the coastal forests in southern Mozambique. These forests have formed on sand dune soils of very low inherent fertility. The forest is dominated by trees belonging to the family *Rubiaceae*. Estimates of forest aboveground biomass are 80 T/ha (John Hatton, personal communication). King (1994) sampled sites on Inhaca Island in Maputo Bay. These consisted of climax dune forest and adjacent one, two and three year-old mixed cultivation fields at Ponte Torres and slowly regenerating secondary vegetation near the Marine Biology Research Station of Eduardo Mondlane University (Woomer and Swift, 1995).

The two research areas are separated by approximately 8 km and both contain deep silica sand soils. Figure 1 illustrates the dynamics of soil organic carbon and total nitrogen in the soils during different stages of shifting cultivation. Soil organic carbon declines rapidly following land clearing and burning from an initial content of 2.7% to 0.9% within three years. After cropping, abandoned, degraded lands were lower in soil organic matter and slow to recover their vegetation cover and soil organic matter contents. Dynamics in total soil nitrogen closely reflect those of carbon. When assumptions are made concerning the bulk density of sand, and the biomass of crop yields prior to land abandonment, a preliminary carbon budget for the forest and adjacent cropping systems may be calculated (Table 1).

Figure 1. Soil organic carbon and total soil nitrogen dynamics in a coastal dune forest converted to mixed agriculture on Inhaca Island, Mozambique (Serra-King, 1995).



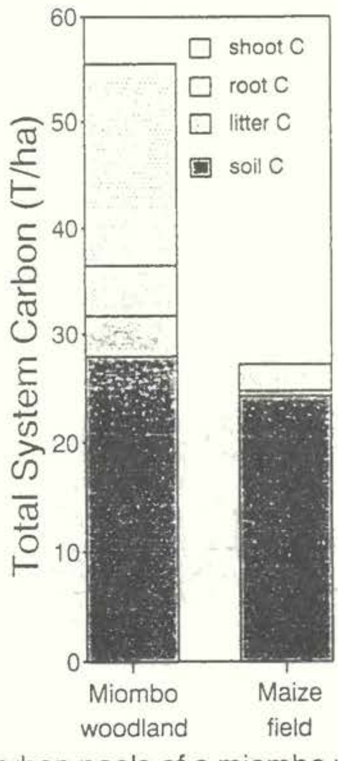
*Miombo Woodlands.* The miombo woodlands of Southern Africa represent the largest, relatively undisturbed zone of natural vegetation remaining in Africa extending from Tanzania and Mozambique, through Zimbabwe and Zambia into Angola and Zaire. Miombo woodlands are dominated by leguminous tree species belonging to the genera *Brachystegia* and *Julbernardia* often with grass understorey vegetation, and have open canopies that shed their leaves during the dry season.

Table 1. Preliminary estimate of carbon dynamics resulting from slash-and-burn agriculture in coastal Mozambique.

| carbon pool | land use | |
|---|---|---|
| | dune forest | mixed cropland |
| | T C/ha | |
| surface soil & litter | 22 | 12 |
| aboveground vegetation | 36 | 3 |

These deciduous woodlands are under continuous conversion by ranches, commercial farmers and smallholders. In Zimbabwe, a natural woodland and adjacent cropping systems are under study by researchers at the University of Zimbabwe and Soil Productivity Research Laboratory. The differences in carbon storage between the woodland and a maize-based cropping system is presented in Figure 2. Note that the largest decline in carbon storage results from vegetation removal and that the soil carbon has not become greatly reduced.

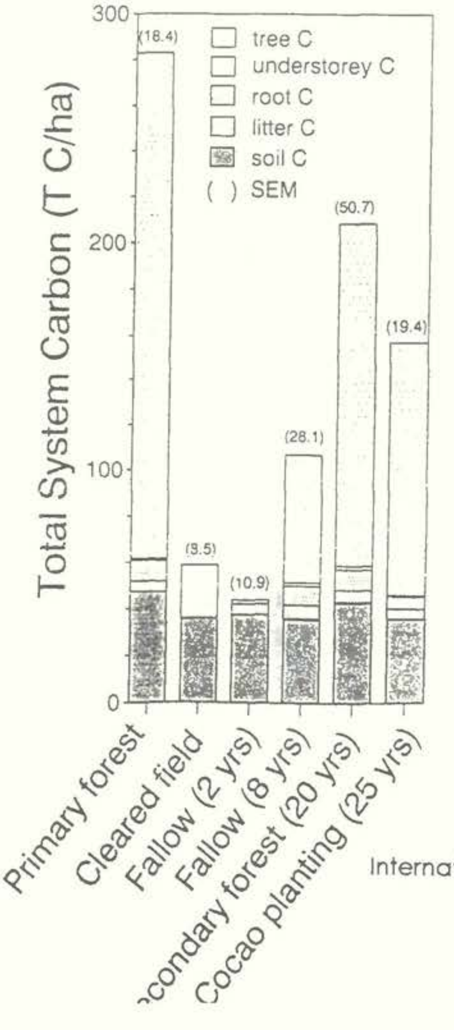
Figure 2. Total system carbon in a miombo woodland and a six year-old adjacent maize field at Marondera, Zimbabwe.




Carbon pools of a miombo woodland converted to maize production

*Southern Cameroon*. Smallholder slash-and-burn agriculture has been widely practised in the lowland humid forests of Africa. The Alternatives to Slash-and-Burn Program (Bandy, 1994) has identified the area south of Ebolowa, Cameroon as representative of larger areas of equatorial Africa and is focusing efforts on two villages there, Makoe and Mengomo. TSBF is working with scientists at the Institute Research Agriculture (IRA) to characterise the carbon and nutrient dynamics in adjacent land uses in Ebolowa. These land uses include primary forest, land cleared for maize-groundnut production, fallows of various ages, secondary forest and cacao plantings under secondary forest (Figure 3). The carbon storage in the humid forest is much greater than that of the dune forest or miombo woodland (284 vs 58 and 56 T C/ha, respectively) and a greater proportion is contained in vegetation (85% vs 62% and 49%, respectively. Carbon loss is greatest immediately following land conversion and continues into the young fallow. Logging did not take place prior to land clearing and many farmers do not fell all large trees during the first year of cultivation, hence the tree biomass in the "cleared" field. Farmers often fell and burn the remaining trees prior to the second year of cropping and then are forced to abandon the lands due to competition from weeds particularly *Chromolina odorata*. After the short cropping interval (2 years), the fallow and secondary forest recover quickly with greater than 67% recovery of total system carbon, compared to the original forest, occurring in the 20 year-old secondary forest. Also note that cacao well established within a secondary forest contains approximately one half the total system carbon of the original forest and 2.5-fold more than lands cleared for field crops.

Figure 3. The storage of carbon in different land uses resulting from slash-and-burn agriculture in the humid forests of southern Cameroon.



Smallhold Farmers and Carbon Sequestration

The smallhold farmers objective is not to sequester carbon but rather to provide for their families needs and to achieve improvement in their standards of living. On the other hand, there is an important societal concern for environmental protection, some of which, particularly maintenance of biodiversity and atmospheric change, are directly impacted by farmer's practices. As a result of these differences, securing of farm livelihood vs conservation of environmental resources, there a both points of divergence and convergence between smallholder agricultural practices and carbon sequestration.

*Points of Divergence.* When new lands are cleared for agriculture, some disruption of the natural ecosystem is inevitable. Most field and cash crops require full sunlight and cannot be productively cultivated under closed tree canopies. After trees are felled by farmers, the sheer total amount of litter lends burning as an attractive option both for improved of field access and nutrient recycling.

The loss of carbon from soils and organic inputs is a necessitated when these farmer resources are exploited by farmers. Nutrient mineralisation and decomposition are overlapping and often identical processes (see Woomer et al., 1994). Many nutrients are mineralised through the assimilation of organic substrate by soil organisms and through their eventual death. Furthermore, soil tillage incorporates surface litter into soil and disrupts aggregates which physically protect soil organic matter (Tisdale and Oades, 1982).

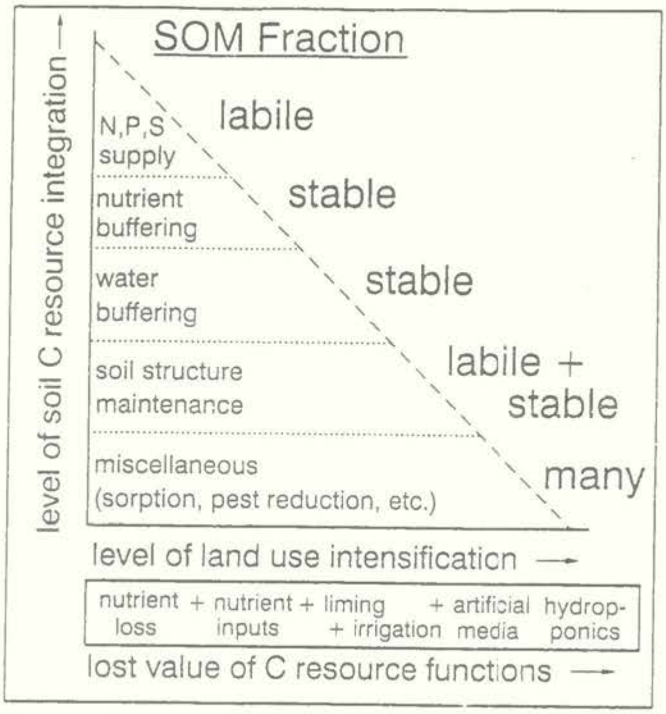
In many cases, soil erosion may result in increased carbon sequestration as sedimentation tends to place greater amounts of SOM into anaerobic conditions in which decompositional processes are reduced (van Noordwijk et al., in press). The "protection" of organic matter undergoing sedimentation holds true on watershed or larger scales but soil loss acts to the detriment of individual farming systems.

*Points of Convergence.* Organic matter improves soil properties including nutrient and water retention, immobilisation of toxic cations and aggregate stabilisation (see Woomer et al., 1994). Improved soil offers returns to farmers during periods favorable to crop growth and reduces risks during periods of plant stress. Furthermore, the roles occupied by soil organic matter substitute for external inputs that would otherwise require investment by smallhold farmers (Figure 4) with limited cash reserves and availability to credit. Many perennial crops have root systems more expansive or deeper that allow for improved nutrient cycling (Young, 1989).

Increased reliance upon perennial plants provides opportunity to diversify farming operations as a mechanism for risk reduction and to broaden the variety of farm products. Tree crops include those which provide fruit and nuts, wood and structural materials, fuel and livestock feed (ICRAF, 1992). Other perennial plants provide indirect benefits to the farm. The use of live fences on farm boundaries or grass strips along the slope contour can contain or exclude livestock, deter intruders and reduce soil erosion (Young, 1989).

Conservation tillage and retention of crop residues have the potential to simplify farm operations, improve crop yields and reduce or reverse declines in soil organic carbon. Conservation, or minimum, tillage as practised on mechanised farms of temperate North America places emphasis on straw and stubble management to protect the soil surface. Emergent weeds are usually controlled by herbicides rather than shallow tillage. This land management strategy has been demonstrated to increase soil organic matter with time, alter the community composition of soil organisms (Beare *et al.*, 1994) and to reduce soil erosion. While it may be argued that this form of conservation tillage has small application to smallholders who rely on hand labor and draft animals, other areas of the tropics, particularly in South America, routinely avoid soil tillage although often in conjunction with annual burning. Another complicating factor in the use of crop residues as soil amendments is the use or sale of these organic resources as a livestock feed or fuel.

Figure 4. Soil organic matter occupies different functional roles in the maintenance of plant growth which operate at different levels of land use intensification (after van Noordwijk *et al.*, in press).




## Planned Farm Changes

The loss of total system carbon resulting from conversion of natural ecosystems to agriculture by smallholder farmers was illustrated in the carbon dynamics case studies earlier in this paper. These losses mainly result from the discrepancies between farmer objectives and carbon storage on the lands these farmers manage. Yet the potential for increased carbon stocks resulting from more compatible farm activities require greater exploration. In this section I prepare a stepwise "best case" scenario that compares carbon accumulation as a smallholding becomes more diversified from annual field cropping of cereal grains to one with live fences, contour strips, woodlots and orchards.

*Initial conditions.* An idealised smallholding is illustrated in Figure 5. This two hectare farm consists of fields where annual crops are grown as intercrops, boundary hedgerows of indigenous and naturalised shrubs and a homestead with surrounding home gardens and trees in the near fields. The home consists of thin poles patched together with mud, dirt floor and thatch roof. The only other farm structure is a livestock enclosure containing two cattle. No attempts are made to reduce soil erosion and no crop residues are retained in the field. Manure from the livestock, approximately 1200 kg/yr, is returned to the home garden and immediately adjacent fields. The total system carbon, including surface and sub-surface soils to a depth of 50 cm is 52 T C/ha.

The initial farm consists of two ha with a surface soil (0-20 cm) organic carbon content of 1% and a bulk density of 1.3 g/cm$^3$. The subsoil (20-50 cm) organic C is 0.5% with a bulk density of 1.4 g/cm$^3$. A maize bean intercrop is grown each year with yields of 1200 kg/ha, the harvest index is 32% and roots represent 20% of the aboveground biomass. All biomass is 45% carbon. The farm is surrounded by a naturalised hedgerow 600 m in length with a biomass of 4 kg m$^2$. The house consists of 120 three meter poles 8 mm in diameter with a density of 0.4 g/cm$^3$. Results in Figure 4 expressed in terms of T C/ha.

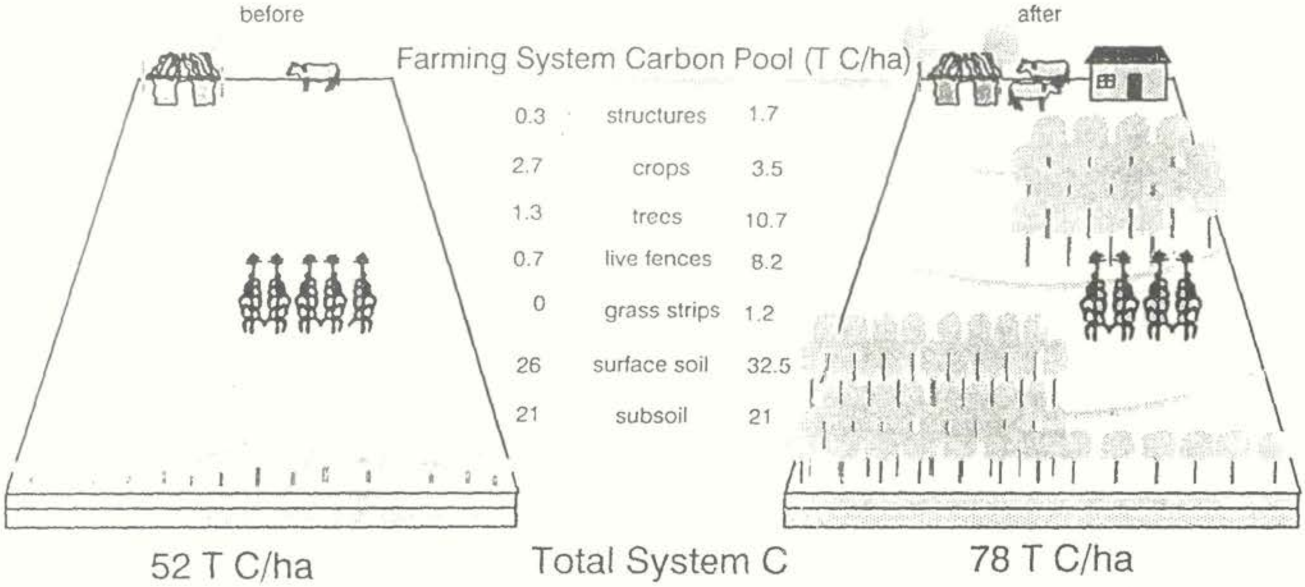
*Stepwise changes.* The right hand side of Figure 5 illustrates the carbon storage resulting after several years of changed farm practices. First, 1 m grass strips are established at 15 m intervals along the slope contour. Once established, these grass strips are periodically trimmed for use as livestock feed thus reducing dependence upon crop residues for the same purpose. While the carbon contribution from these grass strips is slight, reduction of soil erosion, in conjunction with other soil conservation practices, allow for increases in total soil organic carbon.

Next, the boundary hedgerows are interspersed with tree species that are subsequently used as soil organic inputs, animal feed and structural materials. A 1250 m$^2$ woodlot is established on the lowest field along the slope, and a fruit orchard of the same size established near the house. Crop yields on the remaining fields are increased 50% through the use of improved crop varieties and hybrid seed, increased addition of organic inputs including crop residues, manure and hedgerow prunings on the remaining cropland, and when fertilisers applied in a strategic fashion.

As a result of increased productivity, the farm family is able to double the number of livestock and build a larger, wood framed, floored and sided house. After several years the trees comprising the live fence, orchard and woodlot have achieved a girth of 20 cm at breast height and contribute 16.9 T C/ha while occupying 2900 m$^2$ (14.5% farm area). As a result of these planned changes, the two ha farm now contains 78 T C/ha, an increase of 60% when compared to the initial condition.

Farm changes are based upon the following assumptions. Grass strips are one meter wide, placed at 15 m intervals and contain 4 kg m$^2$. Trees planted in the live fence are spaced 1 m apart, in the orchards at 50 m$^2$/tree spacing and in the woodlot at 16 m$^2$/tree. All trees have grown to 20 cm diameter at breast height and biomass is calculated using the allometric equations of Brown *et al.* (1989). Crop yield are estimated at 2400 and 1200 kg/ha season, respectively, using the same components of yield as in the initial farm but on 0.125 less land area, that occupied by trees. The new house consists of 10.6 m3 of wood with a density of 0.4 g/cm$^3$ and C content of 45%.

Figure 5. A scenario for increased carbon storage on an agricultural smallholding.




## Combatting Desertification of Dryland Agriculture

Desertification is defined by the United Nations Environment Program as " Land degradation in arid, semiarid and dry subhumid areas resulting mainly from adverse human impact " (UNEP, 1992). Dryland areas cover 1282.5 x $10^6$ ha of Africa excluding the 672 x $10^6$ ha hyperarid zone out of a continental total of 4256 x $10^6$ ha (UNEP, 1992). Also in Africa, 33.3% of the population inhabit the dry-subhumid and semiarid drylands. It is the agriculturalists in these area that the approach presented in this paper has most relevance. To condemn these smallholder farmers, and their management practices, for devegetation and desertification is to ignore their daily struggle under changing climatic conditions. To imagine smallholder farmers as an army in the battle against desertification is also unthoughtful without compensation to smallholders given their situations. Desertification is in fact an absence of vegetation that would otherwise sequester biomass carbon and accumulate soil organic matter, how then is desertification, and climate change, to be confronted. We have Desertification Atlases to assist us target projects, International Agricultural Research Institutes able to recommend germplasm and International Agreements authorising us to proceed. What next?

The least degraded lands have been demonstrated as those that can be best revegetated in the most economic fashion (UNEP, 1992). I recommend that the smallholder agricultural sector in areas at risk of desertification be provided with the tools they require to maintain and re-establish vegetation shelter belts, windbreaks and grass strips. Let innovative water harvesting and zero runoff techniques move to the forefront, whether these originate from local communities or research institutes. Let us build tree seedling nurseries on sufficiently large scales to enter the economics of scale. Let us distribute tree seedlings that are healthy, inoculated with microsymbionts, hardened and with reserves of slowly-released, otherwise limiting nutrients in their rooting media at no cost to farmers in return for guarantees of their husbandry and protection from grazing animals. We must procure and distribute massive quantities of drought tolerant grasses and shrubs for planting adjacent to smallholdings, again at no financial cost to farmers in return for their husbandry. And then let us hope for rain before moving forward into the next most severely desertified areas.

## Acknowledgments

The results presented for the carbon storage in Africa ecosystems we used through the courtesy of network scientists participating with TSBF, specifically Alexander Serra King and J Hatton from Eduardo Mondlane University, Maputo, Mozambique; Peter Frost and Linus Mukurimbura in Zimbabwe; and Jean Kotto-Same with the Institut Research Agronomique (IRA), Nikabbissan, Cameroon. The collaboration of these scientists is greatly appreciated as part of the TSBF African Network (AfNet) and Alternatives to Slash & Burn Program (ASB). AfNet is funded through grant provided by the Rockfeller Foundation and ASB is funded by UNDP and the Global Environmental Facility. These donor organization are also appreciatively acknowledged.

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# Management options to increase carbon storage in cultivated dryland soils

Peter Grace, Jeff Ladd and Kim Bryceson

Synopsis

This paper presents projected potential long-term changes in carbon storage in marginal drylands in southern Australia as a function of crop rotation, soil type and climate as predicted by the SOCRATES model.

SOCRATES (SOIL ORGANIC CARBON RESERVES AND TRANSFORMATIONS IN AGROECOSYSTEMS) is simulation model designed to estimate changes in surface soil organic C as influenced by crop and pasture rotations, nitrogen fertiliser addition, disease, grazing intensity and climate. SOCRATES contains a simplistic plant growth model which is essentially a means of producing either leguminous or non leguminous dry matter.

**Key Points**

1. The abundance and formation of soil organic matter in dryland agro-ecosystems has, within the last decade, assumed increasing importance within the context of resource sustainability and the separate, but related, issue of global climate change. Cropping systems, in particular, have come under close scrutiny as both the world's human population and production costs for major food grains have increased.

2. Cropping of dryland soils in marginal regions and an emphasis on economic, rather than ecological sustainability, has generally led to a decline in soil organic carbon stores in these soils. Soil structure cannot be maintained, infiltration is reduced, biological activity severely affected, leading to a vicious cycle of events usually culminating in excessive soil loss and, ultimately, desertification.

3. In semi-arid regions, where dryland farming is practiced, changes in soil properties resulting from agricultural and pastoral activities have been extreme. Declines in soil organic matter are widespread with a concomitant decline in soil structure . Soil crusting, compaction and surface runoff are an increasing problem . As infiltration rates decline soil and water erosion increases and serious land degradation follows.

4. Reducing the incidence of dry fallows, increased crop residue retention, nitrogen fertiliser applications and a shift to mixed cereal-legume based passtures (as opposed to monoculture) can slow the spiral toward land degradation.

5. Cultivation of cropped and native soils for extended periods invariably leads to a lowering of their total organic carbon content. This reduction in carbon is usually rapid after the initial cultivation event, particularly when there is disturbance of native grassland or woodland soils. the higher organic matter soils tend to lose a greater proportion of organic carbon compared to soils with low initial levels.

**Key words: dryland cropping, conservation tillage, no till, crop residue retention**

## Introduction

The abundance and formation of soil organic matter in dryland agro-ecosystems has within the last decade assumed increasing importance within the context of resource sustainability and the separate, but related issue of global climate change. Cropping systems in particular have come under close scrutiny as both the world population and production costs increase. The total area of rainfed croplands in drylands has been estimated as 457 million hectares, which is approximately 10 % of the world's dryland as classified by UNEP (1992). Of this area, nearly half is listed as being degraded to some extent, with an additional 4 million hectares of rainfed croplands being lost each year in the world's drylands, either as a result of erosion or urbanization.

In Australia alone, where the majority of cropping is found in the semi-arid region, changes to soil properties resulting from agricultural and pastoral activities have been extreme. Declines in soil organic carbon are widespread with a concomitant decline in soil structure showing soil crusting, compaction and surface runoff to be an increasing problem and infiltration rates decreasing (Chartres *et al.*, 1992) increasing soil, wind and water erosion and the potential for desertification. It has been estimated that up to 39% of organic C in cultivated surface soils of Australia has been lost between 1860 and 1990 (Gifford *et al.*, 1990). Globally, soil C losses as a result of deforestation, shifting cultivation and arable cropping have made significant contributions to increased levels of atmospheric $CO_2$ (Post *et al.*, 1990). Man-induced change for that matter must now be considered a significant, or for that fact, co-dominant influence with climate on global agricultural production.

## Environmental influences

A knowledge of the influence that inherent soil properties such as texture have on carbon turnover and the stabilization of organic inputs is essential in determining a soil's ability to be a net sink or source for C. An increase in the stabilization of organic matter and a reduction in the rate of decomposition is a feature of soils of high clay content (Amato and Ladd, 1992; Sorensen, 1983). High $CaCO_3$ contents have also been found to promote the stabilization of relatively undecomposed plant materials (Duchaufour, 1976). Abiotic influences on soil organic matter dynamics, such as moisture, temperature, aeration and composition of plant residues are reasonably well understood (Paul, 1984), however in dryland environments, superimposed on these properties and processes is the fact that most soils are deficient in both nitrogen and phosphorus. This is a direct result of inputs not meeting outputs, reduced ground cover and increased losses of nutrients by erosion in this environment. This is a vicious circle of events which is difficult to reverse without a dynamic and informed management approach (as opposed to hapahazard or historical decision making).

## Management effects

Cultivation of cropped and native soils for extended periods invariably leads to a lowering of their total organic C content. This reduction in C is usually rapid after the initial cultivation event, particularly when there is disturbance of native grassland or woodland soils. The higher organic matter soils also tend to lose a greater proportion of organic C compared to soils with low levels (Mann, 1986), which is consistent with substrate depletion as described by first order kinetics. The extent of this degradation is usually dependent on tillage intensity, residue composition and placement and fertilizer inputs.

Reduced and no-tillage management tends to concentrate crop residues and associated organisms in the surface layer (Doran, 1980) resulting in distinct profile stratifications which may greatly improve surface soil aggregation, increasing infiltration rates and water use efficiency, a critical factor in dryland cropping. Conventional cultivation techniques and the use of high intensity tillage operations such as moldboard and chisel ploughing generally causes a greater reduction in carbon stocks compared to reduced or no-till systems (Agenbag and Maree, 1989; Havlin *et al.*, 1990; White, 1990), however this is not always the case in drylands with some detailed studies by Roget and Rovira (unpublished) showing the carbon benefits in no-till systems to be very transient (Figure 1). This is indirectly supported by Kirkegaard (1995) who found no conclusive evidence of yield benefit in no-till systems in a detailed survey of twenty-eight long-term cropping trials in the semi-arid regions of Australia. This suggests that the improvement in aggregate stability and infiltration usually associated with no-till agriculture actually produces a more favourable environment for microbiological activity in the longer term, increasing the rate of decomposition of organic matter and the availability of simple C substrates which have been linked to increased incidence of root pathogens.

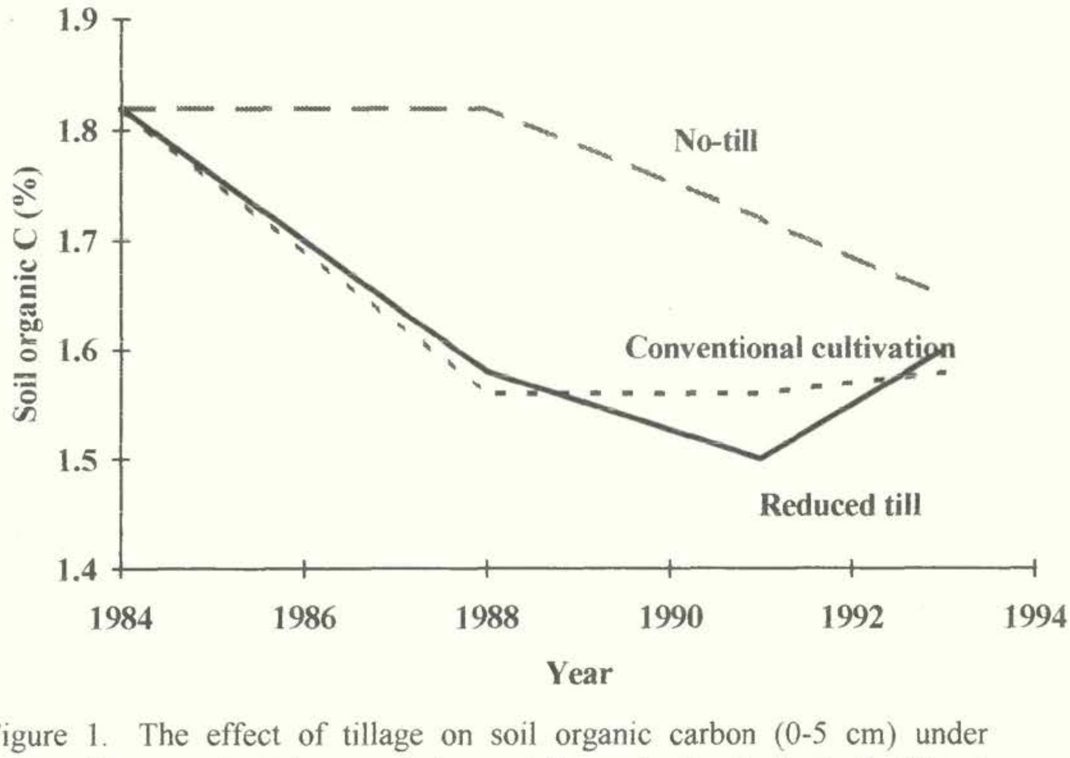


Figure 1. The effect of tillage on soil organic carbon (0-5 cm) under continuous wheat in a sandy-loam at Kapunda, South Australia (Roget and Rovira, unpublished).

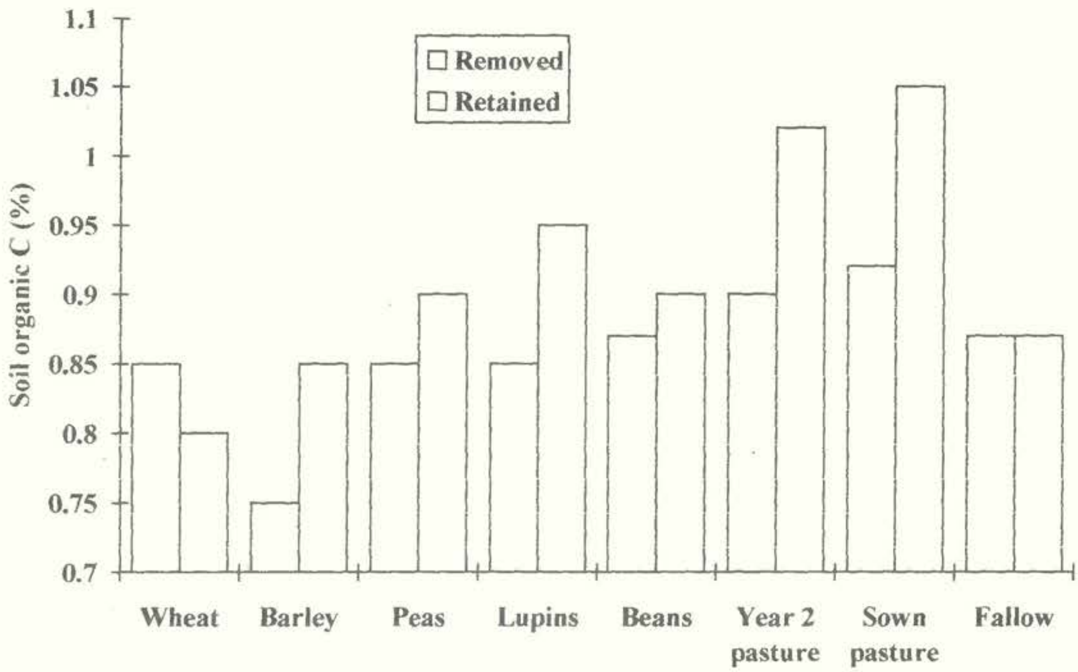


Figure 2. Soil organic C (0-10 cm) in response to residue management at the Tarlee Rotation Trial, South Australia (Schultz, unpublished).

Long-term trials have also demonstrated that the repeated application of fertilizers, particular nitrogen, frequently increases topsoil organic C stores through an increase in the amount of residue being returned (Figure 2). In many of the poor quality dryland soils in Australia, C levels have increased above those of virgin conditions once nutrient deficiencies of N, P, S, and in some cases trace elements, are overcome and yields increased (Russell and Williams, 1982). Incremental increases in soil C in concert with plant carbon increases in response to fertilizer nitrogen applications are not commonplace in low organic matter drylands, even with supplemental irrigation (Figure 3). This is compounded by the decreased capacity of the coarse textured soils, which dominant the rainfed croplands in the semi-arid regions, to retain organic matter in the same proportion as heavy textured soils (Amato and Ladd, 1992). Plant selection in the rotation sequence will also have a direct impact on residue production in subsequent years, particularly the N benefit of including grain legumes or "break" crops such as *Brassica* spp. to control cereal root diseases (Kirkegaard *et al.*, 1993).

A common management practice in dryland arable agro-ecosystems has been the use of fallowing to reduce the risk of crop failure through water stress. Whilst fallowing may increase yield and residue production, the increased water availability promotes greater activity of the soil biota. Whilst an increase in the frequency of fallow phases in the cropping sequence will accelerate the depletion of organic carbon stocks in semi-arid soils, the inclusion of sown pastures is generally associated with an accumulation (Russell and Williams, 1982) or at the very least, a decrease in the rate of decomposition of organic carbon in these soils (Figure 4). Significant increases in soil organic C have been reported in a wide range of dryland soils for wheat rotations which have included 2-4 years of pasture (Grace *et al.*, 1995; Rowland, 1980; Whitehouse and Littler, 1984). The quality, or legume content of the pasture will also have an effect on the potential for that system to continue to sequester carbonaceous material. Legume dominant pastures are preferred as the residual N made available to the next crop may boost dry matter production of subsequent crops. In contrast, immobilization of mineral N accompanying the decomposition of grass dominant pastures may lead to a reduction in crop production. Grass dominant pastures also tend to provide a suitable hosts for diseases which may effect subsequent crops e.g. the root pathogen *Gaeumannomyces graminis* var *tritici* (Ggt) (Rovira, 1994).

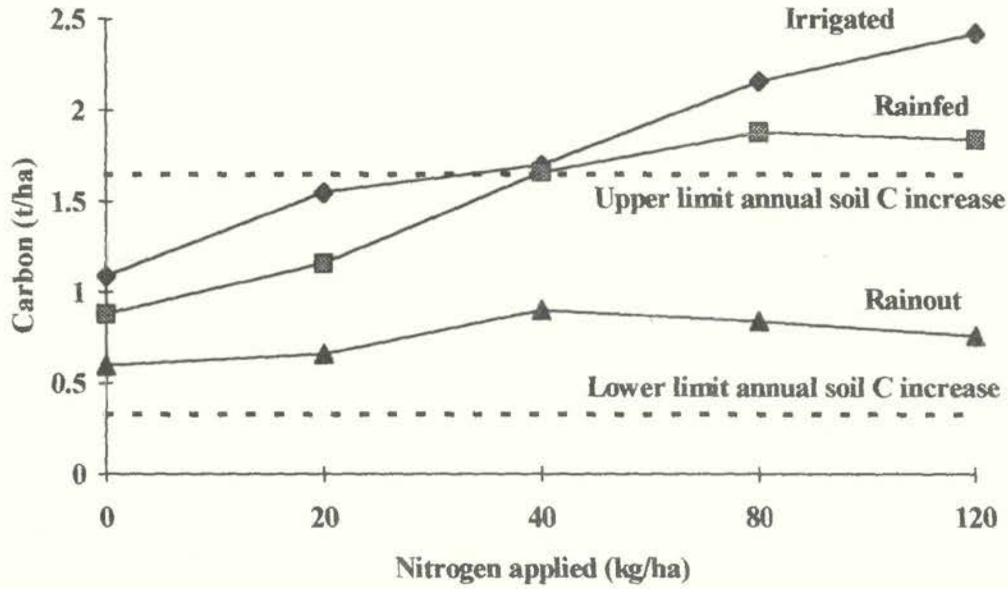


**Figure 3. Total plant carbon (at anthesis) and associated changes in soil organic carbon (0-15 cm) in response to fertilizer nitrogen addition to wheat during the 1993 growing season at Walpeup, Victoria (Kidane Georgis, unpublished).**

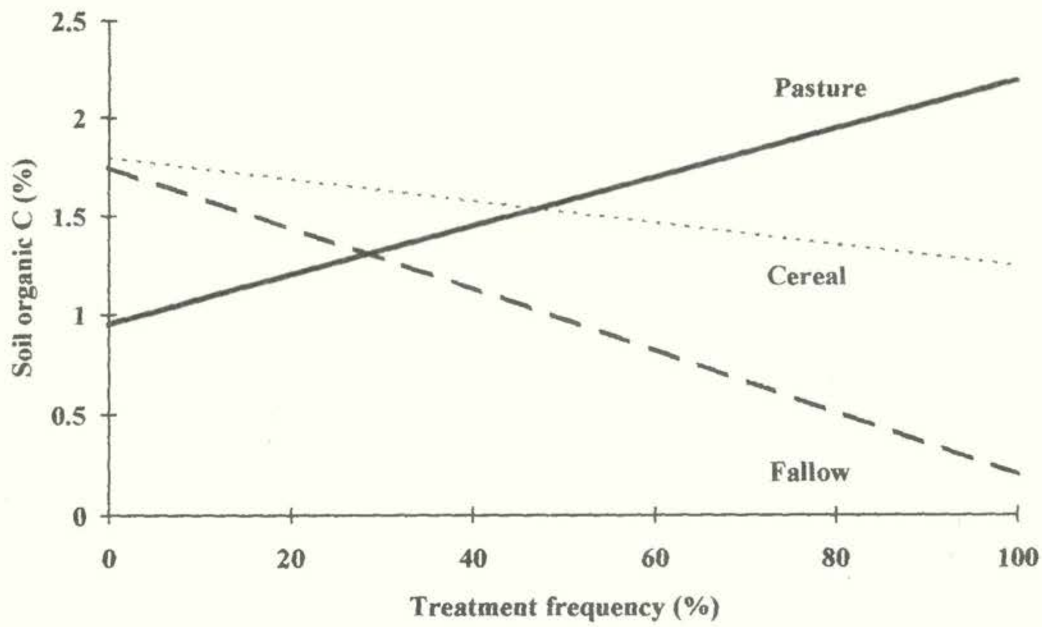


**Figure 4. The effect of long-term rotation and treatment frequency on soil organic carbon (0-10 cm) as measured in 1993 at the Permanent Rotation Trial at the Waite Agricultural Research Institute, South Australia.**

Whilst the majority of studies would indicate the success of reduced tillage and rotations in improving a soils ability to sequester C and be more productive, the exceptions rather than the norm may tell us more about the long-term consequences of our management. Strategies to improve C storage and ecological sustainability are definitely soil and environment dependent. Single indicators of soil quality or health must be considered as questionable considering the complexity of the soil-plant system. The full potential of the effects of management on C cycling in agro-ecosystems can only really be assessed in the context of whole system simulation models. These allow the integration of the many basic empiricisms describing the processes and properties in C turnover and allow feedbacks of crop production, disease and climate to be fully coupled. It is within this context of strategy development that we have produced a simple but robust simulation model encapsulating our current knowledge for promoting C sequestration and reducing degradation in semi-arid agro-ecosystems. To make the model applicable to the decision makers from the farm to regional level we have emphasized the use of easily accessible input data. Users also have access to a simple utility which will change some key parameters thus allowing a re-calibration the model to their own environment. In this paper we will briefly describe the concepts within the simulation model; produce a number of scenarios for improving C storage in rainfed drylands and describe a methodology for developing management strategies to combat degradation and sequester C at the regional level.

## Model structure

SOCRATES (Soil Organic Carbon Reserves And Transformations in agro-EcoSystems) is a simulation model designed to estimate changes in surface soil organic C as influenced by crop and pasture rotations, N fertilizer addition, disease, grazing intensity and climate. The model contains a simplistic plant growth model which is essentially a means of producing either leguminous or non-leguminous dry matter. This calculation is based on the relationship between growing season rainfall (including an estimate of stored water) and productivity after adjustments are made for water use efficiency in the system (French and Schultz, 1984). A linear regression is specified for each crop or pasture for the potential yield in a certain environment and the yield is then adjusted for crop-soil water use efficiency (a function of run-off and evaporation). As a strong relationship exists between C accumulation, aggregate stability and infiltration (Tisdall and Oades, 1982), the water use efficiency in the model will also change in response to fluctuations in annual C stores. Nitrogen fertilizer use efficiency in SOCRATES is assumed to be 50%. The individual crops considered (but not restricted to) in the model are canola, barley, wheat, oats and grain legume, with the model also capable of estimating pasture productivity. In our case we consider it to be a legume dominant pasture but the user will shortly be able to select the proportion of legume and grass in his pastures for a more accurate model run.

The soil model is based on four major components. All plant material can be divided into decomposable and resistant material using concepts initially described by Jenkinson (1990). The decomposable material is readily degraded by microbes and is related to the more succulent parts of the plant. It mainly consists of sugars and carbohydrate. The resistant material is associated with the plants structure itself and consists of cellulose and lignin. These are usually the woody parts (or similar) of a plant. The soil components consists of microbial biomass and humus. The microbial fraction is further subdivided into a transient unprotected fraction, which is involved in the initial stages of crop residue decomposition crop residues and a protected microbial fraction which is actively involved in the decomposition of native humus and microbial metabolites (Ladd *et al.*, 1995). Unless specified by the user, 2% of the initial organic C store is protected microbial biomass, with the remaining 98% being stable humus.

The generic description of decomposition in the model will produce microbial material, humus and carbon dioxide in proportions which are dependent on soil texture, or more specifically the cation exchange capacity of a soil. These proportions and the specific decay rates for each component of the model were initially calibrated using the $^{14}C$ data of Ladd *et al.* (1995). The first order decay rates currently used in the model are 0.84 $w^{-1}$ for decomposable plant material (i.e 84% of the material will degrade in one week at 25 degrees C at optimum moisture conditions), 0.06, 0.95, 0.055 and 0.0009 $w^{-1}$ for resistant plant material, unprotected and protected microbial biomass and stable organic matter respectively. The decay rate for the resistant plant fraction in SOCRATES is significantly faster than those specified in the CENTURY (Parton *et al.*, 1987) and Rothamsted (Jenkinson, 1990) C models because by definition we consider this material to be recognizable light fraction which is capable of being removed prior to a soil carbon analysis being performed.

The effect of temperature on decomposition is based on a $Q_{10}$ relationship of 2.0. With respect to soil moisture we have simplified the soil water calculations considering the model is based on seasonal cycles. We then field calibrated the model using and yield and soil carbon data from the Permanent Rotation Trial at the Waite Agricultural Research Institute (summarized in Grace *et al*, 1995) and have found that in a wide range of semi-arid systems, decay rates should be set at 26% of the optimal rate when growing cereal crops and 90% of the optimal rate for fallow situations (where water is plentiful). For pastures, because of the additional root production which may extract more water the value used to modify the rate (16%) is lower than the value used in the cereals.

## Model simulations

The Mallee region of north-western Victoria and eastern South Australia offers an dryland agro-ecosystem which may be considered a worst case scenario for developing management strategies to store soil organic C. Average annual precipitation at the Mallee Research Station at Walpeup is 350 mm (110-546 mm), with a mean annual temperature of 16.5 degrees C. The climate is typically mediterranean with the organic C content of sandy-loams found in this region rarely exceeding 1.0%. Their clay content is approximately 13% with a cation exchange capacity of 20 mmol $kg^{-1}$. Appreciable amounts of carbonate can be found below 30cm with widespread occurrence of calcrete outcrops. Cereal-pasture rotations are commonplace in this environment, with the shift away from the use of long (15 month) fallows. The legume-based pastures (*Medicago parragio*) and crop residues are usually grazed but stocking densities in excess of 3 sheep $ha^{-1}$ are rare. Typically 40-70% of aboveground biomass is removed through grazing.

Potential grain yield for crops and and dry matter production for pastures was estimated using data from French and Schultz (1982) and personal communications with the authors (Table 1). Water use efficiency for grain and pasture production (Figure 5) is a function of April-October rainfall plus any stored water in the profile. Phosphorus applications in excess of 20 kg $ha^{-1}$ $yr^{-1}$ are virtually essential with single nitrogen addition in excess of 50 kg $ha^{-1}$ considered rare in this environment as precipitation events are usually spasmodic and intense thus providing an ideal situation for leaching losses on the sandy-loams. The retention of crop residues in this agro-ecosystem has become more widely accepted in the past twenty years however disease carryover and weed problems are usually compounded in this situation unless a conventional tillage practice, burning or herbicides are applied.

Table 1. Regression parameters for estimating potential yield[1] in southern Australia agro-ecosystems using the SOCRATES model.

| Plant | Intercept (mm) | Slope (kg $mm^{-1}$) |
|---|---|---|
| Wheat | 110 | 20 |
| Barley | 90 | 20 |
| Peas | 130 | 15 |
| Pasture | 70 | 45 |

[1]Yield = Intercept + Slope * (Growing season precipitation + stored water (mm))

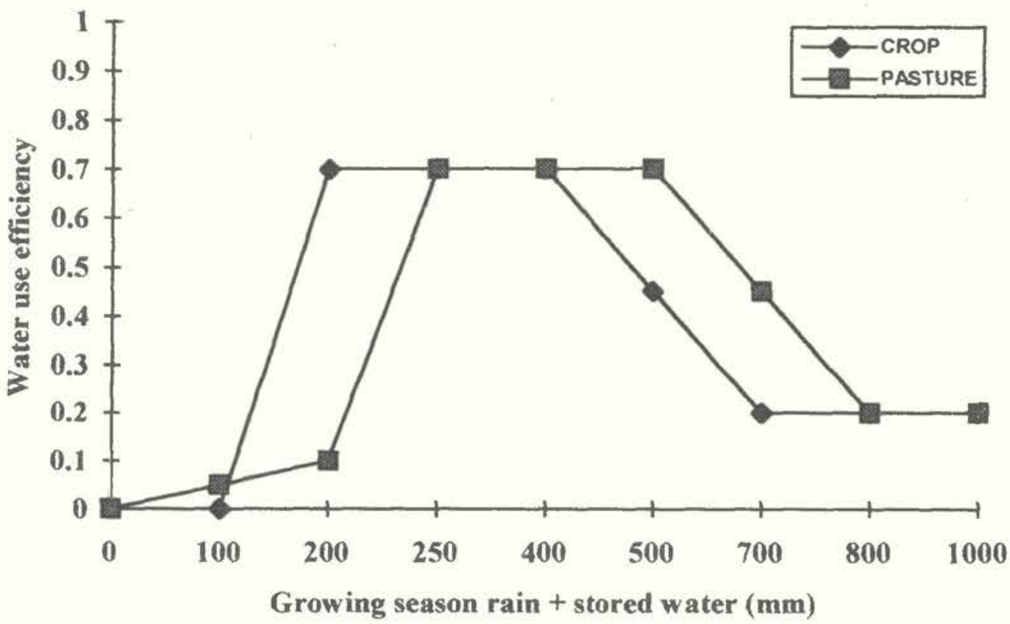


Figure 5. Water use efficiency relationships affecting yield estimates for crops and pastures in the SOCRATES model.

It is clear that with crop residue removal whether by burning or heavy grazing, soil organic carbon C in the surface 10 cm will be depleted by 20 %, or 132 kg C $ha^{-1}$ $yr^{-1}$ over 20 years of a wheat-pasture-fallow rotation (Figure 6). Even with additional N inputs through the use of grain legumes and fertilizers the best result we can achieve with residue removal is to remain at

It is clear that with crop residue removal whether by burning or heavy grazing, soil organic carbon C in the surface 10 cm will be depleted by 20 %, or 132 kg C $ha^{-1}$ $yr^{-1}$ over 20 years of a wheat-pasture-fallow rotation (Figure 6). Even with additional N inputs through the use of grain legumes and fertilizers the best result we can achieve with residue removal is to remain at our initial level of organic C after 20 years. With residue retention we are still unable to sequester enough C to exceed our initial values in all cases except where fertilizer N is applied. Residue retention increases C storage (compared to removal) by an additional 24-60 kg C $ha^{-1}$ $yr^{-1}$ for the same rotations (Figure 7) and an additional 150 kg C $ha^{-1}$ $yr^{-1}$ if we apply 100 kg N $ha^{-1}$, however this latter scenario would have to be considered impractical in this environment without specialized equipment to apply the fertilizer in a series of split applications. Using average wheat and pea yields of 2.0 and 1.4 t $ha^{-1}$ respectively for the strategy where our N input is 50 kg $ha^{-1}$, our net profit using Australian market prices as of September 1995, before taking into account capital machinery or living costs would be US$69 $ha^{-1}$ $yr^{-1}$. Using a simplistic assumption that the cost of living is US$30,000 $yr^{-1}$ and the purchase of specialized crop residue handling equipment during the course of the 20 year rotation will cost US$6,000 $yr^{-1}$, a landholder with a 500 ha farm entirely devoted to this operation will lose US$3 $ha^{-1}$ $yr^{-1}$ in sequestering 1.2 t C $ha^{-1}$ over the course of 20 years, a total cost of US$50 t C $ha^{-1}$. Considering this simulation was undertaken using a coarse textured soil in a marginal environment, this cost of US$50 t C $ha^{-1}$ should be considered the upper limit when using the relatively high input four phase rotation management strategy outlined in the simulation.

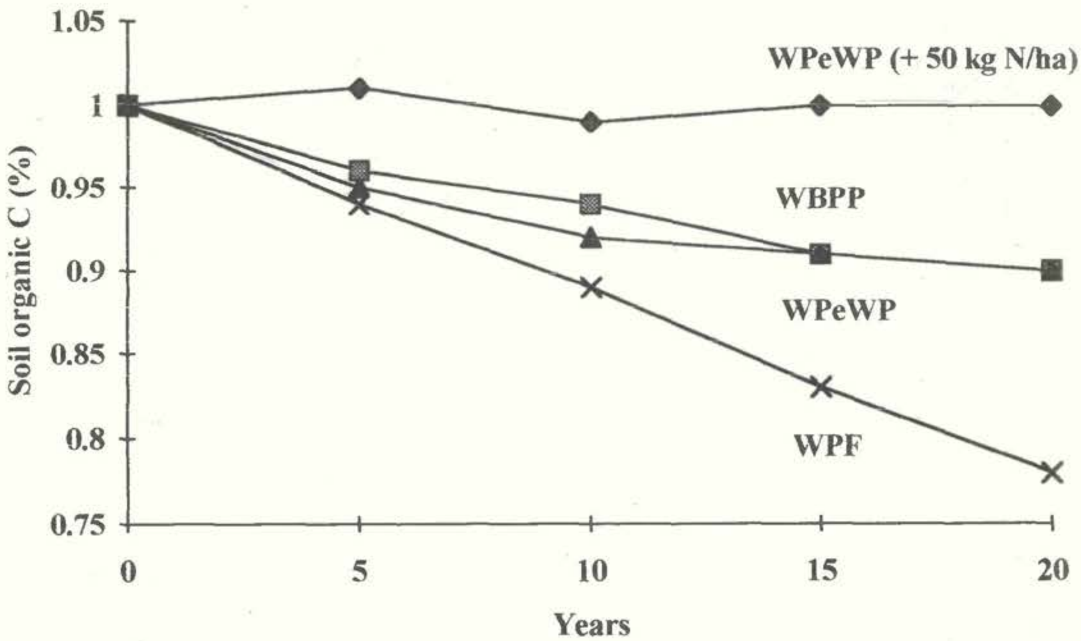


Figure 6. Changes in soil organic C (0-10 cm) in response to residue removal, crop rotation and nitrogen application as simulated by the SOCRATES model for the Mallee region of north western Victoria.

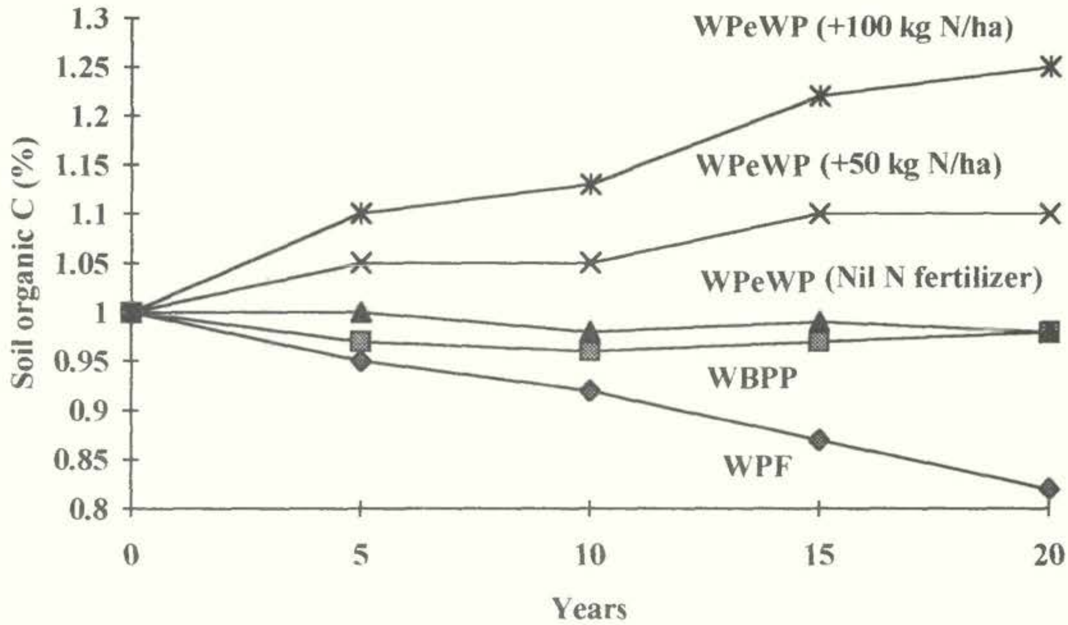


Figure 7. Changes in soil organic C (0-10 cm) in response to residue retention, crop rotation and nitrogen application as simulated by the SOCRATES model for the Mallee region of north western Victoria.

## Future developments

Realistic management strategies to increase or at best reduce the loss of soil organic C in dryland agro-ecosystems must be done with state-of-the-art models that accurately represent our current understanding of soil organic matter decomposition and be complemented by data that enables us to extrapolate our results accurately. The current description of SOCRATES segregates native soil organic matter into microbial biomass and humus. Whilst we support the concept that soil organic matter is made up of a number of distinct 'pools' which are delineated by their ability to be biologically degraded over a particular period of time, we have chosen these two fractions purely on the basis that at least one, the microbial biomass, is biologically recognizable (as opposed to conceptual) and can be measured through routine analytical techniques such as the chloroform fumigation-incubation technique (Jenkinson and Powlson, 1976). In recognizing that the use of a single humus pool does pose problems, particularly with simulations in excess of 100 years, we are also aware of the important role that physical protection plays in the turnover of organic matter (Ladd *et al.*, 1995) and the need to quantify substrates associated with this fraction. It is within this context of providing accurate analytical data for simulation models that we are currently modifying SOCRATES to accommodate an extremely stable physically protected fraction which has been identified through a novel technique employing high energy ultraviolet photo-oxidation (Skjemstad *et al.*, 1993). The <50 um fraction separated from whole soils is exposed to a UV source for varying periods of time, with four hours considered sufficient to give an estimate of the physically protected pool. Infra-red spectroscopy has identified the materials external to both clay- and silt-sized aggregates as proteinaceous in nature and the materials within the aggregates as humic acids. These humic acids appear to be physically protected from photo-oxidation rather than chemically resistant.

This high energy UV photo-oxidation procedure shows a consistent amount of C is physically protected at any one of the long-term field sites that have been assayed (Table 2), although variation between sites and treatments may be large (0.09-0.61%). The three oldest cropping trials sampled (Dooen, Glen Innes and Urrbrae) range from 70 to 83 years in duration and are found on markedly different soil types (grey clay, clay-loam and sandy-loam respectively). However, the physically protected organic matter fraction estimated by UV photo-oxidation has been found to be a consistent proportion of the total soil organic C (31%) for a wide variety of samples taken over the course of each trial. For forty-nine samples assayed across 8 trials at 7 different locations, this proportion was approximately 26%. Solid state $^{13}C$ NMR studies have shown that most of this protected C is in the form of charcoal (inert C) which appears to be a relic of fires prior to European settlement. In particular, >80% of the C in the Glen Innes and Longeronong protected pools is in the form of charcoal (Skjemstad, unpublished). Infra-red spectroscopy is also being used to qualitatively identify the presence of charcoal and reduce the number of expensive NMR analyses in providing a more exact description of soil organic matter in drylands.

Preliminary modelling studies using the Waite Permanent Rotation trial dataset indicate the protected pool measured by UV photo-oxidation may overestimate the size of the inert pool as required by the Rothamsted C model, but is a better measure of the passive pool of the CENTURY model. The charcoal value however appears to give a good estimate of the inert pool.

Table 2. The contribution of physically protected organic carbon as measured by UV photo-oxidation to total soil organic carbon at selected long-term trials in Australia.

| Trial | Samples Assayed (No.) | Organic carbon (%) | 'Protected' carbon (%) | 'Protected'/ Organic carbon |
|---|---|---|---|---|
| Urrbrae | 12 | 1.91 +/- 0.50 | 0.66 +/- 0.12 | 0.32 |
| Glen Innes | 9 | 2.05 +/- 0.39 | 0.60 +/- 0.05 | 0.29 |
| Dooen | 6 | 1.02 +/- 0.06 | 0.35 +/- 0.04 | 0.34 |
| Walpeup (MM1) | 4 | 0.43 +/- 0.07 | 0.09 +/- 0.02 | 0.21 |
| Rutherglen (ley) | 3 | 2.29 +/- 0.44 | 0.41 +/- 0.03 | 0.18 |
| Rutherglen (RR1) | 6 | 1.34 +/- 0.34 | 0.39 +/- 0.03 | 0.29 |
| Tarlee (rotation) | 4 | 1.42 +/- 0.24 | 0.30 +/- 0.03 | 0.21 |
| Wagga (SATWAGL) | 5 | 1.92 +/- 0.30 | 0.53 +/- 0.05 | 0.28 |

SOCRATES as described is a process model dealing with information from one point in space over a specified time period - historically or predicted. What happens when we require its application at a local, regional, national or world-wide scale? To develop SOCRATES and others of similar nature as powerful tools for planning and management purposes of areas rather than individual points, requires the integration of appropriately geographically referenced data sets with the model. Such data sets in this instance are rainfall, temperature, soil type, land zone and chemical analysis data (specifically cation exchange capacity and initial soil organic C values). However, we run into the problem that in most instances these data sets exist as point samples and the methodologies employed to convert them into a "spatial data set" are extremely important in analysing any predictions made by the models using them. Spatial variability within the data sets is a critical issue and in our on-going work to convert SOCRATES from a process model into one that has a strategic spatial approach, we are employing a variety of techniques to create accurate spatial data sets from the original point data.

The methodology we are employing involves the use of land systems and soils class data from the South Australian Department of Primary Industry. A 'land system' is an area of land with a particular set of features that are distinquishable from the surrounding landscape. These features are geology, soils, topography, climate, and vegetation, with the particular types of potential land degradation or erosion, and limitations to land capability closely related to individual land systems. Land capability refers to the capacity of the land to support a particular land use and sustain that use in the long term and categorized into one of eight classes according to its suitability for agricultural production. The capability of the land, and therefore the particular class to which it is placed, depends on the nature of its physical characteristics or limitations. Land capability is ultimately determined by a quality factor, which is usually represented as one or more limiting factor. These include (but are not restricted to), salinity, acidification (pH), water repellence, flooding potential and water erosion potential. Land systems data is input into a GIS as point data and converted to polygon coverages and then spatial or raster based surfaces (to allow cell-based modelling). The POLYGRID function in ARC INFO allows the operator to interactively choose the size of the grid cells and the priority of the features within the cells but does not employ an interpolation method - individual data points occupy individual cells.

For the climate and soil chemical analysis data which essentially drive the model, we have found that the most appropriate methods of creating a spatial surface for these data were the interpolation procedures of Laplacian spline (Hutchinson, 1991, Bryceson 1993) and kriging (Brooker, 1979). As indicated by Dubrule (1984) there is a theoretical link between these techniques and which one is used is dependent on the application and data. In our case we are specifically using kriging and once created the data surfaces are registered to a Lambert Conformal projection and cell-based modelling within the GRID module of ARCINFO is carried out. Land capability attributes can then be overlaid on this projection and management strategies can be developed which are entirely site specific.

## Conclusions

Increasing soil organic C stocks in cultivated drylands is necessary to improve primary production, reduce erosion and if feasible be utilized as a sink for offsetting the production of carbon dioxide from fossil fuels and similar sources. Model simulations suggest that even in high input, conservation tillage farming systems, it is an onerous task to sequester significant amounts of organic C. Accurate estimates of C storage can only be made with detailed spatial data and only then can site specific management strategies be developed..

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# Impact of carbon storage through restoration of drylands on the global carbon cycle

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## Synopsis

The paper describes the use a global carbon cycle model, GLOCO version 2, to analyze the potential for storage of carbon in drylands. A brief description of the model is followed by a response analysis to different management options which increase the storage of carbon in drylands. The extent of degradation of the drylands is estimated, to formulate a future scenario for management actions. These actions are then studied


## Key Points

1. *The increase in atmospheric carbon dioxide buildup is well recorded both* by direct measurements, most notably the Mauna Loa record, and indirect *measurements dating back many decades.* Whether this rise in atmospheric carbon dioxide will result in global climate change will be debated still for many years, as well as the positive or negative effects that such changes may have on human activities and terrestrial vegetation. Nevertheless, it is important to identify options to reduce the rate of atmospheric carbon increase. No single option will probably be sufficient by itself to significantly reduce the rate of increase.
2. Intensive use of woodlands, grasslands and arid areas, has resulted in significant degradation of vegetation cover and soil quality, leading in many cases to desertification. Associated with these activities is a net carbon loss from the terrestrial biomes and soils to the atmosphere. Concurrently, the burning of fossil fuels, and the conversion of natural biomes to pastures and agricultural uses results in a large flux of carbon to the atmosphere, which only after a long time partially redistributes to the oceans and the remaining natural biomes.
3. Several "management" options have been proposed to reduce the buildup of carbon dioxide in the atmosphere and reduce the likelihood of changes in global climate, including reduction of fossil fuel burning and degradative land use changes, storage of carbon in the oceans, etc. Another option is to look at the potential for terrestrial carbon storage by restoring degraded drylands and desert fringes to their original carbon content, or in some cases even increase the carbon storage capacity if it is economically feasible. This alternative has the advantage of sharing the cost of carbon storage with the land's restoration, providing multiple benefits.
4. The GLOCO model, a global carbon cycle model with eight terrestrial biomes (woodlands, grasslands, deserts, tundra, temperate, tropical and boreal forests, and agricultural lands) is a useful tool to analyse change. It allows detailed analysis of carbon and nitrogen fluxes. The GLOCO model can also be used to study land use change and fossil fuel emission scenarios. The model is used to study the transient response of actual vegetation, which is more realistic than looking at equilibrium conditions of potential vegetation.
5. Using estimates of the potential land area suitable for restoration in woodlands, grasslands, deserts and tundra, as well as estimates of the rate at which restoration can proceed, it is estimated that carbon storage in these biomes can range up to 0.8 billion tons of carbon per year (Gt C/yr), with a corresponding reduction in atmospheric buildup of 0.5 Gt C/yr, which

represents up to 15% of the average annual atmospheric carbon buildup in the next century, 3.5 Gt C/yr, assuming the IPCC 92d scenario. Clearly, a global strategy for reducing atmospheric carbon dioxide concentration will require the implementation of multiple options.

6. The advantage of carbon storage in restored drylands is that it comes as a side benefit to programs that are justifiable in terms of land management as well. If dryland restoration is justified by a direct environmental benefit then the benefit of reduced atmospheric carbon comes at no cost. Another perspective is to state that the direct environmental benefit of restoration and the indirect benefit of atmospheric carbon reduction can together be used to justify the costs.

7. With over 3,500 Mha of drylands degraded, 27% of the terrestrial biosphere, the potential for storing additional carbon as biomass in these regions is significant. Actions taken to increase the net primary productivity of these biomes may result in storage of 0.2 Gt C/yr. Increasing the SOM production in the agricultural drylands may store an additional 0.2 Gt C/yr. And restoring marginal agricultural land to its original natural state can result in the storage of an additional 0.2 to 0.45 Gt C/yr in these biomes, for a total terrestrial biome storage of up to 0.8 Gt C/yr. This corresponds to a net decrease in atmospheric carbon buildup of 0.5 Gt C/yr.

Key Words: Modeling, carbon sequestration, $CO_2$, biomes, soil organic matter

## Introduction

The increase in atmospheric carbon dioxide buildup is well recorded both by direct measurements, most notably the Mauna Loa record, and indirect measurements dating back many decades, e.g. the Siple ice core data (Keeling *et al.*, 1989). Whether this rise in atmospheric carbon dioxide will result in global climate change will be debated still for 2.54many years, as well as the positive or negative effects that such changes may have on human activities and terrestrial vegetation. Nevertheless, it is important to identify options to reduce the rate of atmospheric carbon increase. No single option will probably be sufficient by itself to significantly reduce the rate of increase.

There are some actions that may be taken at no cost or relatively low cost that will reduce the average per capita use of energy, for example efficiency gains in electricity, oil and gas use in residential, commercial and industrial sectors (Rubin *et al.*, 1992). Other actions under study involve the long term storage of fossil fuel emissions in the deep ocean (Keller and Goldstein, 1995; Kheshgi *et al.*, 1994; Golomb *et al.*, 1992; Marchetti, 1977). It is also likely that the natural biomes (forests, grasslands, woodlands, tundra and deserts) are already participating in increased storage of a fraction of the carbon from fossil fuel emissions, although physical evidence for this is scant (Kauppi, 1992; Kenk and Fischer, 1988). Actions which can be taken to accelerate storage of carbon in the terrestrial biosphere may be relatively low cost and thus may partially offset some of the emissions from fossil fuel burning.

Concurrent with actions to reduce carbon buildup in the atmosphere are the activities undertaken by the United Nations Environmental Programme (UNEP) and individual nations to combat desertification and to restore the ecological quality of their drylands (UNEP, 1992). Indirectly, these activities may result in storage of a fraction of fossil fuel emissions and thus share the overall cost of both restoring drylands and reducing atmospheric carbon. Joint implementation programs may be designed between countries emitting large amounts of carbon to the atmosphere and those needing funds to combat desertification, to partially offset the emissions. Intraregional offsetting of fossil fuel emissions is possible by combining projects between local power generating utilities and land management agencies.

This work's objective is to use a global carbon cycle model, GLOCO version 2, to analyze the potential for storage of carbon in drylands. First, we provide a brief description of the model, followed by a response analysis to different management options which increase the storage of carbon in drylands. Then we estimate the extent of degradation of the drylands, to formulate a future scenario for management actions. These actions are then studied with the GLOCO model individually and in combination, to determine a realistic range for carbon storage in drylands.

## GLOCO Model

The GLOCO model was developed by R. Hudson *et al.* (1994) for the Electric Power Research Institute (EPRI) for performing risk analysis on management options to reduce the rate of carbon buildup in the atmosphere. GLOCO, ver. 2 (Hudson et al., 1995), comprises eight terrestrial biomes with a detailed mechanistic description of the processes within each biome that involve the carbon and nitrogen cycles. The terrestrial biomes are linked through a common atmosphere with low latitude and high latitude oceans. The oceans are further described by surface and deep subcompartments. The oceanic model is based on the HILDA model of Siegenthaler and Joos (1992), which has been calibrated using the isotopic carbon tracers, $^{13}C$ and $^{14}C$. The GLOCO v.2 model was calibrated by Hudson *et al.* (1994) using historical atmospheric $CO_2$ concentrations, as well as the Geochemical Ocean Sections Study, GEOSECS (Takahashi et al., 1981 a, b) oceanic profiles of dissolved organic carbon, alkalinity and phosphate.

The model allows the user to input all the parameter values governing each biome process (e.g. net primary productivity, litterfall, carbon to nitrogen ratios in the various tissues, etc.), each ocean process (e.g. biological uptake of carbon, convection and dispersion of carbon within a layer and to other layers, etc.) and atmospheric processes (e.g. methane oxidation to carbon dioxide), as well as the relationship between temperature and atmospheric $CO_2$ at the individual biome level (e.g. as predicted by a Global Climate Model). In addition, historical and future anthropogenic activities affecting the global carbon cycle are included, such as fossil fuel emissions (as carbon dioxide and as methane); anthropogenic nitrogen emissions and deposition rates in different biomes, land use changes either from a natural biome to an agricultural biome or reverting agricultural land to the original biome (which requires time to allow the vegetation to grow back) and shifting cultivation, as well as forestry activities.

The model provides as outputs carbon dioxide and methane concentrations in the atmosphere, projected temperatures in each biome and ocean, carbon concentration and alkalinity in the oceans, carbon and nitrogen in each terrestrial biome subcompartment (e.g. wood, litter, soil organic matter, etc.) and the area in each biome, as well as an overall carbon mass balance for each biome, ocean and atmosphere.

The eight terrestrial biomes considered in GLOCO are: temperate grasslands, woodlands, desert, tundra, temperate forest, tropical forest, boreal forest and agricultural biome, which is actually broken down into seven subbiomes depending on the vegetation prior to cultivation. The Whittaker and Likens (1973) ecosystem types were aggregated by Hudson et al. (1994) on the basis of carbon pool sizes and net primary production (NPP); for this study, it is important to note that the woodland comprises tropical and temperate woodland and shrubland as well as savanna.

The terrestrial biomes are all modeled based on a generic biome. The main carbon stocks (subcompartments) in each biome are: foliage, stem, roots, litter, soil organic matter (SOM) and humus. In each biome subcompartment there is a balance between carbon and nitrogen dictated by the user specified carbon to nitrogen ratios (C:N). Both carbon and nitrogen from anthropogenic activities participate in the fertilization of the biome, and both nutrients are lost through rivers to the oceans due to weathering and leaching, albeit at different deposition, fixation and leaching rates for each nutrient, and with different rates for each biome.

The main processes in each biome are presented in Table 1, along with the main variables in each process. Most processes are temperature dependent, and some are limited by nitrogen or carbon availability, or the ratio of these nutrients. Some processes are simply first order and some involve a transition from a first to a zeroth order process (e.g. gross primary productivity). A complete mathematical description of the model is available in Hudson and Gherini (1995). Each biome is calibrated independently to achieve an equilibrium condition and then the parameters are transferred to the full GLOCO model.

## Response Analysis

There are three main options in GLOCO for looking at carbon storage due to drylands management: 1) restoring degraded pasture and agricultural land to its original biome (either woodland or grassland); 2) increasing the net primary productivity of the degraded biome; and 3) increasing the production of soil organic matter (SOM) in the portion of agricultural land that corresponds to either woodlands or grasslands. It is also possible to model an increase in nitrogen fertilization of the drylands either by increasing the natural rate of nitrogen fixation or by increasing the rate of nitrogen deposition due to indirect anthropogenic activities, but these options have not been studied here.

Converting degraded pasture and agricultural land to woodlands or grasslands results in a net average annual storage in the terrestrial biosphere of about 0.03 Gt C/yr per Mha/yr converted, with little difference whether the original biome is a woodland or a grassland (Figure 1). The GLOCO model assumes that the agricultural land is reverted to its original biome (i.e. grassland cannot be converted first to agriculture or pasture and then to woodland). However, if the agricultural land is simply abandoned, it may result in additional soil erosion and possibly desertification (Garcia-Ruiz *et al.*, 1994), suggesting that reverting agricultural land to its original biome may require at least some management.

The annualized storage is averaged over a 100 year simulation, since the initial storage is slower as the carbon and nitrogen stocks in the biomes' subcompartments adjust to the new conditions. A background fossil fuel emissions scenario of 6 Gt C per year is imposed, and only the net increase or decrease in carbon storage is evaluated, subtracting any carbon fertilization effect on the natural biome. Neither the woodlands or the grasslands are very sensitive to carbon fertilization (Keller and Goldstein, 1994), such that in the event that the actual level of fossil fuel emissions is much different in the future, the results would not be affected significantly. It should be noted that the response of the drylands to land-use changes is considerably smaller than the response of the temperate, boreal or tropical forests, which ranges from 0.09 to 0.14 Gt C/yr per Mha/yr converted, for the same fossil fuel emissions.

The actual reduction in atmospheric carbon buildup is about 60% of the storage in the biosphere, due to partitioning of carbon among oceans and other natural biomes, for an annualized decrease in the atmospheric carbon buildup of about 0.02 Gt C/yr per Mha/yr converted (Figure 2). For example, if 5 Mha of agriculture is converted every year to woodlands, with a total conversion of 500 Mha after 100 years, the carbon stored in the terrestrial biome is 15 Gt C, and the decrease in carbon buildup in the atmosphere is about 10 Gt C.

Another action that may be undertaken is to increase the average Net Primary Productivity (NPP) of the drylands, namely the woodland and grassland biomes. This may be achieved by tree planting (Ahlback, 1994; Nair, 1984), the use of halophytes (Douglas, 1993), or actions to raise the productivity of arid lands (Skoupy, 1993), by restoring the quality of the soils and/or the vegetation cover in drylands which have been overgrazed or farmed intensively, or where erosion and salinization processes have impoverished the soil (Dregne, 1990; Szabolcs, 1992, 1989; Crosson and Stout, 1983). Small, medium and large scale projects have been undertaken in South America (Mendoza, 1993), North America (Hurt, 1986), the Middle East (Omar and Abdal, 1994), Africa (Darkoh, 1989), India (Sinha, 1993) and China (Zhenda and Tao, 1993), to name just a few examples. Figure 3 presents the response analysis of four biomes to an increase in average NPP. The largest response is obtained from the woodlands, where even a 10% increase in average NPP results in an annualized change in carbon storage in the woodlands biome of 0.2 Gt C/yr. These simulations were also conducted for 100 years and with a background fossil fuel emissions rate of 6 Gt C/yr, with only the net increase shown here. In this case, we have a net storage of about 20 Gt C in 100 years.

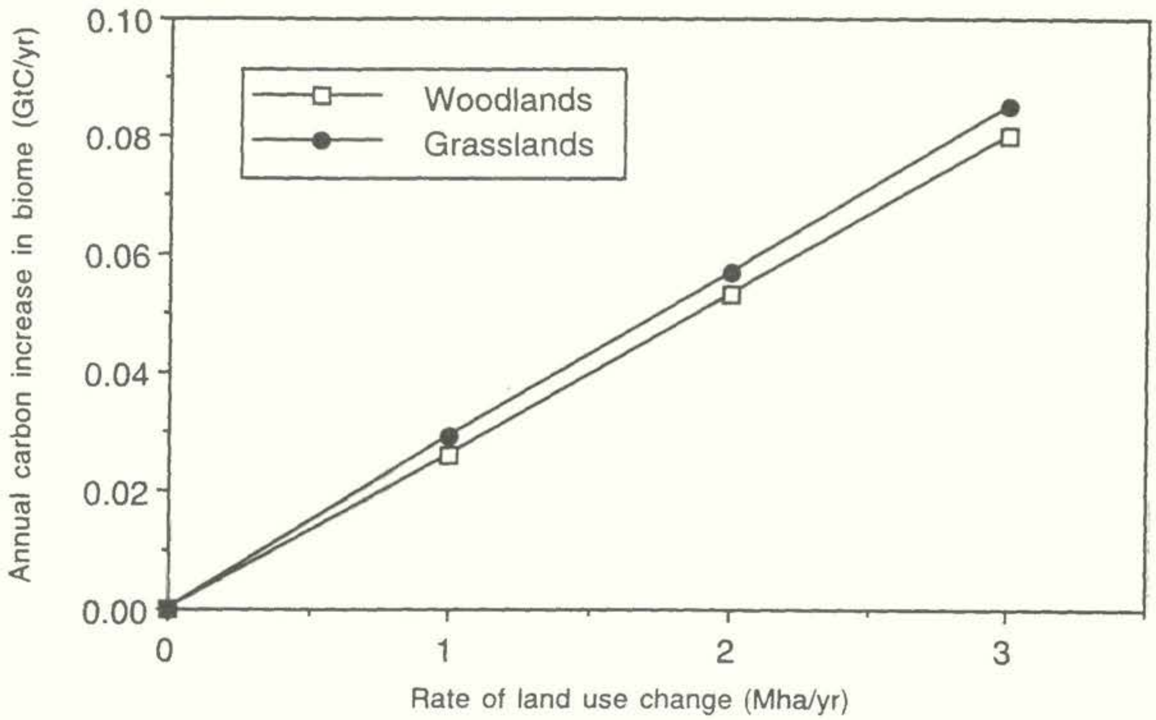


Figure 1. Average annual carbon storage in the terrestrial biome upon restoration of degraded pasture and agricultural land to the original dryland biomes, as a function of the rate of land use change. A background fossil fuel emission scenario of 6 Gt C/yr was used during the 100 yr simulation.

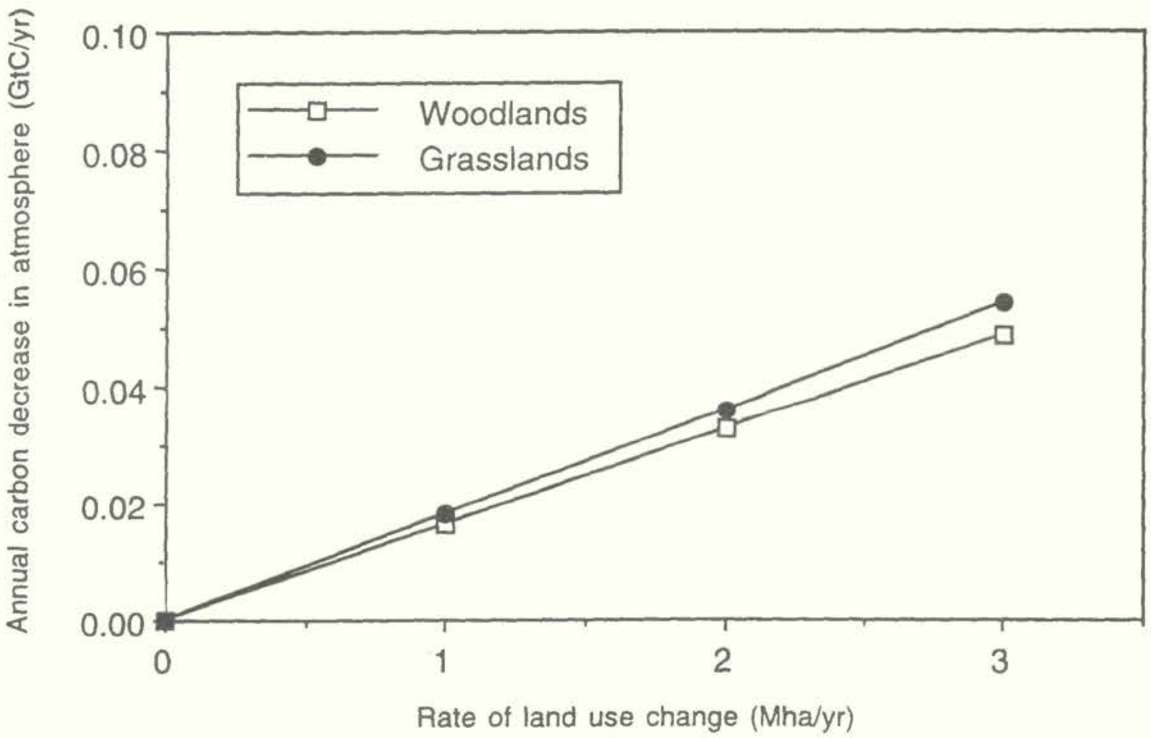


Figure 2. Average annual decrease in the carbon stored in the atmosphere upon restoration of pastures and agricultural land to the original dryland biomes.

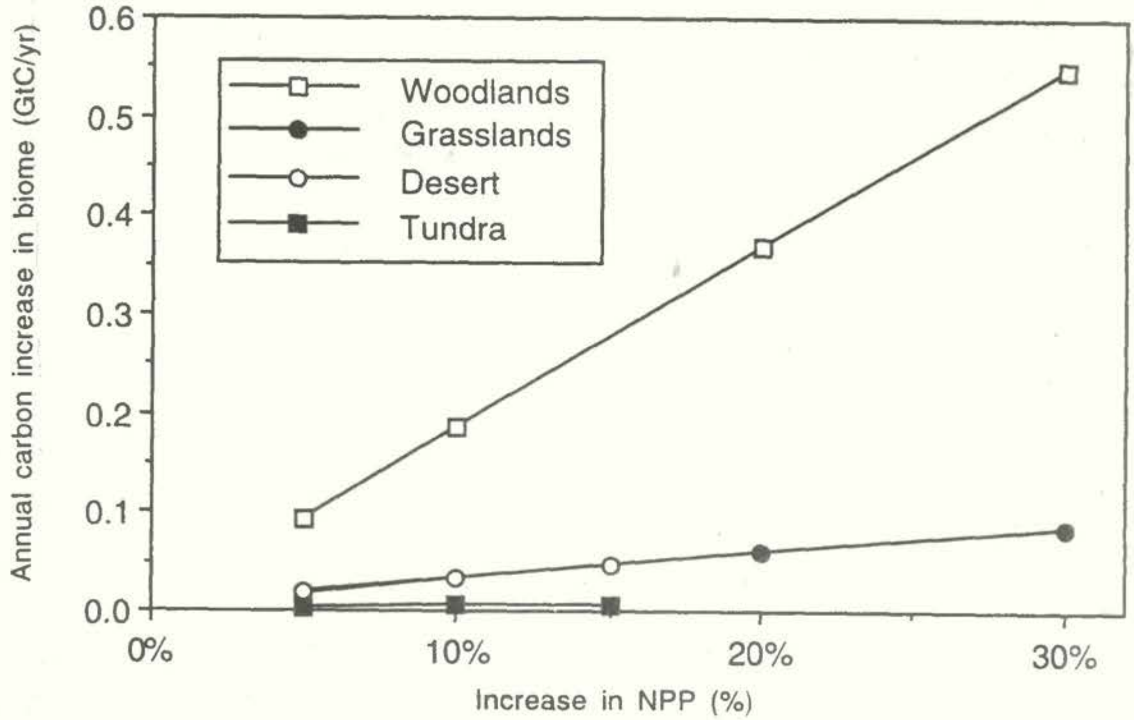


Figure 3. Annualized change in carbon stored in the terrestrial biomes as the average Net Primary Productivity (NPP) of the dryland biomes is increased, compared to the NPP used in the historical calibration.

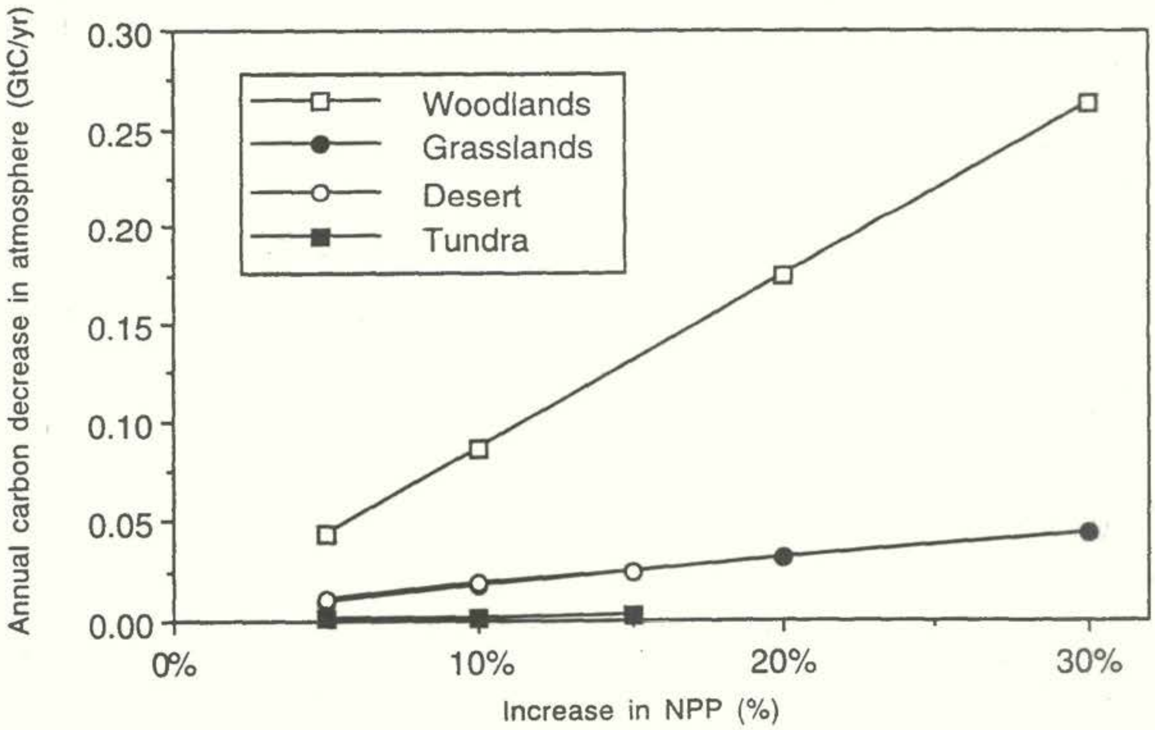


Figure 4. Decrease in carbon buildup in the atmosphere due to the increase in average NPP of the dryland biomes.

The corresponding average decrease in atmospheric carbon buildup is about 0.09 Gt C/yr for a 10% increase in NPP in the woodlands biome, with no land use changes in any of the biomes (Figure 4). The response of the grasslands is almost identical to desert, with only about 0.04 Gt C stored per year in each biome for an average 10% increase in NPP. Deserts in the GLOCO model correspond to hyper-arid lands, with an aridity index (actual precipitation over potential evapotranspiration ratio) of less than 0.05. The tundra biome has a negligible response to an increase in NPP.

It is interesting to note the differences in biome subcompartment carbon storage per unit area due to an increase in NPP. The grasslands store most of the additional carbon as litter (Figure 5) or SOM, whereas the woodlands store most of the additional carbon as wood (Figure 6) and some increase in litter and SOM. Note the large difference in scales.

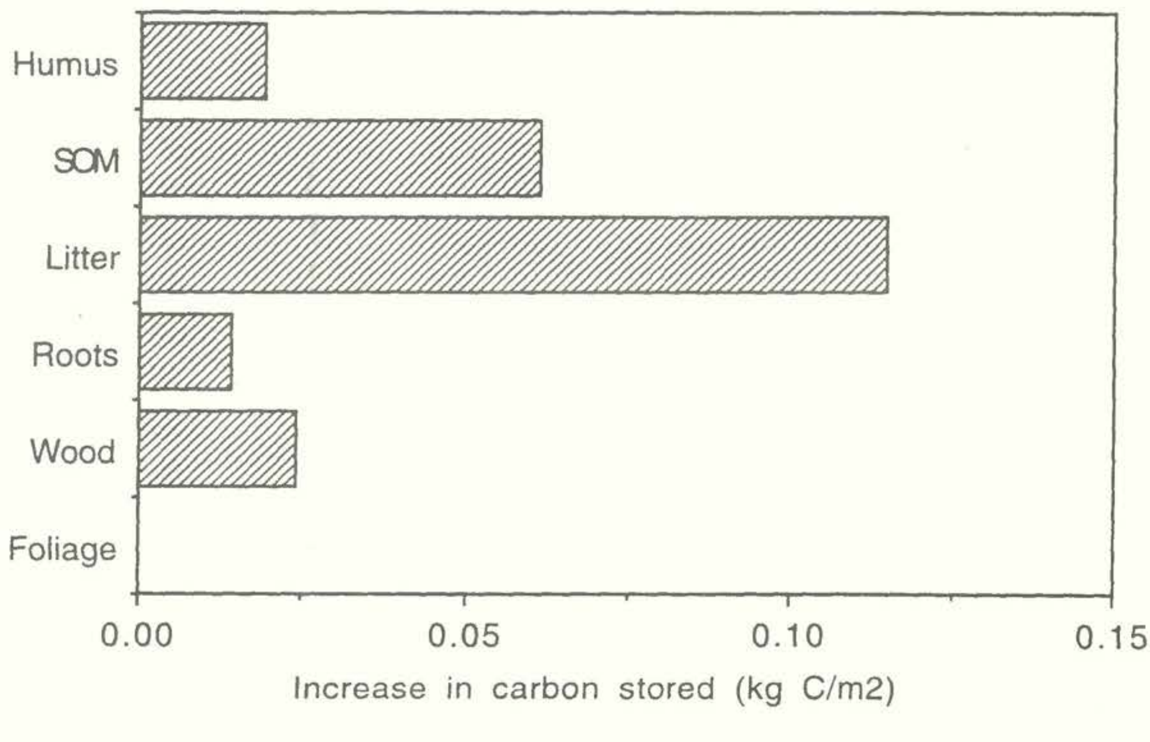


Figure 5. Partitioning of carbon among compartments of the grasslands biome for a 10% increase in grasslands NPP, during a 100 year simulation.

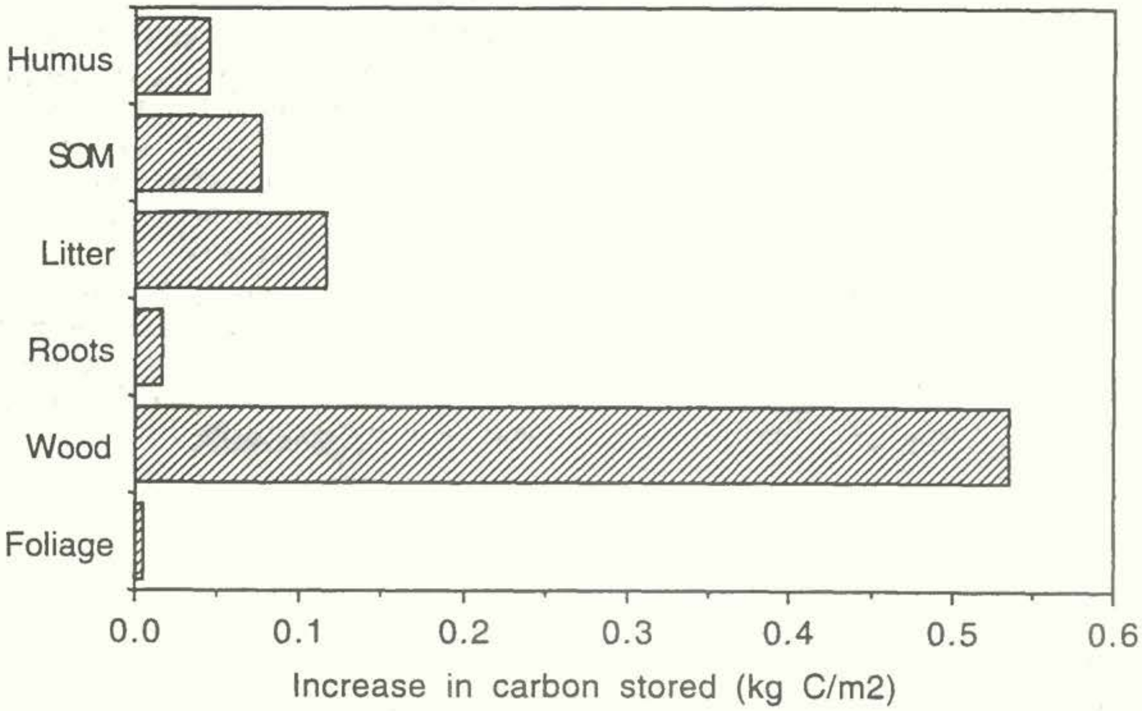


Figure 6. Increase in carbon storage in the various compartments of the woodlands biome for a 10% increase in woodlands NPP.

The third analysis involves the increase of SOM production in the fraction of the woodland or grassland biomes which has been converted to agriculture. This may be achieved for example by introducing deep-rooted grasses (Fisher *et al.*, 1994) in pastures, the appropriate choice of crop combination and crop rotation or an analysis of all the factors affecting SOM (Parton *et al.*, 1987). In Figure 7 it can be seen that even for a modest 10% increase in SOM production in the agricultural portion of either of these biomes, the net increase in carbon stored in the terrestrial biosphere increases by an annual average of 0.04 to 0.06 Gt C/yr. An increase of 20% in agricultural SOM productivity would result in an average storage of 0.08 to 0.1 Gt C/yr. The fraction of the grasslands that has been converted to agriculture or pasture is more sensitive than the woodlands agricultural/pasture fraction to an increase in SOM production, both due to its carbon and nitrogen stocks and its temperature sensitivity.

The corresponding reduction in atmospheric carbon is 0.02 to 0.03 Gt C/yr for a 10% increase in SOM production (Figure 8). All of these simulations have also been conducted during a 100 year period to allow for the carbon and nitrogen stocks to equilibrate in the biomes' subcompartments, with a background carbon fertilization due to a 6 Gt C/yr fossil fuel emissions rate, and with the effects of the carbon fertilization subtracted from the total response.

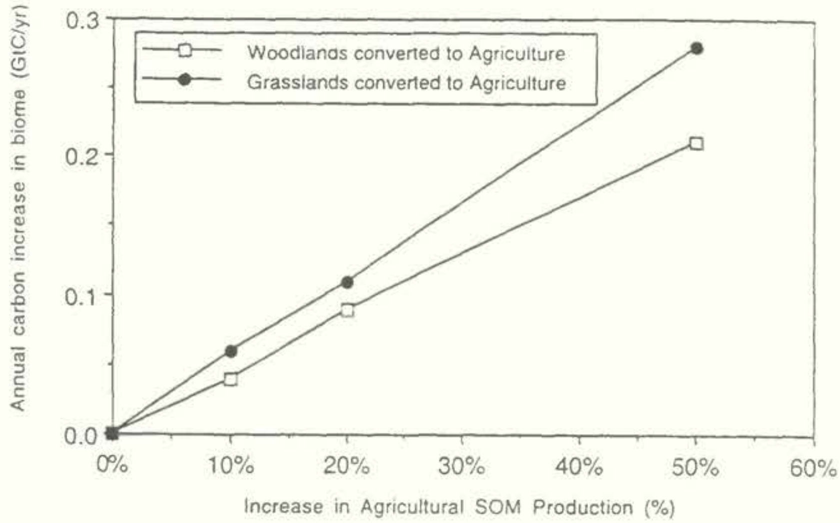


Figure 7. Increase in annualized carbon storage in the fraction of agriculture land originating from woodlands or grasslands, as the agricultural Soil Organic Matter (SOM) production is increased.

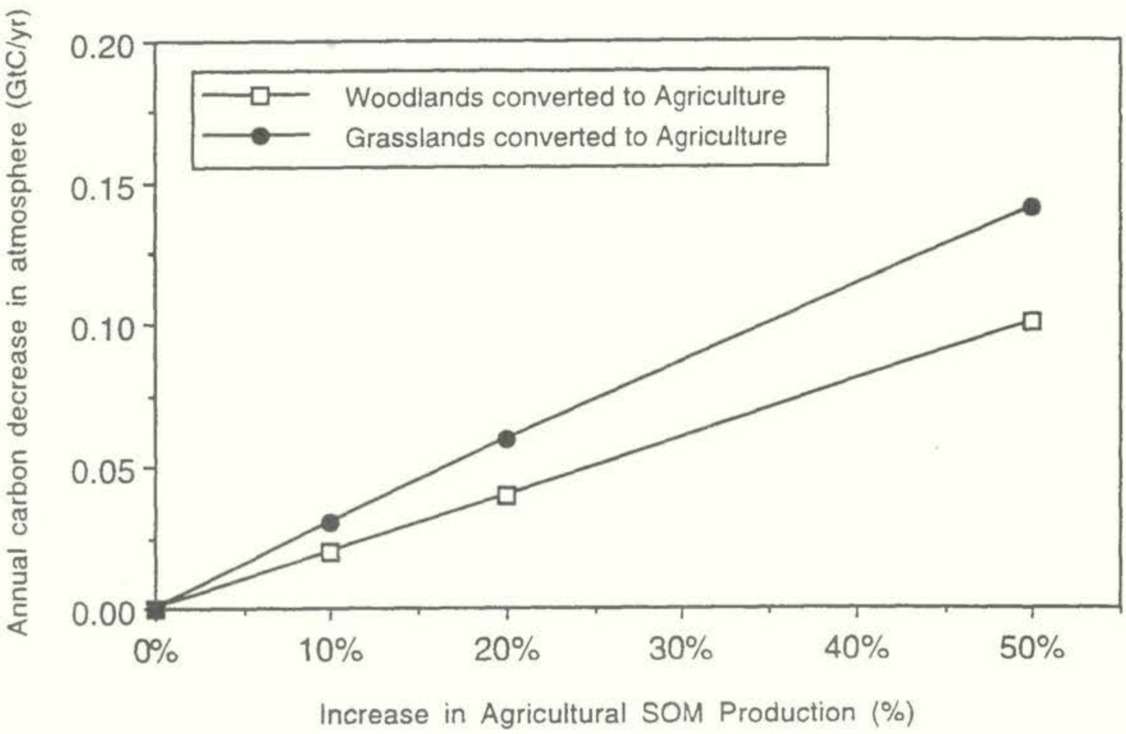


Figure 8. Average annual decrease in carbon stored in the atmosphere due to an increase in agricultural SOM production.

Historical Land Use Changes and Carbon Loss

Figure 9 presents the historical change in land use as compiled by Hudson *et al.* (1994) from several sources (Houghton and Hackler, 1994; Houghton, 1991 a, b; Houghton *et al.*, 1991 c, d; Houghton *et al.*, 1983). The total terrestrial biome area, 12,800 Mha, is conserved from 1700 to 1990. The rapid increase in the agricultural biome is the result of the combined conversion from the other biomes, especially the temperate forest in the 1700 and 1800's and the woodlands, grasslands and tropical forests in recent decades. In the temperate forests from 1950 on, agricultural land has been abandoned or actively converted to forest again, resulting in a slight increase in the area of this biome.

The net change from 1700 to 1990 in terrestrial carbon storage, using the GLOCO model and considering the reconstruction of the fossil fuel emissions from Keeling (1991) and Marland and Boden (1991), is shown in Figure 10. The terrestrial biosphere has lost almost 100 Gt C in these 290 years, which is the net from gains in agriculture and losses in most of the natural biomes, notably the tropical and temperate forests, as well as the grasslands and woodlands. The actual carbon loss is smaller than simply due to land use change, since there is a significant carbon and nitrogen fertilization effect in the temperate and boreal forests. This calculation does not include land degradation after the land use change.

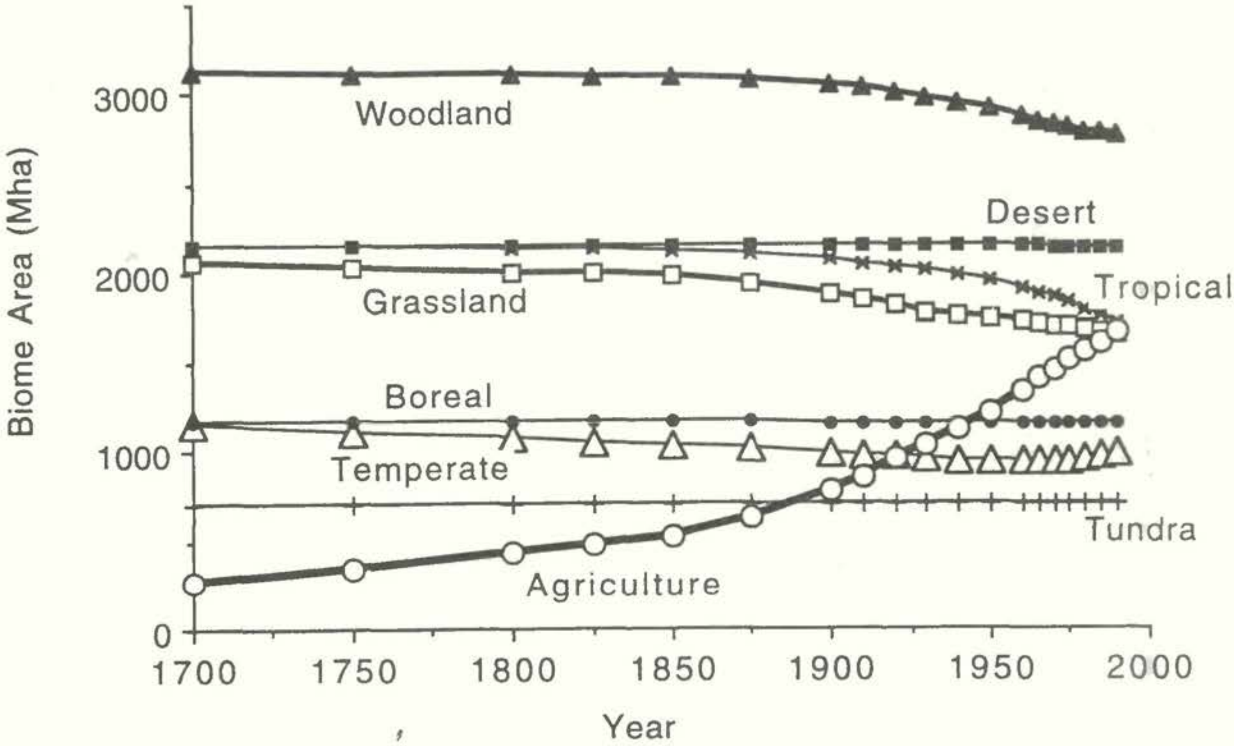


Figure 9. Historical land use changes from the preindustrial natural biomes to agriculture and as agricultural land is abandoned in some biomes. Shifting cultivation is also considered.

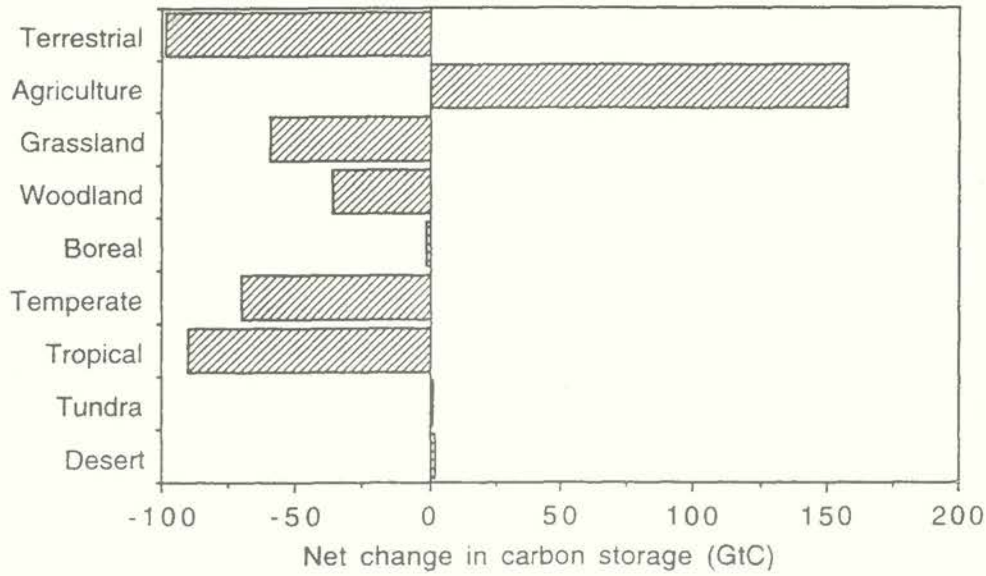


Figure 10. Change in carbon storage in the terrestrial biomes from 1700 to 1990 using GLOCO and considering land use changes and fossil fuel emissions. The net change in terrestrial biome carbon is shown in the first bar, and the breakdown among biomes is given in the lower bars.

With respect to the drylands degradation, it is estimated (Dregne *et al.*, 1991) that of the approximately 5,160 Mha of drylands (which excludes the hyperarid deserts), 69% has been degraded, mainly by the loss of vegetation cover, but in a significant fraction also accompanied by soil degradation, mainly erosion. The breakdown is shown on Figure 11, where it is clear that most of these lands (88%) are used as rangelands, with a high level of degradation, around 73% of all rangelands, mostly due to overgrazing. Irrigated and rainfed croplands have fared slightly better, with 30% and 47% degradation of their corresponding areas, respectively. Most of the degradation in this case is due to salinization, alkalinization, waterlogging and impoverishment due to lack of crop rotation. Some of these lands are also been lost due to urbanization, and are being replaced by using the best rangelands for cultivation. Unfortunately, there is no specific assessment of the amount of carbon lost due strictly to degradation.

Degradation has been classified from moderate to very severe (Figure 12), indicating an acute loss of biomass, both above ground and below ground, as well as significant soil degradation (Dregne *et al.*, 1991). The level of degradation is actually highest in North America (85%) and lowest in Australia (55%). There exists a potential of roughly 3,500 Mha for eventual restoration and carbon storage.

## Future Scenarios for Carbon Storage

Based on the area of the drylands that has been degraded, and the extent of degradation, it is reasonable to assume that actions to combat desertification can indeed increase the average net primary productivity of the woodland and grassland biomes, as well as the SOM production in the agricultural (irrigated and rainfed cropland) converted from these biomes.

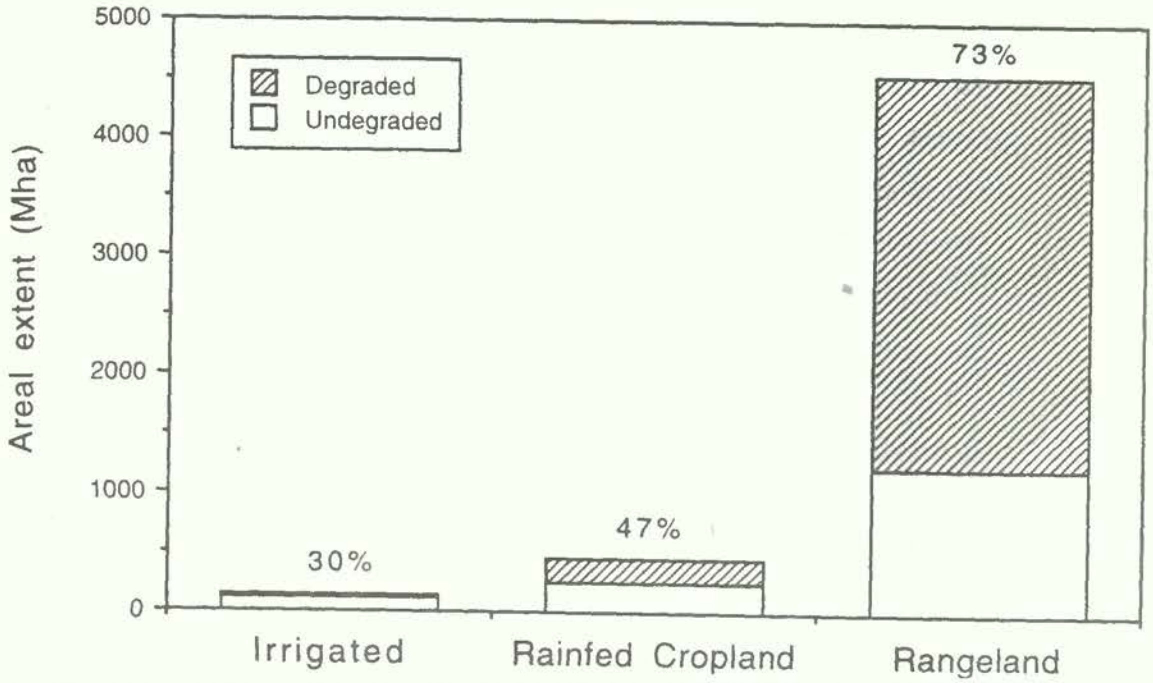


Figure 11. Estimated drylands area which has been degraded.

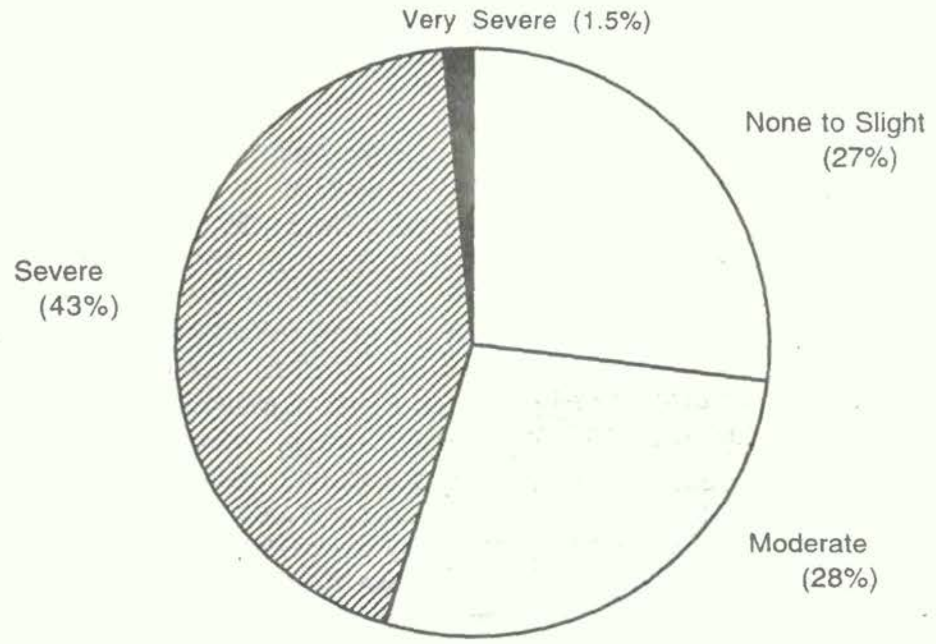


Figure 12. Extent of drylands degradation.

Whether the land that is currently cultivated can be converted back to the original biome may be difficult given the need for agricultural land in many developing nations. However, about 30% of cultivated drylands is in North America and Australia, which may be more easily reverted to grasslands and woodlands.

Three specific scenarios were assumed: 1) 8 Mha of cultivated land are abandoned yearly and revert to 5 Mha of woodlands and 3 Mha of grasslands; 2) the average NPP of both of these biomes is increased by 10%; and 3) the average SOM production in the agricultural component of these biomes is increased by 25%. In addition, all three actions were combined, to see what could be a reasonable upper bound for annual carbon storage in drylands.

The simulations were performed considering the Intergovernmental Panel on Climate Change (IPCC) 92d fossil fuel emissions scenario (IPCC, 1992), which assumes continuing growth of emissions but with some actions taken to reduce the rate of growth of the emissions. This will result in some carbon fertilization of the biomes. In addition, no other land use changes are considered, to reduce the complexity of the analysis. Land use changes could be allowed in other biomes, as they will most certainly occur in reality, but the net carbon emissions will probably be only a small fraction of the fossil fuel emissions, providing only a small additional carbon fertilization. Finally, since temperature is an important factor in most biome processes, it is assumed that the average net global increase in temperature is around 3 °C per $CO_2$ doubling; specifically for the woodlands the estimated increase is 2 °C per $CO_2$ doubling, and for the grasslands it is 4.5 °C per $CO_2$ doubling. The largest increase in temperature predicted by most Global Climate Models is in the higher latitudes, and we have considered this in the model according to the geographic distribution of the terrestrial biomes.

The effect of the IPCC 92d scenario alone is to increase the terrestrial biosphere storage of carbon by around 283 Gt C from 1990 to 2100 (Figure 13). Atmospheric carbon is predicted to rise by around 375 Gt C, or approximately 3.4 Gt C/yr. In terms of the three biomes of interest, the grasslands biome is predicted to store an additional 51 Gt C due to the carbon fertilization effect, whereas the carbon storage in the woodlands biome is practically unchanged (Figure 14). The small decrease in carbon stored in agriculture is the result of weathering processes.

The projected land use changes result in an additional 45 Gt C stored in the terrestrial biosphere, or about 0.05 Gt C per Mha reverted. This is more than was obtained in the response analysis due to the higher carbon fertilization of grasslands, as well as a larger temperature increase, since the IPCC 92d scenario results in a higher atmospheric $CO_2$ concentration. The increase is largest in the grasslands, for a net increase of 61 Gt C over the IPCC 92d scenario. The woodlands gain 56 Gt C. Due to the larger response of the grasslands to carbon fertilization, their net storage is greater even though the rate of conversion from agriculture is smaller. These gains are partially negated by the loss in the agricultural biome of 65 Gt C. Because of the gains in the drylands, the increased storage in the forests is diminished by about 7 Gt C.

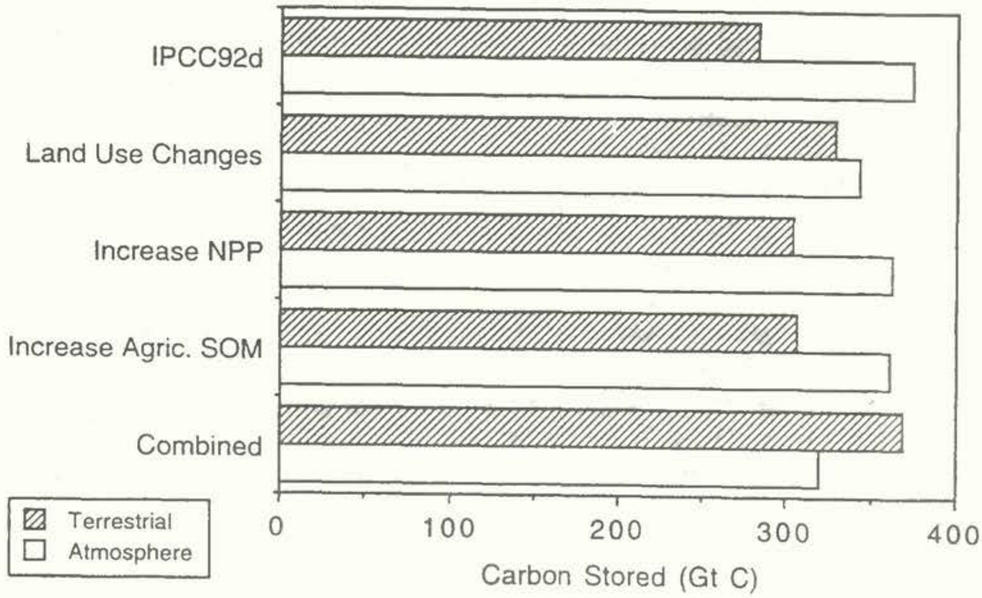


Figure 13. Potential storage of carbon in the terrestrial biomes and corresponding change in atmospheric carbon storage based on the future scenarios assumed. The base fossil fuel emissions scenario is IPCC 92d and the simulations are from 1990 to 2100.

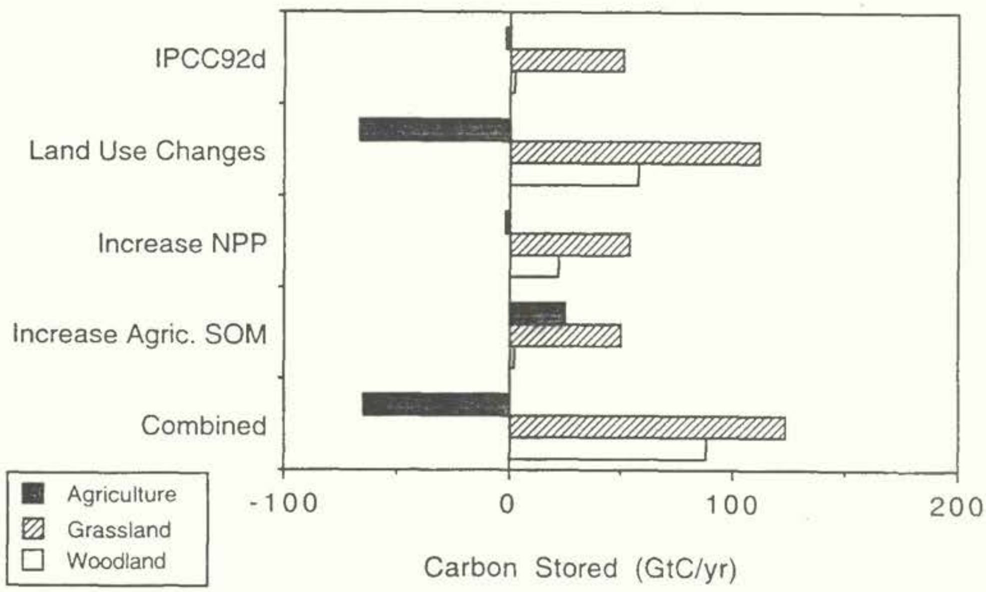


Figure 14. Carbon stored in agriculture, grassland and woodland for the future scenarios. Note that a significant fraction of the IPCC 92d fossil fuel emissions is stored in the temperate, boreal and tropical forests, as well as in the oceans (not shown).

The actual increase in atmospheric carbon is only 343 Gt C, or 32 Gt C less than the IPCC 92d scenario with no land use changes. This translates to an average reduction of 0.29 Gt C/yr in the rate of carbon buildup.

If the average NPP of the woodlands and grasslands is increased by 10%, with no land use changes, the additional carbon stored in the biomes relative to IPCC 92d is 21 Gt C, or around 0.19 Gt C/yr. Most of the gain is in the woodlands (20 Gt C) with minor additional carbon stored in the grasslands (3 Gt C). The resulting effect on the atmosphere is an incremental carbon storage of 362 Gt C, 13 Gt C less than the IPCC 92d scenario. The average reduction is 0.11 Gt C/yr. Considering the level of degradation of the drylands, the potential increase in NPP may be larger than 10%, with a corresponding greater carbon storage.

An increase in the average SOM production of 25% results in carbon storage in the terrestrial biosphere of 23 Gt C, roughly 0.21 Gt C/yr. The gain is essentially in the agricultural biome. The atmospheric increase is 14 Gt C less than the IPCC 92d scenario with no land use changes, for an average annual reduction of 0.12 Gt C/yr.

The combination of all three actions, land use change, increasing NPP by 10% and agricultural SOM by 25%, coupled with the IPCC 92d emissions results in a total increase in terrestrial biosphere carbon of 369 Gt C, for a net increase of 86 Gt C. The combined increase in biome storage is an average 0.78 Gt C/yr. Atmospheric carbon has a total increase of 319 Gt C, or a net reduction from the IPCC 92d of 56 Gt C. The annualized decrease in carbon buildup in the atmosphere is on the order of 0.5 Gt C/yr. Figure 15 summarizes the carbon storage in the terrestrial biomes and the reduction in atmospheric carbon buildup for the various scenarios.

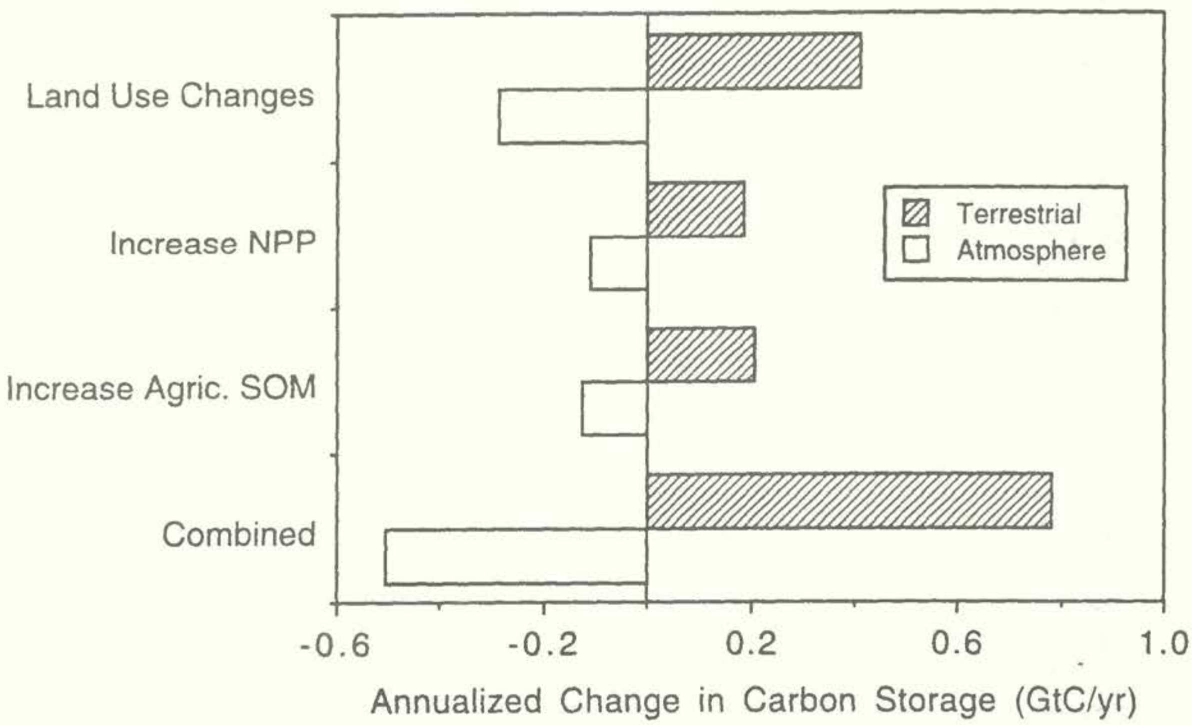


Figure 15. Annualized change in carbon storage in the terrestrial biomes and agriculture for the future scenarios.

## Discussion

The GLOCO model has been applied to explore the response of the woodland and grassland biomes, which comprise most drylands, to various land management actions which result in carbon storage. The model is particularly suitable for this type of analysis since it is designed to study land use changes, the effect of carbon and nitrogen fertilization, the changing climate (temperature) and the role that the various biome process parameters play on the global carbon cycle. In addition, since GLOCO couples the terrestrial and the oceanic models through the atmospheric model, it provides a more realistic oceanic buffering of the atmospheric response to land use changes, which could not be captured in a model only contemplating the terrestrial biosphere.

With over 3,500 Mha of drylands degraded, 27% of the terrestrial biosphere, the potential for storing additional carbon as biomass in these regions is significant. Actions taken to increase the net primary productivity of these biomes may result in storage of 0.2 Gt C/yr. Increasing the SOM production in the agricultural drylands may store an additional 0.2 Gt C/yr. And restoring marginal agricultural land to its original natural state can result in the storage of an additional 0.2 to 0.45 Gt C/yr in these biomes, for a total terrestrial biome storage of up to 0.8 Gt C/yr. This corresponds to a net decrease in atmospheric carbon buildup of 0.5 Gt C/yr.

Considering a fossil fuel emissions scenario with some control measures to reduce the rate of increase in emissions (IPCC 92d), the rate of carbon buildup in the atmosphere is around 3.4 Gt C/yr. The effect of storing carbon in drylands through their restoration could represent a reduction of around 15% in the total atmospheric carbon buildup. If dryland restoration is justified by a direct environmental benefit then the benefit of reduced atmospheric carbon comes at no cost. Another perspective is to state that the direct environmental benefit of restoration and the indirect benefit of atmospheric carbon reduction can together be used to justify the costs.

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# Modeling the anthropogenic degradation of drylands and the potential to mitigate global climate change

**Alan Grainger**

Synopsis

This paper shows the links between controlling desertification and mitigating global climate change. It evaluates previous global estimates of the area of desertified drylands and argues that better global monitoring of desertification is a top priority, not only for international efforts to control desertification but also for global environmental change research. The paper distinguishes between the physical suitability of degraded land for carbon sequestration and its actual availability, and reviews the lessons learned from previous experience about the constraints on the availability of desertified land for afforestation. It argues that present static geographic information system (GIS) techniques for suitability appraisal are inadequate to cope with the complex dynamic relationships between land use, degradation and climate change in dry areas. It suggests how to develop a dynamic GIS-based national spatio-temporal model of land use change in a country with extensive area of drylands. The model could be used to simulate future trends in desertification, the carbon emissions resulting from it, and changes in the area of physically suitable degraded land under the influence of desertification and global climate change.

Key Points

1.Desertification is the degradation of land in dry areas and has two main components: soil degradation and vegetation degradation. Both of these contribute to the net transfer of carbon from the biosphere to the atmosphere and raise greenhouse gas concentrations. Reducing the rate of desertification should therefore help to mitigate global climate change.

2.Desertification occurs primarily because changes in environmental and/or socio-economic conditions cause an area to become marginal for the land use, or the intensity of the land use, being practised there. Since global climate change could lead to substantial areas of the world becoming marginal for existing land uses, knowledge of the causes of desertification and how to control it will become very relevant when planning how to mitigate and/or accommodate to global climate change.

3.There are two main mitigation strategies in dry areas: (a) increasing carbon sequestration by revegetation; (b) conserving existing carbon stocks by controlling desertification. The dividing line between the two options is poorly defined because protecting the regeneration of existing vegetation should also raise biomass densities.

4.Degraded drylands have a lot to offer a future global carbon sequestration programme, particularly in expanding carbon sequestration, but estimates of the present area of desertified land are very inaccurate and still largely based on subjective expert assessments. A comprehensive global monitoring programme for desertification, which makes greater use of measured data and satellite imagery, is therefore essential both to improve the planning of desertification control programmes and to assess more accurately the potential to mitigate global climate change by controlling desertification and revegetating degraded drylands.

5.However, because human development and the influence of future global climate change will cause desertification to continue, even as attempts are made to control it and restore degraded drylands, static appraisals of the potential for desertification control and global climate change mitigation will be inadequate. New dynamic spatial models are therefore required to provide planners with the tools they need, and since these will also allow those modelling global climate change to take much greater account of the terrestrial component of global environmental change and terrestrial - atmospheric interactions, they should help to raise the status of desertification in global environmental change research and provide a greater incentive to obtain better data on its extent and rate of change.
Key Words: rehabilitation, socio economics, carbon sequestration, GIS, remote sensing, land use

## Introduction

Desertification is one of the major terrestrial components of global environmental change but has not received the recognition that it deserves. Far more effort has been directed to global climate change, even though it is still only a potential problem, while desertification is a problem now. Greater priority might be given to controlling desertification if it were placed much more firmly within the overall framework of global climate change mitigation. This is justified by carbon emissions into the atmosphere that result from dryland degradation and the role which restoring degraded drylands could play in mitigating global climate change by sequestering more terrestrial carbon.

This paper assesses the potential to restore degraded lands to sequester more carbon to mitigate global climate change, describes a GIS method to use the degree of vegetation degradation to estimate the physical suitability of land for restoration, distinguishes between physical suitability and actual availability in the light of social, economic and political constraints, and outlines a dynamic national model which could be used to simulate future trends in desertification, and the resulting carbon emissions and changes in the distribution of degraded land physically suitable for restoration.

## Desertification and climate change

### *Definitions*

Desertification has been defined as "land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities" (UNEP, 1995).

*Land degradation* has two main components: *vegetation degradation* and soil degradation. Vegetation degradation involves the "temporary or permanent reduction in the density, structure, species composition or productivity of vegetation cover" (Grainger, 1992). It may be vertical, as biomass density is reduced or ecosystem structure changed by human impacts, and/orhorizontal, as the spatial fragmentation of vegetation cover is increased.

*Soil degradation involves a deterioration in soil quality as soil is eroded,* compacted, waterlogged and/or salinised. Vegetation degradation often makes drylands more susceptible to soil degradation. Desertification does not refer to an abrupt transfer of productive land to desert, but to a spectrum of degradation from slight to severe, depending on the degree of vegetation and soil degradation. Only the most severe desertification is irreversible (Dregne, 1983).

*Desertification and Drought*

Desertification is intimately connected with climate change, although the nature of the connection is often confused. It was first recognized as a major global phenomenon in the 1970s when the organizers of the UN Conference on Desertification in 1977, convened in response to the Sahel drought, placed the problems that arose during that drought within the wider context of land degradation. For while drought is a normal feature of dry areas, leading to a temporary reduction in biomass, desertification is a long-term process of degradation. Drought is usually an indirect cause, or catalyst, of desertification as degradation accelerates when land use intensifies in drought periods.

*Desertification and Global Climate Change*

Major shifts in global climate, on the other hand, could cause desertification directly, although global climate change, like desertification is a long-term phenomenon. Until comparatively recently, the degradation of land that accompanies the expansion and intensification of human settlement was only thought to have relatively local external costs, e.g. in reducing landscape quality and the suitability of land for farming. But now degradation, and the net transfer of carbon into the atmosphere that usually results from this, is acknowledged as having global external costs because of the contribution it could make to global climate change through the greenhouse effect.

The occupants of drylands have had long experience of adapting to droughts and other natural hazards. There is greater scope to tackle the causes of anthropogenic hazards, like global climate change, in this case by reducing greenhouse gas emissions or sequestering more terrestrial carbon. Sharing the costs of mitigation in an equitable way will require taking into account a country's ability to pay, the opportunity costs it is asked to bear, and its historic contribution to global external costs through both land degradation and fossil fuel use (Grainger,1995a).

## The physical suitability of degraded lands for restoration

Drylands could play two roles in a world programme to mitigate global climate change. First, they could be afforested or otherwise revegetated to increase carbon sequestration. Second, action could be taken to control desertification and reduce carbon emissions. Most of this paper will focus on the first option. Lands where vegetation and soil carbon stocks are most degraded appear to have the greatest suitability for restoration to sequester carbon, although restoring them would also be physically challenging and costly. Those evaluating potential mitigation strategies have to assess the balance between their physical and socio-economic aspects as carefully as possible.

*The Potential for Carbon Sequestration on Degraded Tropical Lands*

Early assessments of carbon sequestration potential simply looked at the area of additional forest plantations needed to offset the rise in atmospheric carbon dioxide content. To fully sequester the entire current annual net increment in atmospheric carbon dioxide content from all sources, estimated as 2.9 gt C per annum in the 1980s (Detwiler and Hall, 1988), would require 465 million ha of new plantations, according to Sedjo and Solomon (1989). But they assumed a uniform plantation growth rate, and taking account of variations in growth rate between humid and dry areas - e.g. 15m3 per ha per annum in humid areas, 10m3 per ha per annum in moister savanna areas, and 5m3 per ha per annum in semi-arid and montane areas - suggests that 600 million ha would be needed instead (Table 1) (Grainger, 1990a).

However, even if 600 million ha of degraded land were indeed available, it would take a planting rate of 12 million ha per annum - twelve times the annual tropical afforestation rate for 1976-80 (Lanly, 1981) to cover it with forest plantations in 50 years. It would seem more feasible to opt for a lower planting target and at the same time bring the deforestation of closed tropical forests under control. If the present planting rate was tripled to 3 million ha per annum and the deforestation rate in the humid tropics cut to 0.9 million ha per annum by 2020, the carbon sequestered in tropical plantations in that year would offset carbon emitted from tropical deforestation (the Low Scenario in Table 2) (Grainger, 1990a). However, it is now appreciated that a major increase in tropical forest area can only be achieved by planting trees in ways that meet the needs of local people, e.g. by establishing agroforestry systems or restoring natural woodlands (Grainger, 1991).

*Controlling Dryland Deforestation and Forest Degradation*

The above simulations assumed that deforestation rates in dry forests continued unchanged between 1980 and 2020, because estimates of the rates were so inaccurate, and it was difficult to simulate future trends in deforestation in the same way as for tropical moist forests. But biomass densities are much lower in dry forests than in most forests, e.g. mean values of 33.5 tC/ha for open forest and 55 tC/ha for dry closed forest in Africa (Brown and Lugo, 1984) , compare with 100-200 tC/ha in tropical rainforest. According to one estimate, the deforestation of dryland forests only accounted for 12% of all carbon emissions from tropical deforestation in 1980 (Grainger, 1990a), and given the great variation in tree densities in open forests (Unruh et al, 1993) this could be an overestimate.

However, the above estimates of emissions refer only to those from vegetation degradation, and owing to the substantial carbon stocks in dryland soils, the overall impact of controlling deforestation would be much greater than indicated here. In addition, because of the great difference between the biomass densities of degraded open forests and their potential biomass densities, protecting their regeneration would sequester a lot of carbon, although it will be difficult to protect large areas of open forest from continuing degradation.

Table 1. Afforestation scenarios to achieve a range of reductions in present annual atmospheric carbon dioxide increment over a fifty year period Source: Grainger 1990a

| $CO_2$ Increase Reduction (%) | Total Planted Area (million ha) | Target Planting Rates (million ha per annum) |
|---|---|---|
| 100 | 600 | 12 |
| 50 | 300 | 6 |
| 42 | 250 | 5 |
| 25 | 150 | 3 |
| 10 | 60 | 1 |

*Table 2: Net carbon emissions in 2020 after combining carbon uptake by new tropical forest plantations and carbon emissions from tropical moist forests.*

| Planting Rate# (million ha per annum) | Net Carbon Emissions in 2020 (gt C per annum) | |
|---|---|---|
| | High | Low |
| 12 | -1.269 | -1.499 |
| 6 | -0.370 | -0.599 |
| 5 | -0.220 | -0.449 |
| 3 | 0.080 | -0.150 |
| 2 | 0.230 | 0.000 |
| 1 | 0.380 | 0.150 |

NB. The tropical moist forest deforestation rate declines to 3.7 million ha per annum (High Scenario) and 0.9 million ha per annum (Low Scenario) in 2020. Deforestation rates in dry areas stay constant at 1.2 million ha per annum (closed forests) and 3.8 million ha per annum (open forests). Negative net emissions indicate net sequestration

## The Area of Suitable Degraded Tropical Land

The first requirement before mounting a world forest-based carbon sequestration programme is sufficient degraded tropical land to rehabilitate. In the 1980s there were an estimated 2,077 million ha of degraded tropical lands (Grainger, 1988), of which most - 1,651 million ha was desertified dryland (Table 3). But Grainger (1990a) concluded that in dry areas successful afforestation was only possible on 331 million ha of degraded rainfed and irrigated croplands - most desertified rangelands were too dry and the area of degraded moist rangelands was not known very accurately. After removing logged forests from the 758 million ha of land with potential for forest replenishment - because they are best left to regenerate naturally - this left 621 million ha suitable for afforestation, which satisfied the above target. Source: Grainger (1988)

Table 3. Area of Degraded Tropical Lands in 1980 with Potential for Forest Replenishment (million ha)

| | Degraded Lands | |
|---|---|---|
| | All | With Potential for Replenishment |
| Forest fallows | 203 | 203 |
| Logged forests | 137 | 137 |
| Deforested watersheds | 87 | 87 |
| Degraded drylands | 1,651 | 331 |
| Total | 2,077 | 758 |

*Global Estimates of the Area of Desertified Drylands*

Drylands clearly account for a significant proportion of all degraded tropical lands with potential for forest replenishment. But estimates of the area of degraded tropical lands have had to rely on estimates of the extent of desertification that were acknowledged as very inaccurate even by those who produced them. The area of degraded dryland in Table 3 is based on an estimate by Mabbutt (1984) that of the 4,700 million ha of hyper-arid, arid, semi-arid and sub-humid lands, 2,000 million ha were moderately or more severely desertified, and of this 1,651 million ha were in the tropics. The figure of 2,000 million ha excluded slightly desertified areas in natural deserts, where natural biological productivity was too low to support cultivation or grazing and human impact was minimal, and large areas of remote, unwatered land, classified as rangelands but seldom used as such. Including the latter would raise the total desertified area to 3,200 million ha (Dregne, 1983).

More recently, the UNEP World Atlas of Desertification (UNEP, 1992) estimated that 1,035 million ha of drylands suffered from soil degradation in 1991. This was only half Mabbutt's previous estimate for at least moderately desertified land but took no account of vegetation degradation. The main source of data was the Global Assessment of Soil Degradation (GLASOD) made by UNEP and the International Soil Reference and Information Centre (ISRIC)., based on "a compilation of existing information and of expert knowledge made available by more than 250 soil and environmental experts worldwide..." So a high degree of subjectivity still remained. In a subsequent assessment of the extent of desertification, UNEP (1991) revised the Atlas estimate of the extent of soil degradation from 1,035 to 1,016 million ha, estimated that 649 million ha suffered from at least moderate degradation, and by including a further 2,576 million ha of rangelands suffering from vegetation degradation but not soil degradation, raised the total area of desertified drylands to 3,592 million ha. The equivalent figure to the area of suitable degraded dryland used in the 1988 assessment in Table 1 - the area of at least moderately desertified rainfed and irrigated cropland - was 213 million ha.

Lack of data on vegetation degradation in UNEP (1992) might have been offset by FAO's recent assessment of tropical forest resources for 1990 (FAO, 1993). In 1980 FAO estimated the area of the open forests, the characteristic vegetation type found in drylands, as 734 million ha (Table 6) (Lanly, 1981). However, the data categories in FAO (1993) were not compatible with those in Lanly (1981), as All Tropical Forest was not divided into Open and Closed Forest as before. The area of "Dry and Very Dry Forest" was given as 246 million ha but this seems rather low, and because of classification problems was not reliable. A best estimate of the area of open forests in 1990 made using the FAO (1993) data was 649 million ha (Grainger, 1995b). FAO (1993) revised downwards the 1980 estimates in Lanly (1981), so not all the 85 million ha "loss" of open forest from 1980 to 1990 constituted deforestation.

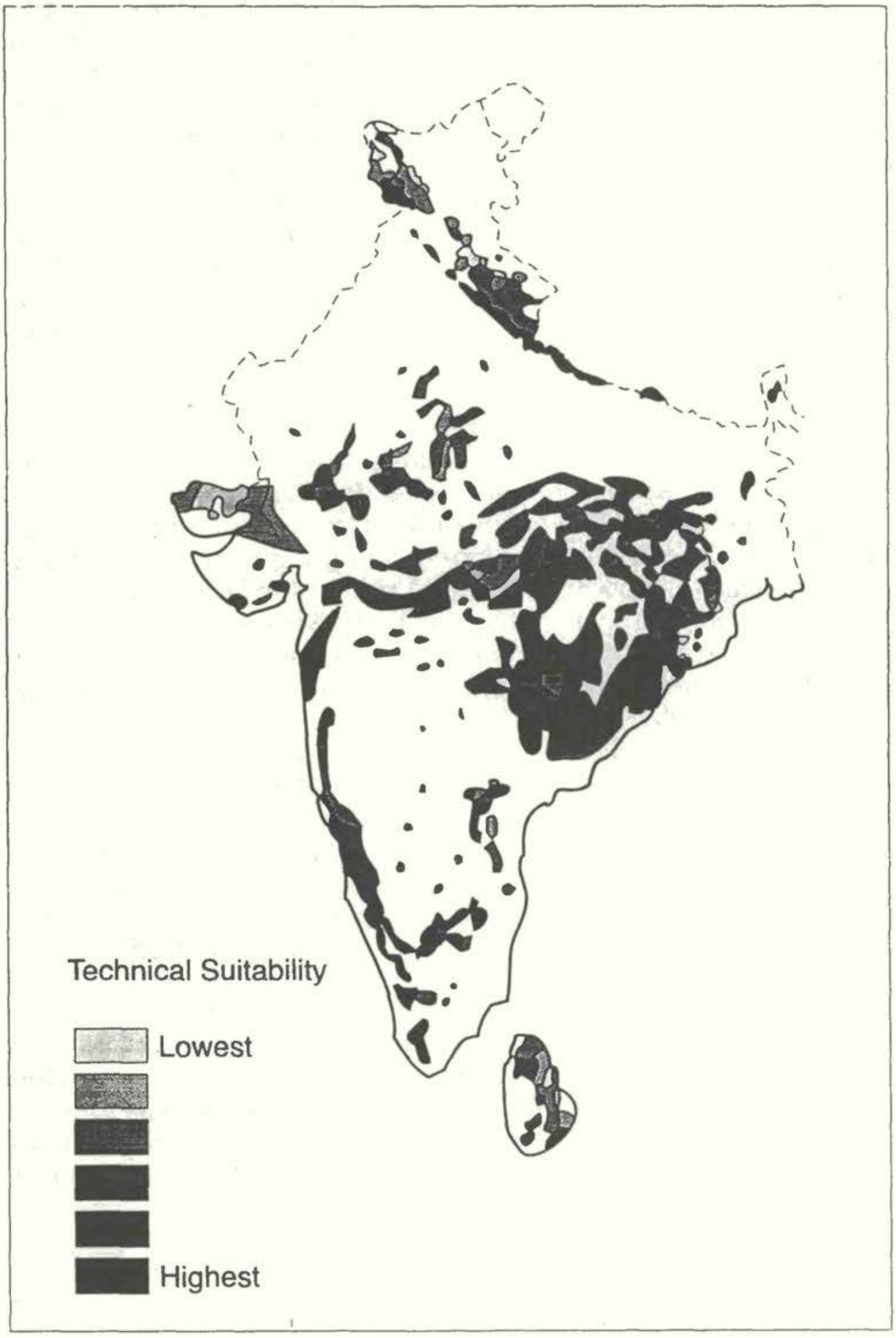
*Estimating the Distribution of Degraded Vegetation in Asia*

During the 1990s, various initiatives have been taken, under the auspices of the Intergovernmental Panel for Climate Change, to obtain improved estimates of the area of degraded tropical lands. In one study, Iverson et al (1993) used satellite imagery and geographic information system (GIS) techniques (Burrough, 1986) to assess the physical suitability of degraded forest lands in continental South and Southeast Asia for carbon sequestration, based on the degree of vegetation degradation. The region includes dryland areas and the method could be used to assess dryland degradation generally.

Physical (or technical) suitability was assumed to be proportional to the difference between potential and actual biomass density, i.e. the degree of vegetation degradation. Potential biomass density was estimated as a function of precipitation, temperature, soil texture, soil quality, elevation and slope, and represented by a potential carbon sequestration index (PCSI) on a scale of 1 to 100. Actual biomass density was represented by an actual carbon sequestration index (ACSI) on a similar scale, and estimated in two ways: (a) by a model based on climate and settlement data; (b) by a global vegetation index derived from low-resolution satellite data (the two estimates showed good correlation).

Digital maps of PCSI and ACSI for remaining open and closed forests in continental South and Southeast Asia were produced using GIS software and digital maps of environmental variables and forest cover (the latter was an FAO map based mainly on visual interpretation of Landsat satellite imagery).

Figure 1. Technical suitability for degraded forests in continental south and southeast Asia for carbon sequestration



The degree of degradation was estimated by subtracting the distribution of ACSI from that of PCSI using the GIS. The resulting map of was divided into six degradation classes (Figure 1). The lower the class, the less degraded the forest and the lower its technical suitability for carbon sequestration. Forest biomass density in the lowest class was 50 Mg C per ha lower than its potential value, and that in the highest class was over 250 Mg C per ha lower.

Two thirds of all forest in the region could sequester a further 50-150 Mg C per ha, and so make a significant contribution to climate change mitigation (Iverson et al, 1993). The degree of degradation of drylands in the study region appeared to be relatively moderate: only 15% of existing forest in dry lowland areas and 22% of existing open forest (the two categories overlapped but were not identical) were in the top three suitability classes. But in India, which contains a high proportion of the dry land in the region, 39% of all forest was in the top three classes (Table 4). These results should be treated with caution, given the inaccurate maps of present forest cover and land use, and the errors involved in estimating ACSI and PCSI and using regression equations to predict their regional distributions.

Table 4. Degree of Degradation of Present Forest Lands in Continental South and Southeast Asia, in Dry Lowlands, Open Forests and India (million ha)

| | Dry Lowlands | Open Forests | India |
|---|---|---|---|
| Very Low | 1.1 | 8.3 | 9.6 |
| Low | 2.9 | 9.7 | 12.4 |
| Medium | 6.1 | 12.1 | 24.0 |
| Medium High | 1.6 | 6.6 | 20.3 |
| High | 0.1 | 1.7 | 8.5 |
| Very High | 0.1 | 0.4 | 0.6 |

Source: Iverson et al (1993)

Degradation of dry forests could also have been systematically underestimated, and estimation of both potential and actual biomass density is made particularly difficult in dry areas by the climatic fluctuation which is such a characteristic feature of them. However, the study was the first of its kind to use spatial data bases and appropriate models to estimate the large-area distribution of degraded forest lands.

*The Need for Improved Monitoring of Dryland Degradation*

Given the important role of land degradation in global climate change, it is vital to monitor it more frequently and accurately than at present. Better data on vegetation degradation in drylands is urgently needed to produce a more comprehensive global assessment of desertification. When Dregne (1977) defined desertification as: the impoverishment of arid, semi-arid and some subhumid ecosystems by the combined effects of man's activities and drought. It is the process of change in these ecosystems that can be measured by the reduced productivity of desirable plants, alteration in the biomass and the diversity of the micro and macro fauna and flora, accelerated soil deterioration and increased hazards for human occupancy. he was clearly treating desertification as a quantifiable phenomenon. There have been discussions about possible sets of indicators since 1977, but despite the detailed evaluation of criteria for indicator selection (Mabbutt, 1986), these discussions have still not reached a conclusion.

Bearing in mind the continuing queries about whether desertification is a real phenomenon or just a myth (Binns, 1990; Thomas and Middleton, 1994), the sooner that desertification can be defined in terms of measurable indicators, and comprehensive monitoring begins, the sooner it will take its rightful place within global environmental change research.

## From physical suitability to actual availability

### *The Limitations of GIS Assessments*

Land that appears from GIS assessments to be physically suitable for revegetation may not be actually available for this purpose. Much of the planet's natural vegetation cover has been transformed or modified to satisfy human needs for food, wood etc. So the actual biomass density in an area may well be less than the potential biomass density of its climax ecosystem but is not degraded in relation to cultural or commercial norms, i.e. the biomass density of a well managed forest or rangeland. Even if it is, social, economic and political constraints may still limit availability.

### *Constraints on Actual Availability for Afforestation in Dry Areas*

A number of lessons about actual availability can be learned from past experience with afforestation schemes designed to combat desertification (Grainger, 1990b):

1. Government forestry personnel are usually too limited in numbers and facilities to manage large areas of dispersed forest or engage in large-scale tree planting.
2. In such circumstances, mobilising rural people to plant trees in their own areas - social forestry - seems the best solution.
3. However, many rural people are wary about participating voluntarily in social forestry schemes on communal lands for which the distribution of the rewards is uncertain. They will participate if they are paid and either the government or a non-governmental organization bears part of the risk.
4. They may also be unwilling to plant trees on their own lands if this places their crops and earnings at risk, or they fear that planting trees will give someone else rights over their land. Tenant farmers feel particularly vulnerable to landlords taking back their farms if they have improved them by planting trees.
5. Larger farmers usually lead the way in social forestry programmes by planting on their own lands. Smaller farmers follow later when they see that the system works.
6. Communal tree planting in dry areas tends to happen in response to acute threats, e.g. encroachment of sand dunes on a road or village, or damage by strong winds on villages or crops.
7. The success of social forestry schemes varies greatly between countries. In India moderate success has been achieved, but in Africa progress has been slow.
8. Local people will be unwilling to commit themselves fully to any project if they have not been consulted properly beforehand, e.g. about how a community forestry project will affect their individual property rights. They

may appear to go along with the project, but when it ends they may do all they can to destroy its achievements, either directly, e.g. by cutting down trees, or indirectly, e.g. by failing to protect them from livestock browsing.

9. Forestry projects often fail if intended to satisfy either an external need, or a local need determined by outsiders that does not correspond to needs perceived by local people. All too often, forestry projects have been established to satisfy a perceived shortage of fuelwood, when the real local priority is more fodder for livestock. It is unlikely that afforestation projects funded as part of a worldwide global climate change mitigation programme will receive a more enthusiastic welcome from rural people than anti-desertification schemes have in the past. Forest-based mitigation projects must be designed to meet real local needs, not those imagined by outsiders, if they are to be successful. Technologically-oriented strategies, like fast growing plantations, may be less successful than strategies which build upon existing local practices, such as the management of natural open forests.

*Spatio-Temporal Trends in Availability*

The above constraints can lead to various spatial and temporal delays in implementing social forestry projects and these will also prevent a global carbon sequestration scheme from meeting its targets immediately. Four types of delay can be identified which effectively result in actual availability changing in space and time.

1. Organizational Delays. These delays occur when transmitting policy from central government to foresters on the ground. Intermediate layers of bureaucracy within the forestry department and other government organizations, under pressure from local interest groups, can influence the speed of policy implementation.

2. Motivational Delays. Social forestry programmes to achieve forest protection or afforestation with the voluntary participation of local people cannot start before a preliminary programme has created awareness by educational and publicity campaigns and convinced people to participate, and this takes time.

3. Socio-Economic Delays. As mentioned above, there will also be a lag in the socio-economic status of participants as projects begin with a few of the larger landholders and spread gradually to the larger number of smallholders.

4. Spatial Diffusion. Adoption of afforestation and forest management practices practices usually begins in one or two locations and then spreads from there. This is necessary for simple logistical reasons, but also reflects the fact that most successful large-scale afforestation schemes rely on the power of example.

Constraints on reducing emissions caused by desertification

Controlling desertification and reducing the carbon emissions resulting from it would also mitigate global climate change. It is technically possible to control desertification, e.g. by reducing overcultivation, overgrazing and deforestation, and some progress has been made (Grainger, 1990b), but various obstacles stand in the way, including: (a) the fact that some of the underlying causes of desertification are far away from where it appears most acutely; (b) the complexity of environmental management in dry areas.

*Spatio-Temporal Desertification Processes*

Desertification in its most acute form may appear to be localised in arid rangelands but this is often the result of a sequence of land use changes taking place on a national scale in response to population growth, economic development, government policies and other underlying causes. Population growth increases demand for food and settlement areas, leading to the expansion of farmland and a rise in population density in some areas.

Economic development and government policies promote the cultivation of cash crops for export (such as groundnuts), the expansion of the market economy at the expense of traditional subsistence economies, and urban growth. The driest areas in any country tend to be economically an politically peripheral in relation to the core area, and their inhabitants are least able to invest in more productive and sustainable agriculture on. marginal lands and counter pressures from the core.

Four spatial desertification processes have been identified that link land use change with its underlying causes: expansion, confinement, displacement and commons failures (Grainger, 1992).

**Expansion.** In this process, rising demand for food production in poor tropical countries is supplied by increasing farmland area rather than yield per hectare, because of a general inability to invest in more productive agricultural practices. This has been very evident in Africa in recent decades. Much of the expansion tends to occurs on marginal land which can soon become degraded.

**Confinement.** Sometimes, however, expansion is impossible, so land use has to be intensified within a limited area, resulting in the process of confinement. There are various examples of this: (a) the expanding "grey halo" of deforested land around towns, as trees are cut down for fuelwood and charcoal to supply the energy needs of urban dwellers; (b) degradation of pastures around villages and near to boreholes along nomadic herd movement routes, caused by overgrazing; overgrazing of marginal rangelands, when nomadic pastoralists are excluded from moister areas by expansion of other land uses or the imposition of management schemes.

**Displacement.** As national land use changes, land may be reallocated to more productive uses. If export cash crop cultivation expands in wetter areas where it is more productive and profitable, subsistence or commercial rainfed cropping tends to be displaced onto lands that are marginally suitable for it, and nomadic pastoralism is displaced in turn onto even more marginal lands. Overcultivation on marginal croplands and overgrazing on marginal rangelands leads to desertification, the latter often being the most noticeable, but the original cause of the overgrazing was a change in land use far away in more humid areas over which pastoralists had no control.

**Commons Failures.** For millennia, nomadic pastoralists managed common property rangeland resources using various "social control" mechanisms. Today, however, these traditional mechanisms have disintegrated with the encroachment of the market economy and the decline of nomad cultures and attempts by governments to introduce "rational" range management schemes in the Sahel as substitutes have failed. The real "tragedy of the commons", as many herders exploit limited grass resources, is more of a cultural tragedy than a failure of management technique, since previously, management mechanisms and cultures were interlinked. If governments and

development agencies persist in viewing desertification as something that only happens on the fringes of deserts, and ignore the underlying causes of the problem and the influence of changes in land use
on a larger scale, then desertification will remain difficult to control. Since it will take time to overcome these limitations, and the centres of power in a country will be very loath to concede power to the periphery, it will be difficult to rapidly reduce carbon emissions associated with desertification.

*Constraints on the Sustainability of Open Forest Management*

Another requirement for curbing carbon emissions caused by desertification is to curb the rate of deforestation in open forests by managing them more sustainably. However, this will not be straightforward, for two main reasons.

First, in contrast to closed forests, which are often managed for a single use, i.e. timber, open forests are usually managed for multiple uses, e.g. harvesting timber, fuelwood and fodder from the trees, and growing crops and grazing on the ground beneath them. Sustainability can never be assessed in terms of the yield of one product, because the mix of products is always changing in response to local needs.

Second, open forests are "managed" not by a single user, as when a government forestry department or its appointed concessionaire manages a closed tropical forest, but by multiple users. In some areas, where traditional social control mechanisms still operate, users work toward a single, agreed end. Elsewhere, they are in conflict, as local users disagree among themselves, or with outsiders, e.g. pastoralists or fuelwood gangs, who are only concerned with short term exploitation. In the light of this, a new concept of sustainable land management is needed for open forest lands.

Sustainability is determined by both the physical attributes of the forest, and the effectiveness of local and national institutions in managing it. But weak government control, resulting from the poor commercial timber content of these forests, and the decline in traditional social control, leaves a management vacuum. In the 1980s there was a major shift from just establishing plantations in dry areas to managing natural open forests, which had hitherto been disregarded. Rural people depend heavily on such forests for food, fodder, fuelwood, medicines and a variety of other products, even though the forests appear to outsiders to be degraded and unproductive. Attempts to replenish and manage them to meet local needs have therefore met with enthusiastic responses, although coping with multiple uses and multiple users does present major management challenges (Grainger, 1990b).

## A dynamic model of desertification and restoration suitability

A static appraisal of the suitability of drylands for restoration will not give reliable results because, even as attempts are made to control desertification and restore degraded lands, anthropogenic degradation will still continue. In addition, as noted above, spatio-temporal variations in precipitation influence estimates of potential and actual biomass density, and revegetation and degradation trends. These influences will even become more complex if the world's climate changes. So to fully estimate the potential for net carbon sequestration requires a dynamic GIS model that combines: (a) a spatio-temporal appraisal of physically suitable and actually available lands and their biomass density increments after restoration; (b) a spatio-temporal model of desertification processes driven by socio-economic variables; and (c) the effects of global climate change. This section outlines a dynamic national model which could be used for this purpose.

*Modelling Land Use Change*

It was argued earlier that land use change in dry areas can only be understood within the context of land use change over a much wider area, preferably within whole countries. Initial work in modelling the long-term evolution of the broad pattern of national land use - national land use morphology (Grainger, 1995c) - has focused on countries with relatively homogeneous environments, such as those located entirely within the humid tropics (Grainger, 1990c).

Modelling the evolution of national land use morphology in countries that have substantial areas of drylands will be more challenging, because they have far more heterogeneous natural environments, often with humid areas to which the drylands are economically and agriculturally peripheral. It seems fair to assume that much of the desertification occurring today in the dry areas of developing countries results from national land use morphologies changing in response to underlying socio-economic driving forces - growing populations and growing economies - but doing so under the limitations imposed by heterogeneous environments that are highly vulnerable to drought and very susceptible to degradation.

Developing countries in more humid areas also suffer from significant environmental degradation as they develop, but the degradation appears mainly in the form of tropical deforestation. In countries with more heterogeneous environments some deforestation will occur but soil and vegetation degradation will exhibit more complex patterns.

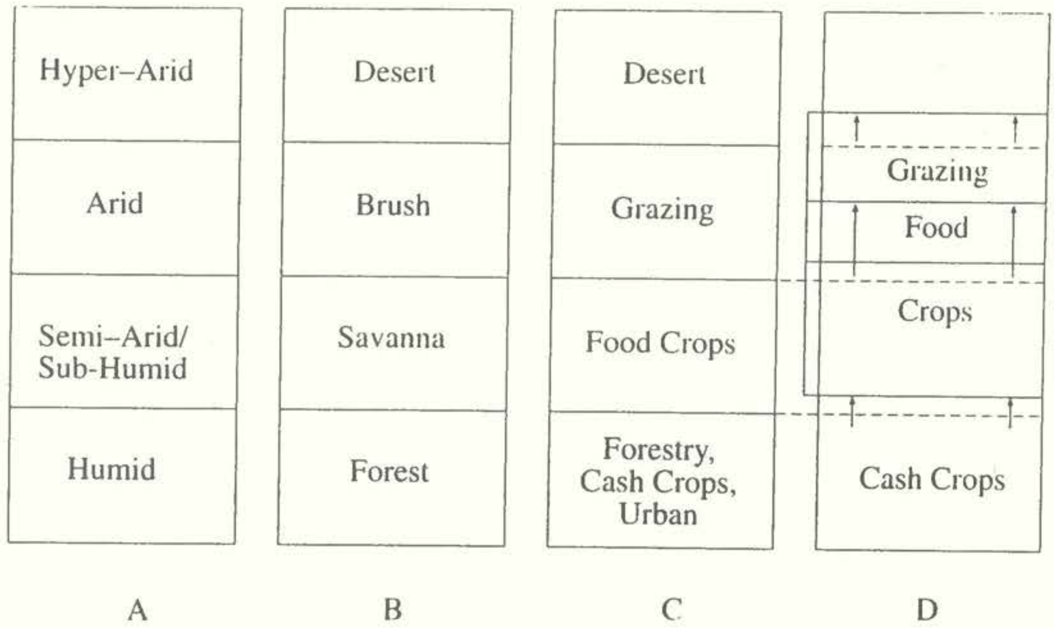
Models of large-scale land use change are inevitably highly aggregated, and do not aim to predict what the land use and degradation will be in particular locations at particular times. But they could assist policy makers in assessing the possible consequences of changes in development policies; they could enable scientists to test hypotheses about desertification by predicting where it might occur given certain assumptions and comparing these predictions with what actually happens on the ground; and they could also allow land use change and the effects of global climate change to be incorporated in global climate models, instead of being largely ignored as has been the case until now.

*Modelling Land Use Change in a Heterogeneous National Environment*

The basic principles of a simple model of land use change in a country with extensive areas of dryland would be similar to those used to model land use change in a country with a fully homogeneous environment, e.g. one wholly within the humid tropics, with uniform soils and no significant variation in topography, and in which all areas are equally suitable for all land uses and equally susceptible to soil degradation. Its basic socio-economic driving forces would be population growth, economic development and government policy, which influence demand for food, non-food crops and other land uses. This would be translated into demand for land, met by transferring land from the forest sector to the agricultural and urban-industrial sectors.

The location of forest clearance would be decided purely by proximity to existing human settlements, to minimise transport costs.

Figure 2. Modeling changes in the National Land Use Morphology of a country with a heterogeneous precipitation distribution: (A) Climatic zones ; (B) Biomes; ( C) Original national land morphology; (D) Modified national land use morphology, following the expansion of cash crops and food crops zones, displacement of food crops and grazing zones and contraction of grazing zone.



To show how this approach could be adapted to a country with a more heterogeneous precipitation distribution, the national land use morphology of such a country is shown in Figure 2. For convenience, it is based on a typical country in West Africa, in which rainfall declines from south to north (Fig. 2A), and there is a corresponding sequence of biomes, from moist forest to desert (Fig. 2B).

National land use morphology has already evolved for some time, becoming divided into four zones: (a) intensive food crop and export cash crop cultivation and forest; (b) rainfed food crop cultivation; © pastoralism; (d) desert (Fig. 2C). It is assumed that urban settlements are concentrated in the south of the country, as is common in West African countries, most of which have primate cities and are poorly urbanised (Haggett, 1983). Because in practice open forests (savanna woodlands) are used for multiple purposes, they are not allocated a separate zone but assumed to be part of the rainfed cultivation and pastoral zones, and possibly the forest component of the intensive cultivation zone as well.

Using the spatio-temporal desertification processes described above, suppose that the following sequence of land use change occurs:

1. Economic development and government policy lead to the expansion of export cash crop cultivation, and the growth of the single urban centre. This displaces rainfed food crop cultivation towards drier areas.
2. Population growth leads to more demand for food, which is satisfied principally by an expansion in the area under rainfed food crop cultivation. (The rise in demand may be reduced by using some of the revenue from cash crop exports to pay for food imports.)
3. The combined expansion and displacement of rainfed food crop cultivation displaces pastoralism onto more marginal lands.

The result of the entire sequence is shown in Figure 2D, with the expansion and displacement of rainfed food crop cultivation shown separately for convenience. These changes could be generated by interfacing a mathematical model to a GIS data layer showing the distribution of land use. The model would incorporate the underlying driving forces of population growth, economic development and government policy, and include algorithms linking these to changes in demand for food, non-food products and land, as in Grainger (1990c).

An actual model would obviously be more complex than in Figure 2. It would contain separate demand functions for export cash crops and domestic food crops. The need for additional land for each type of farming would be simulated according to alternative assumptions about the annual rise in productivity. The ability of each of the four main zones of national territory to supply demand for extra land would be assessed in terms of:

(a) the availability of spare land (e.g. land still forested in the humid zone);
(b) the willingness of farmers in the humid and semi-arid/sub-humid zones to convert from food cropping to cash cropping, reflecting the inertia of land uses to displacement.

The division between food crop cultivation and export cash crop cultivation is also not as strict in practice as implied above, and this will increase the difficulty of modelling displacement. The same would hold true for livestock raising, which has traditionally depended on grazing fallow areas on farms in the dry season.

The balance between the need for new land and its availability would determine the extent to which land uses that would otherwise expand or be displaced are instead confined to limited areas. The model could also take account of possible increases in the productivity of food crop cultivation obtained by investing some of the income from cash crop exports, and changes in demand for domestic food production as export crop prices vary.

*Modelling Degradation*

Land becomes degraded when the intensity of a particular land use exceeds the capacity of the land to support that use at that intensity. So the degradation of land likely to result from land use changes predicted by the above model could be predicted by comparing the actual distribution of land uses with a national land capability classification. This shows the boundaries within which each land use can be practised sustainably at a given intensity, and the physical and economic margins beyond which it is not sustainable and likely to degrade soil and vegetation.

Strictly speaking, the latter refers to what is generally known as a land suitability classification (Carpenter, 1981), but capability is used here instead to avoid confusion with the suitability of land for restoration. The land capability classification could be generated on a national or regional scale by applying a rule-based model to a similar set of GIS environmental data layers to those used to estimate suitability above, e.g. precipitation, temperature, length of growing season, soil type, soil texture, soil quality, elevation and slope. Alternative scenarios could be produced, as appropriate, to take account of varying technical inputs, e.g. fertilizer, irrigation etc., as in Higgins et al (1983).

To see how the model might simulate spatio-temporal trends in land use and degradation, assume for convenience that the initial distribution of land use coincides with the ideal land capability classification, so that no degradation occurs (Fig. 3A). When land use changes as in Figure 2, rainfed food crop cultivation and pastoralism both exceed their margins, degrading land in the super-marginal areas (Fig. 3B). The location of degradation would be estimated by using a GIS to compare the new land use distribution with the land capability classification.

The degree of degradation could then be estimated more precisely by devising separate algorithms to estimate the impact of over-intensive land use on soil and vegetation and using these to interface the derived degradation layer with soil and vegetation data layers. It would be simple to adapt the model for countries containing other forms of environmental heterogeneity, e.g. hilly areas, by including data layers for slope and elevation and using these to estimate possible rates of the erosion of soil by water etc.

However, land degradation is already extensive in many countries. To apply the model to real conditions would therefore require relaxing the initial assumption that current land use is sustainable because the land use distribution matches the land suitability distribution. Currently degraded areas would need to be included in the model, their suitability for different land uses assessed in common with other areas, and their degree of soil and vegetation degradation estimated as carefully as possible given the availability of data and environmental variability. The land use zonation employed here is based, for simplicity, on the physical margin for the viability of a given land use. This will differ from its economic margin but seems a reasonable approximation given the scale and aggregation of our modelling approach.

The zonation appears to be similar to Von Thunen's zonation of land use in an isolated city state, in which the bands of land closest to the city were used for intensive dairying/market gardening and forests (for fuelwood), and those further away were used for extensive cultivation and extensive livestock grazing. But this zonation was determined by each land use's locational rent, based on its market price, production cost and transport cost to market. As Haggett (1983) showed, Von Thunen did not take account of the effect of environmental heterogeneity on land use distribution. Von Thunen "rings" will be evident around cities in dry areas, but will be mainly subsumed within the larger banding pattern of national land use morphology, which in any case aggregates subsistence and commercial agriculture.

Caution is needed when using land suitability classifications in dry areas, as they are subject to highly variable rainfall. Rainfall is only one of the factors determining the suitability of an area for different land uses, but it is a vital one. Because of variable rainfall the physical and economic margins within which a given land use is sustainable at a given intensity will vary from year to year, and this needs to be incorporated in the model. Moreover, the use of "scientifically determined" carrying capacities for arid rangelands has been shown to be highly misleading (Grainger, 1990a).

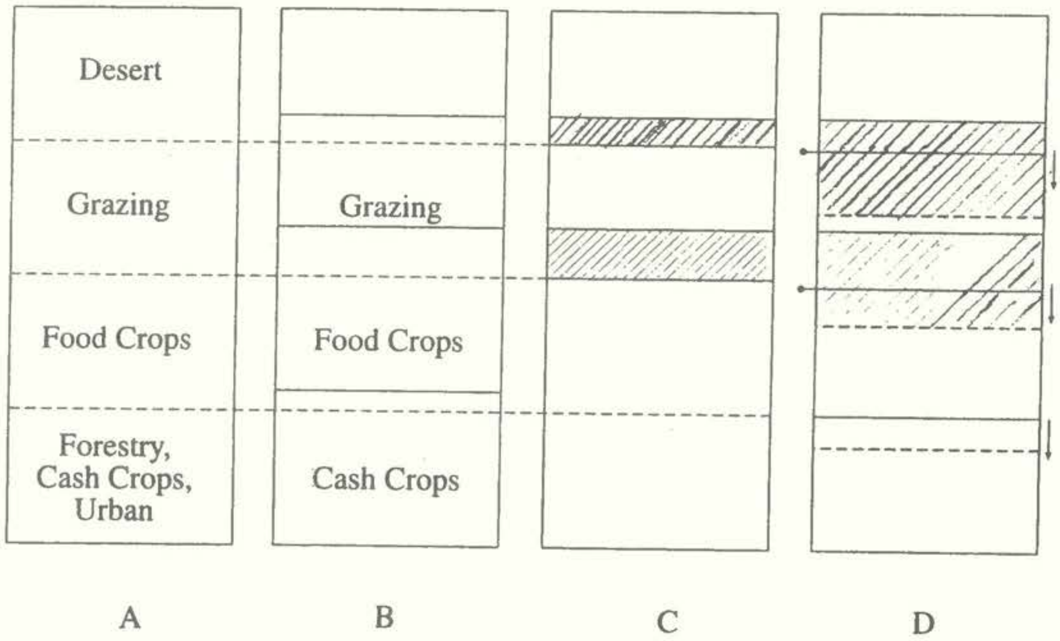
The model would also have to take account of the general intensity and sustainability of land uses in different areas. In some areas that are clearly marginal for a displaced land use, over-intensive land use leads very quickly to degradation. In areas where land uses are confined by external circumstances and forced to become more intensive, land managers may follow the Boserup (1965) route and ensure that greater intensity is sustainable in the long term, or the alternative Blaikie and Brookfield (1987) route and accept degradation as long as they produce enough food to feed themselves in the short term. There is a danger that this kind of modelling might be seen as environmentally deterministic. That is not the intention.

Indeed the whole aim of the model is to show how the expansion of land use in response to socio-economic driving forces can lead to degradation when the expansion encounters environmental constraints. These constraints are still very important in areas where farmers are too poor to invest in fertilizers and other artificial inputs, but alternative scenarios could be simulated to show what might happen if more fertilizer and irrigation were used.

*Modelling the Effect of Climate Change*

Goudie (1994) has warned that "the effects of global warming may, in critical areas, compound the most serious consequences of current human activities" and this is certainly true in the case of desertification. There is great uncertainty about how precipitation is likely to change in present dry areas but if, for example, the country in Figure 2 suffered a net decline in rainfall, rainfall bands would move southwards. The northerly boundaries of land uses would probably follow, but social pressures might still cause them to exceed the capability margins (Fig. 3c). This would result in degradation between the former margin and the new one, to add to that which previously occurred above the former margin (Fig.3d).

Figure 3. Modeling land degradation following changes in the National Land Use Morphology of a country with a heterogeneous precipitation distribution: (A) National land capability classification; (B) Modified national land use morphology; ( C) Distribution of degraded land (shaded area), assuming that all land under an unsuitable use eventually becomes degraded; (D) Distribution of degraded land ( shaded area), assuming that there is also subsequent displacement of mean annual isohyets due to global climate change – dashed lines show new positions of isohyets.



In dry countries, rainfall is highly variable in space and time. Rainfall bands routinely move north and south in a cyclical fashion, and biomass productivity follows (Tucker and Choudhury, 1987). So in a real dryland environment where physical and economic margins oscillate in a north-south direction, the actual pattern of degradation will be determined by complex interactions between land use change, its underlying causes, and the relationship between land use type, land use intensity and land capability, and will tend to look more like Figure 3d than Figure 3a.

*Dynamic Assessment of Restoration Suitability and Risk*

This model could also be used to assess in a dynamic way the physical suitability of degraded land for restoration and carbon sequestration. The only modification to the above structure would be to replace the vegetation and soil data layers by ones showing the existing distribution of vegetation and soil degradation, estimated as described earlier in the paper. By simulating trends in desertification over time, the model could simulate trends in suitability too. It could also be used to predict how vulnerable revegetation and desertification control projects might be to subsequent desertification and climate change, and hence their risk of failure.

## Conclusions

Restoring degraded drylands and controlling desertification could play a major role in global climate change mitigation. Obtaining improved assessments of the area of degraded land that is physically suitable for restoration will require better global data on vegetation and soil degradation than are currently available. Individual research groups have been filling the gaps in data availability by using satellite imagery and GIS analysis, but this cannot substitute for a comprehensive global monitoring programme. A practical set of desertification indicators needs to be chosen soon so that monitoring can get under way and the second UNEP World Atlas of Desertification can be based on a less subjective assessment than the first one.

However, just because an area of land appears to be physically degraded, and physically suitable for restoration, it does not mean that it is actually available. A proper appraisal of the actual availability of land for restoration requires integrating the biophysical and socio-economic attributes of an area, and seeing how these change in space and time.

The contribution that drylands could make to mitigating global climate change will also depend on how physical suitability changes as a result of continuing desertification and global climate change. This is difficult to estimate, as is the overall mitigating effect of controlling desertification, which would increase carbon sequestration and reduce carbon emissions. The dynamic model outlined above could be used to evaluate possible long-term strategies, by simulating changes in national land use and land degradation over time, how these are influenced by global and regional climate change, and how land suitability could be influenced by desertification and global climate change.

Since models of this kind could be incorporated in global climate models, allowing them to treat the terrestrial component of global environmental change and terrestrial-atmospheric interactions in far more detail than at present, they should also help to raise the profile of desertification in global environmental change research, which will give a greater incentive to obtain better data on its extent and rate of change.

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# Forging an economic linkage between fossil fuel burning, climate change and dryland restoration.

**Victor R. Squires and Edward P. Glenn**

**Synopsis**

The case for linking the burners of fossil fuel with global efforts to arrest the rate of land degradation is argued. Evidence is presented that fossil fuel burners would benefit from implementing a program of carbon offsets on drylands. About 0.5 - 1 0 giga ton of carbon can be sequestered annually at a cost per ton of carbon sequestered in the order of $10-18 -This compares favorably with costs in alternative carbon offset programs.

Key Points

1. The world's drylands (excluding the hyperarid regions) cover 5.2 billion ha and are represented on all continents. Globally non-forested drylands represent 43% of the world's land surface. Most are under some form of human management.
2. There is a case for re-evaluating the role of drylands in the global efforts to prevent the increase in greenhouse gases. Because many drylands are degraded their productivity has declined. Technical solutions exist for restoration of most degraded lands. International efforts to restore the drylands (stop desertification) have been hampered by lack of funds.
3. Drylands have the potential to sequester 0.5 - 1.0. giga tons of carbon annually if restored to their ecological potential. The money required to do this might come in part from the burners of fossil fuels through a program of carbon offsets.
4. The major utility companies and oil and gas producers need cost effective and ecologically sound carbon offsets to assist with their targets to reduce or remediate carbon dioxide emissions.
5. Biotic carbon offsets are ways and means for linking fossil fuel burners with global efforts to arrest global climate change through a program of revegetation and dryland rehabilitation. The challenge is to devise ways to raise biomass levels to trap and store carbon in the long term.

**Key words: rangeland rehabilitation, desertification, biotic carbon offsets, carbon sequestration**

## Introduction

Rangelands occupy a large proportion (about 43%) of the Earth's land surface (see chapters 1, 2 and 3). They have a potential to be a sink for significant amounts of carbon, especially if they are restored to their ecological potential (Glenn at al 1993, Ojima et al 1993). However, many rangelands are degraded (Dixon, James and Sherman 1989; Dregne, Kassas and Rosanov 1991) and their productivity is low (Table 1).

There is a case, though, for a re-evaluation of the definition of degraded land in the light of the new objective of maximising carbon sequestration. The definition adopted by United Nations Environment Program (UNEP 1991) depended, to a large extent, on recognising changes in the productivity with special emphasis on changed value for grazing livestock.

**Table 1: Extent of desertification in rangelands within the drylands of the world: millions of hectares**

| | Slight-None | Moderate | Severe | Very Severe | Total |
|---|---|---|---|---|---|
| Africa | 347 | 274 | 716 | 5.3 | 995 |
| Asia | 384 | 485 | 692 | 10.8 | 1188 |
| Australia | 296 | 277 | 55 | 29.0 | 361 |
| Europe | 31 | 27 | 52 | 1.2 | 81 |
| N. America | 72 | 116 | 285 | 10.2 | 411 |
| S. America | 93 | 88 | 184 | 15.3 | 288 |
| Total | 223 | 1267 | 1984 | 71.8 | 3323 |

Source: Dregne, Kassas and Rosanov (1991)

Large tracts of the world's rangelands have undergone marked changes to their botanical composition (a major criterion for defining degradation) and yet many are supporting trees and shrubs (often so-called "woody weeds") which have the capacity to sequester more carbon dioxide than the original vegetation they replace (Scholes; Grainger, this volume). Woody weeds, although of limited value as livestock feed, often are present in higher densities, have a higher standing crop and a higher below ground biomass. Net primary productivity is often higher than the pristine grasslands (Chew and Chew 1965; Ludwig 1987). A re-evaluation of woody weeds is required and their role as carbon sinks should be properly assessed (Scholes, this volume) This assumes greater importance in the light of the recent signing by many countries of the International Convention to limit the emission of so-called greenhouse gases to the atmosphere (UNEP 1994).

Emission of greenhouse gas

The emission of the greenhouse gas $CO_2$ to the atmosphere continues to escalate (Table 2). Coal, gas and petroleum-fired electric power stations account for one-third of the global fossil fuel C emissions (Haggin 1992; Judkins, Fulkerson and Sanghvi 1993). Recent estimates of annual emissions of $CO_2$ by fossil fuel and land use change for the period 1980 to 1989 are 5.4 ± 0.5 Pg of C per year (1 P= $10^{15}$ g=1 Gt) and 1.6 ± 1.0 Pg of C per year, respectively (Post et al 1990). Global oceans are estimated to absorb 2.0 ± 0.8 Pg of C per year, and about 3.2 Pg per year remains in the atmosphere (Houghton 1990, 1992). This calculation leaves an amount of 1.8 ± 1.4 Pg per year unaccounted for. The so called "missing" $CO_2$ (Gifford 1994).

A role for drylands in carbon sequestration?

We have argued ( Glenn et al 1993, Ojima et al 1993) that the world's drylands have a role in sequestering a substantial part of this "missing" C. Globally, non-forested drylands cover 43% of the world's land surface yet they are not currently regarded as important in sequestering C due to overuse and poor management. Excluding the hyper arid regions, which are largely uninhabited deserts, the drylands total 5.2 x $10^9$ ha (UNESCO 1977). This is a very large landbase exceeding the area of cropland (1.4 x $10^9$ ha) or closed forests ($4.4 \times 10^9$ ha) Most of the world's drylands are already under management. Logically, if these terrestrial ecosystems are already absorbing some of the excess C, those parts under direct human management could be managed to store even more carbon (Glenn et al 1993).

**Table 2: World primary energy use and associated $CO_2$ emissions for selected years 1967-1988**

| | | Fossil | | |
|---|---|---|---|---|
| | q | % | total, q | $CO_2$Gt(C)/y |
| **1967** | | | | |
| US | 55 | 96 | 58 | 1.0 |
| World | 165 | 94 | 176 | 3.3 |
| **1973** | | | | |
| US | 70 | 95 | 74 | 1.3 |
| World | 220 | 94 | 235 | 4.5 |
| **1977** | | | | |
| US | 71 | 93 | 76 | 1.3 |
| World | 237 | 92 | 258 | 4.8 |
| **1985** | | | | |
| US | 66 | 90 | 74 | 1.3 |
| World | 260 | 88 | 295 | 5.3 |
| **1986** | | | | |
| US | 66 | 89 | 74 | 1.3 |
| World | 266 | 88 | 302 | 5.4 |
| **1987** | | | | |
| US | 68 | 89 | 76 | 1.3 |
| World | 273 | 88 | 310 | 5.5 |
| **1988** | | | | |
| US | 71 | 89 | 80 | 1.4 |
| World | 282 | 88 | 320 | 5.5 |

Where q = quantity and CGt/yr = gigatons of carbon per year
Source: Judkins, Fulkerson and Sanghvi (1993)

## Drylands and desertification

Desertification is a severe problem throughout the drylands (Dregne, Kassas and Rosanov 1991, UNEP 1994). UNEP does not quantify percentage reduction in net primary productivity (NPP) in each degradation class but from the verbal descriptions we infer that moderate degradation is approximately 25 - 50%, severe 50-75% and very severe >75% loss of NPP due to land use practices. As desertification proceeds, costs rise dramatically (Table 3) as land passes into higher degradation categories which are more costly to restore (Kassas, Ahmad & Rosanov 1991: Dixon, James and Sherman 1989). At present, 70% of rangelands ($3.6 \times 10^9$ ha) are at least moderately degraded.

**Table 3. Global average indicative costs ($U.S.) for direct anti-desertification measures with different categories of degradation**

| | Cost of Restoration ($U.S. hectare) | |
|---|---|---|
| Categories of Degradation | Rainfed Croplands | Rangelands |
| Slight to None | 50-150 | 5-15 |
| Moderate | 100-300 | 10-30 |
| Severe | 500-1500 | 40-60 |
| Very Severe | 2000-4000 | 30-70 |

Source: UNEP (1991)

**Table 4. Global cost in ($U.S. billion) for a 20 year program of direct anti-desertification measures.** Source: UNEP (1991)

| | Rainfed Croplands | Rangelands | Cost/Year[2] |
|---|---|---|---|
| Preventive Measures | 12-36[1] | 6-18 | 0.3-0.9 |
| Corrective Measures | 18-55 | 13-38 | 0.7-1.9 |
| Rehabilitation Measures | 22-59 | 80-120 | 4.0-6.0 |
| Total | 52-40 | 99-176 | 5.0-8.8 |

1. Best and worst case scenario
2. $U.S. billion

Desertification is estimated to be proceeding at a rate of 3.5% per year (UNEP 1994). The value of income foregone due to degraded rangelands is $23 billion per year (UNEP 1991). However, the cost:benefit ratio of restoring rangelands is 1:3.5 on a global basis. The cost depends on the severity of the degradation and the intensity of the land use (Dregne, Kassas and Rosanov 1991: Dixon, James and Sherman 1989). It was estimated (UNEP 1991) that $171-363 billion would be needed over the next 20 years to restore rangelands. Annual costs to restore degraded drylands are in the order of $5.0 - 8.8 billion (Table 4).

Where will the money come from?

The large sums of money required to restore rangelands are probably beyond the means of UNEP (the principal UN agency charged with this responsibility). However, funds could be provided via privately financed carbon offsets programs and special efforts to sequester carbon dioxide in the world's drylands (Glenn et al 1992).

## Carbon offsets: what are they and how do they help?

Carbon offsets are an attempt to reabsorb atmospheric carbon to limit or delay the potential deleterious effects e.g., global warming and climate change. In environmental economics terminology, an offset is an action taken "beyond the stack" in order to compensate for emissions considered excessive by regulations, thus avoiding a tax liability or other penalty for excessive emissions. The major utility companies and oil and gas producers are looking for cost effective and ecologically sound carbon offsets because each year they are emitting 6 billion t of C per year. They will have (collectively) between $5 - 10 billion per year for offsets (Trexler and Meganck 1993). There is however, no guarantee that it can be diverted to drylands (Kinsman and Kaster, this volume and Trexler and Broekoff, this volume).

Carbon dioxide can be removed from stack gas by physical-chemical means at the source. However, this option is expensive (approximately $300 per t of C) and creates additional disposal problems (Flour 1991). The option of absorbing excess C into biomass through global-scale revegetation programs appears attractive (Kinsman and Trexler 1993) despite problems of cost, availability and the difficulty of achieving and monitoring long-term C storage (Trexler 1993). Possible carbon offsets programs to remove carbon dioxide via living vegetation or to store it in the soil have been proposed for every type of ecosystem.

Projected costs of such carbon offsets range from $5 to $200 per t (Trexler 1993) - considerably lower than the cost of C removal from the stack. Wisely chosen projects could have beneficial social and environmental effects in addition to C removal, especially in many of the world's rangelands where population pressure is high and quality of life is low (Trexler and Meganck 1993). Action to improve the biomass on arid rangelands will protect biodiversity, ameliorate the living conditions of the local people as well as retain carbon (Grainger 1992; Ojima, Galvin and Turner 1994).

## How feasible is it?

If terrestrial C storage is attempted, in the order of 2-5 x $10^8$ ha of land will be required to absorb just 25% of C emissions into vegetation (Marland 1988; Vitousek 1990; Trexler 1993). Finding enough land will be challenging, for there are many competing demands (Grainger, this volume). Most human activities reduce rather than enhance C storage on the landscape. It would obviously be desirable to develop a new land base for crop production and C storage. While this may at first seem impractical, it may occur naturally to some extent if global climate change models are correct (Neilson, 1993; Rozenweig and Parry 1994).

Some of these models project changes in the boundaries between the arable regions and rangelands and greater net precipitation in the arid zones with global warming ( Pearman 1988; Pittock 1993; Houghton et al 1990, 1992). It may also be possible to utilize new technologies to develop underused desert land for crop production without waiting for climate change to take place (Glenn et al 1992; Squires and Ayoub 1994).

## Dryland soils as a carbon sink

An intriguing possibility is to use the world's drylands as a C sink (Glenn et al 1993). In the past there has been some degradation with a concomitant loss of C to the atmosphere through oxidisation of soil organic matter. A move away from practices that lead to degradation could reverse this process and begin to sequester C via increasing levels of organic matter in the upper layers of the soil (Gifford 1993; Ojima et al 1993; Parton et al 1993).

Preliminary modelling estimates are based on the CENTURY model (Parton et al 1992). Soil C fluxes are driven, in part, by net primary productivity (NPP) on the land surface. Desertification results in reductions in NPP. Hence it is theoretically possible to calculate (using the CENTURY model and the UNEP data) the differences in soil C flux between scenarios of "acceptable" management regimes versus current levels of "unacceptable" management projected into the future. Simulations of soil C sequestration under three different land management scenarios showed changes in C level of the soils in dryland regions (Ojima et al 1993a). Differences in C flux between "sustainable" (30% biomass removal) and "regressive" management scenarios (50% or 80% biomass removal) were derived (Ojima et al 1993a). Projections based on a double $CO_2$ climate including climate-driven shifts in biome area by the year 2040, resulted in net sinks of 5.6 Gt, 26.8 Gt and 27.4 Gt for three different land cover projections respectively when compared with "optimal" rangeland management.

The increase in soil C storage in this future projection resulted mainly from climate-induced changes in biome boundaries together with a net increase in soil organic matter density resulting from net ecosystem response to climate changes and enhanced $CO_2$ concentrations. CENTURY predicted that over a 50 year period the difference in carbon emissions between the regressive scenario and the sustainable management scenario will be 37 Gt (annual difference = 0.7 Gt) over the whole land base under consideration ($4.5 \times 10^9$ ha of rangelands). calculations based on those of Gifford et al (1990) suggest that the present C pool in rangeland soils (estimated at 417 Gt to a depth of 1m) may be a net sink of 0.6 Gt per year (see also Table 1 on p.336 which is based on the expert consultation at UNEP in 1995).

The cost per t of carbon stored is about $U.S. 10. This is a reasonable cost compared to reafforestation, tree plantations and other C offsets (Kinsman and Trexler 1993). The soil store of C in these rangeland ecosystems is a very important pool since it is stabilized for hundreds to thousands of years, and forms the bulk of the rangeland C pool. At present on severely degraded lands, release of $CO_2$ due to overutilization of plant production, ensure that rangelands are a source of $CO_2$. The benefits of turning this around so that rangelands become a sink are obvious. Dryland soils are generally low in C (Batjes, this volume ) but there is scope to augment this via better management (Tinker and Ineson 1990).

## Carbon offsets - the practicalities

If anti-desertification measures to restore rangelands were funded at the levels recommended by UNEP (1991) and resulted in say 0.5 Gt per year of C sequestration (see predictions in this volume) the costs would be $10-18 per t of C. This is attractive to the utility company who must compare the cost:benefit ratio which obtains for forestry-based offsets programs. Where anti-desertification measures are applied and rangelands are restored, the utility company can claim the difference in total C held between the original degraded lands and the sum average total of restored lands.

Unless adequately regulated the C offsets concept can be misused. An important principle of the offsets idea is that emitted C is "recaptured". There is a risk that an area of rangeland that benefited from offset funding could be damaged by such unpredictable factors as wildfire or climate change or even converted to another use under changed tenure arrangements, thereby resulting in no net reduction. The implications for grazing control in areas of communal grazing also need to be addressed but the apparent difficulties might be simplified by the adoption of national plans of action to combat desertification.

Other potential problems include the following:

- it may be difficult to distinguish a genuine offsets program from opportunistic schemes that operate in contradiction to the spirit of the offsets concept. For a discussion of "carbon fraud" see Trexler (1991).
- as with all sustainable development issues, the question of equity must be considered - a technically feasible C offset project may infringe on the rights of some parties. (see Trexler and Meganck 1994, Trexler and Broekoff (this volume ).
- although pioneering work is underway to clarify the carbon savings of various rangeland restoration schemes, there remains considerable difficulties in estimating the C offsets from anti-desertification measures.
- the length of time that C needs to be stored before being officially "offset" is also a matter for debate.

The whole question of carbon offsets is an evolving one. The likelihood of "tradable" (both locally and internationally) permits being established and the formation of international protocols has been canvassed (Jones and Stuart 1994; Trexler and Meganck 1993; Trexler and Broekoff, this volume ).

The degree to which C offsets will be taken up as an option in the world's rangelands is highly dependent on the ongoing negotiations of the Framework Conventions on Climate Change (FCCC) which was signed at the Rio Earth Summit in 1992 and the Convention to Combat Desertification agreed in Paris in October 1994 (see also Trexler and Broekoff, this volume). A major concern, and a challenge to rangeland scientists, is whether anti-desertification measures, even if fully funded, can be effective. Technical solutions to rehabilitating rangelands are available but if population increases put additional pressure on the landscape, desertification may continue despite external funding for anti-desertification programs (see Stafford Smith et al this volume).

Since the effects of $CO_2$ enrichment on climate are uncertain, those actions that make sense even in the absence of global warming have been recommended as present action steps ("no regrets" policy). Restoration of productivity of the drylands to their ecological potential (Le Houerou 1984) through anti-desertification measures, plantation of halophyte crops on salt-affected soils (Squires and Ayoub 1944) and other management practices (Skoupy 1993) falls into this category of action step.

Creating an economic linkage between fossil fuel burning and restoration of drylands to store C would appear to be a worthwhile goal in view of the importance of drylands in global food production and human welfare.

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# Arid Lands as Carbon Offsets - A Viable Option?

**Mark C. Trexler and Derik J. Broekhoff**

## Synopsis

A series of evaluative criteria (credibility, reliabilityquantifiability and cost-effectiveness), previously applied to energy and forestry-based carbon offset projects are applied to the category of arid land carbon sequestration projects. The general performance of arid land-based carbon sequestration projects is then compared to energy and forestry-based mitigation opportunities in order to evaluate their likely attractiveness to the private sector as carbon offset options. Recommendations are made regarding project types and structures most likely to result infunder interest.

## Key points

1. Arid lands are not an obvious candidate to be pressed into service for climate change mitigation purposes. Such lands tend to have low levels of productivity and relatively modest biomass accumulation potentials. The low biotic potential of drylands, however, could be compensated by their sheer areal extent: 6.1 billion ha worldwide, 5.2 billion of which are arid, semi-arid or dry sub-humid and capable of supporting some form of agricultural production. They cover more than one third of the world's total land area.

2. The link between arid lands and climate change mitigation is clear. Stopping desertification would slow the release of carbon dioxide into the atmosphere; rehabilitating desertified lands can help sequester carbon in plants and soils. Traditional anti-desertification projects have the theoretical potential to serve as climate change mitigation measures

3. On a global scale, there may be significant physical potential for carbon sequestration on drylands, Although very rough, available cost data does not prima facie suggest that cost need be a disqualifying factor. The costs of sequestering carbon on drylands also depend on numerous factors. Costs of specific anti-desertification measures, calculated in terms of climate change mitigation benefit, will depend on the same variables that affect a hectares' physical capacity to sequester carbon, as well as additional variables including labor values, infrastructural requirements, as well as monitoring and verification difficulty.

4. There are many ways to offset carbon dioxide emissions, and therefore many options available to parties interested in the offset projects. Until more experience with offset projects has accumulated and methods of carbon accounting and program implementation are uniformly recognized by funders, regulators,and host countries, careful consideration must be gven to the design and implementation of carbon offset and joint implementation projects.

5. For the purposes of analyzing the attractiveness of these projects to private carbon offsets funders, three basic project types can be identified: improved crop, pasture and rangeland management; planting salt-tolerant crops ( halophytes); and afforestation. These projects can reduce ongoing carbon loss from lands, increase per hectare carbon uptake and storage, and in some cases can result in energy production and fossil fuel displacement.

6.Whether projects on drylands become a target for private sector funding depends on how potential project funders evaluate them. Private investors will want to minimize risk and maximize return. Minimizing risk will often mean looking for projects in areas familiar to investors where political and economic conditions are stable and conducive to private sector goals. Maximizing returns means getting more carbon per dollar invested. Investors will seek out

ways to share costs and to locate organizations capable of assisting in project implementation. In addition, private investors are likely to use several related technical criteria in evaluating projects. These include project credibility,reliability, quantifiability, and cost effectiveness. Projects may also be assessed for their ancillary benefits.
7. Any effort to preserve or sequester a large number of tons of carbon on arid lands would involve significant scientific and organizational challenges. Moving from the theoretical ability of arid lands to contribute to climate change mitigation to the practicality of developing and implementing climate change mitigation projects is a large step.

**Keywords: global warming, carbon offsets, joint implementation, desertification, climate change mitigation, private sector, Global Environment Facility (GEF)**

## Introduction

Arid lands are not an obvious candidate to be pressed into service for climate change mitigation purposes. Such lands tend to have low levels of productivity and relatively modest biomass accumulation potentials. Any effort to preserve or sequester a large number of tons of carbon on arid lands would involve significant scientific and organizational challenges. The potential role of arid lands in climate change mitigation, however, continues to be advocated.

First, to the extent that climate change mitigation is perceived as an issue area with considerable funding potential from both public and private sources, countries and interest groups with an arid lands focus are reluctant to be left out. Global warming, for example, is one of three priority areas under the Global Environment Facility's (GEF) mandate; arid lands and desertification programs are not mentioned. As a result, there have been ongoing attempts to have anti-desertification projects classified as global warming projects for GEF purposes.

Secondly, the sheer amount of arid land around the world that is already degraded or which faces significant degradation in the foreseeable future is huge. Because there is so much arid land, even modest carbon stock changes on a per hectare basis could have potentially significant global implications.

Moving from the theoretical ability of arid lands to contribute to climate change mitigation to the practicality of developing and implementing climate change mitigation projects is a large step. This is particularly true with respect to the potential for private sector funding for climate change mitigation purposes ("carbon offsets"). The potential for carbon offset funding is integrally linked not only to technical issues such as project credibility, reliability, quantifiability, and cost-effectiveness, but also to political and policy developments in the climate change arena, including the rapidly developing area of joint implementation. A review of potential arid land-based carbon offset project types suggests that procuring private sector funding for these projects will be difficult, particularly in the United States. Success will require a clear focus on designing projects that stand out on the basis of the evaluative criteria used here.

## The Link Between Arid Lands and Climate Change Mitigation

### *A. The Conceptual Basis*

At first glance, the possibility of large-scale carbon sequestration on drylands might appear unlikely. Compared to other biomes, these lands are less productive and have less biomass accumulation potential. Semi-arid and arid lands, for example, accumulate just a fraction of the hundreds of tons of carbon per hectare that can accumulate in temperate and tropical forest systems, and at a fraction of the annual rate.

The low biotic potential of drylands, however, could be compensated for by their sheer areal extent: 6.1 billion ha worldwide, 5.2 billion of which are arid, semi-arid, or dry sub-humid and capable of supporting some form of agricultural production (Dregne, *et al.*, 1991). They thus cover more than one third of the world's total land area, a greater portion than is covered by either cropland (1.4 billion ha) or closed forest (4.4 billion ha) (Glenn, *et al.*,1993).

Human activity degrades arid lands at an alarming rate. Population has risen rapidly in these regions in the last 30 years, and the increasing and intensive use of marginal land to sustain this population has had a substantial impact: around 70 percent of arid lands used for agricultural production have been desertified (Dregne, *et al.*, 1991). Current estimates place the rate of desertification of economically productive arid lands worldwide at around 3.5 percent, though the rate ranges from 0.1 percent to more than 10 percent in the driest and most severely affected regions (Dregne, *et al.*, 1991). The loss of net primary productivity from these lands has obvious economic and climatic implications. The economic loss has been estimated at more than $40 billion per year (Dregne, *et al.*, 1991). Worldwide, arid lands are certainly a net source of atmospheric carbon, although current estimates are rough. While per-hectare emissions of carbon on these lands may be small, the sheer number of hectares being degraded means that over time they could have a cumulatively significant impact on the global carbon cycle.

The link between arid lands and climate change mitigation is therefore clear. Stopping desertification would slow the release of carbon dioxide into the atmosphere; rehabilitating desertified lands can help sequester carbon in plants and soils. Traditional anti-desertification projects—including improvement of crop- and rangeland management, revegetation and afforestation of desertified lands, dune stabilization, and cultivation of drought-resistant crop species—all have theoretical potential to serve as climate change mitigation measures. The prominence of climate change as an international political issue and the substantial amounts of public and private funding which may become available to address it provide sufficient reason to examine the potential of arid lands to mitigate climate change (Trexler and Meganck, 1993).

### *B. A Review of the Potential of Arid Lands to Sequester/Preserve Carbon*

The per hectare capacity of drylands to sequester carbon depends on numerous factors, including the intervention used and the land's relative productivity, (*i.e.*, whether the land is arid, semi-arid, or dry sub-humid). Several analysts have attempted to quantify globally the amount of carbon that could be sequestered as a result of extensive anti-desertification measures. One preliminary estimate suggests that a global program to restore arid rangelands could sequester 0.5 Gt of carbon per year over 100 years (Glenn, *et al.*, 1993). A related study by Ojima, *et al.*, uses the CENTURY model to estimate how much carbon could be sequestered over 50 years through sustainable management of 4.5 billion ha of grasslands and drylands. Taking current rates of erosion and desertification as a baseline, Ojima *et al.* conclude that 37 Pg of carbon would be preserved or sequestered in grassland and dryland ecosystems through sustainable management. This amounts to 0.7 Pg of carbon per year.

The costs of sequestering carbon on drylands also depend on numerous factors. Costs of specific anti-desertification measures calculated in terms of climate change mitigation benefit will depend on the same variables that affect a hectare's physical capacity to sequester carbon, as well as additional variables including labor values, infrastructural requirements, as well as monitoring and verification difficulty. UNEP has estimated that a 20-year program to globally combat desertification on all economically productive drylands would cost between US $10.0 and US $22.4 billion per year (Dregne, *et al.*, 1991). Although very rough, UNEP's estimates have provided most of the current cost-per-ton calculations for sequestering carbon on arid lands.

Based on UNEP's component cost estimates for combatting desertification on rangelands (US $99-176 billion over 20 years), Glenn *et al.* estimate the cost per ton of carbon benefit at US $10-18 (Glenn, *et al.*,1993). They also estimate that cultivation of halophytes (for forage, feed, and oil seed) and improved arid pasture- and cropland management could sequester another 0.5 Gt per year, at a similar cost. Ojima *et al.*'s modeling arrives at a cost of US $10 per ton of carbon. This result should not be too surprising since they use the same UNEP component cost estimates (Ojima, *et al.*, 1993).

These studies suggest that, on a global scale, there may be significant physical potential for carbon sequestration on drylands. Although very rough, available cost data does not *prima facie* suggest that cost need be a disqualifying factor. Determining the practical potential for sequestration efforts, however, requires an assessment of the social and economic feasibility of implementing anti-desertification measures on a large scale (Smith, *et al.*, current proceedings). It will also require much more detailed cost assessments.

## C. *Potential Arid Land-Based Carbon Offset Types*

Desertification has many causes. As Kassas, *et al.*, state: "Desertification reflects not only the inherent fragility of particular land resource systems but is also indicative of the pressures generated by growing populations, the increasing need for food and agricultural produce, economic growth, demands of trade and external debt, macro economic policies to support State objectives and the myriad of other by-products generated by the development process" (Kassas, *et al.*, 1991). Projects to combat desertification are thus necessarily varied and often complex. For the purposes of analyzing the attractiveness of these projects to private carbon offset funders, three basic project types can be identified: improved crop, pasture, and rangeland management; planting of salt-tolerant crops; and afforestation.

These projects can reduce ongoing carbon loss from lands, increase per hectare carbon uptake and storage, and in some cases can result in energy production and fossil fuel displacement.

1. Improved crop, pasture, and rangeland management

The primary proximate cause of desertification is overuse or misuse of drylands for agriculture. Because of this, most anti-desertification programs aim to improve the management of these lands. Activities in this category can include reducing the incidence of fallow, increasing crop residue retention, nitrogen fertilizer applications, and shifting to mixed-cereal legume-based pastures (as opposed to monocultures) (Grace, current proceedings). They may also include soil and water conservation activities, rangeland management and rehabilitation, and revegetation to stabilize sand dunes.

All these activities may sequester carbon, and in theory the provision economic incentives or subsidies could promote their implementation. In practice, successful implementation of measures to improve agricultural land management usually requires a high degree of local community consultation and involvement.

An analysis of two anti-desertification and anti-drought programs in the United States and Syria suggests common elements to success: application of new or appropriate technologies; availability of additional funds; and societal and managerial restructuring (Kassas, *et al.*, 1991; Kassas, 1988). The effectiveness of both programs relied on the "political will as shown by both governments and positive public participation by the communities concerned" (Kassas, *et al.*, 1991). Land management projects are likely to suffer from unsatisfactory implementation and poor longevity without community participation in their planning and development.

A good model of a typical arid-land carbon offset project can be found in two recently approved projects sponsored by the Global Environment Facility (GEF) and the United Nations Sudano-Sahelian Office (UNSO) (Global Environment Facility, 1992a,b). These five-year projects—in the Sudan and Benin -- are intended to demonstrate the viability of fostering improved land management in arid or semi-arid regions in order to sequester carbon. Both projects involve communities in improving rangeland management, as well as developing various tree-planting programs.

In the Sudan, local participants have been instructed in revegetation and stabilization of sand dunes, and in the planting of windbreaks around croplands. The Benin project involves primarily afforestation, but also involves protecting woody savanna through development of improved rangeland management and intensification of agricultural production on croplands. For both projects, carbon accumulation is assumed to result from increased accumulation of biomass on affected lands, as well as from decreased deforestation in surrounding areas (in Benin). In addition, it is predicted that project efforts will be sustained beyond the five-year implementation period (out to 20-30 years) and will be replicated by surrounding communities, resulting in additional indirect carbon benefits.

Assessment of the success and effectiveness of these programs continues. Both projects involve large land areas, and rely upon extensive coordination of people and communities in countries with relatively unreliable political and economic conditions. A significant portion of the carbon benefits of these projects derives from the extent to which program activities will be perpetuated by surrounding communities. These factors will likely influence a final assessment of the projects as carbon offsets.

2. Cultivation of Halophytes

There is significant potential for increasing carbon storage on drylands through the cultivation of salt-tolerant halophytes (Glenn, *et al.*, 1993). Of the 5.2 billion ha of potentially productive world drylands, approximately 0.7 billion ha are affected by salt. According to one estimate, 55 desert regions worldwide containing approximately 130 million ha could support halophyte production (Glenn, *et al.*, 1991). Planting halophytes could help reverse desertification by restoring productivity to these lands.

Most halophyte planting programs involve their use as crops. While halophytes could be planted simply as a means to restore productivity to salinized and desertified soils (halophytes remain viable even when irrigated with seawater), they have traditionally been used for grazing, and offer promising potential for forage, feed, and oilseed (Glenn, *et al.*, 1993). They may also have potential as biomass energy crops. Sustainable cultivation schemes for halophytes involve increasing soil organic matter by plowing under unused residues. Glenn *et al.* estimate that intensive halophyte production could sequester 0.6-1.2 Gt of carbon per year, if carried out on all viable lands.

Though they are probably economically and environmentally beneficial in their own right, halophytes have been studied specifically for their potential to absorb and sequester atmospheric carbon. Field trials funded by the Electric Power Research Institute and the Salt River Project have determined that the most promising species of halophytes can absorb as much as 4-8 tons of carbon/ha/year under ideal circumstances including irrigation (Glenn, *et al.*, 1993). Effective sequestration depends on how much carbon goes into long-term storage in soils or on how much carbon from fossil fuels is displaced from using halophytes as energy crops. Glenn, *et al.*, estimate that plowing under unused halophyte crop residues alone could sequester carbon for about US $12 per ton, based on typical costs for Arizona. The feasibility of large-scale halophyte production and actual costs from a carbon offset perspective remains to be tested.

3. Reforestation and Afforestation

In the context of combating desertification, most reforestation schemes are part of larger land management programs. Trees can serve as hedges and windbreaks and can provide a source of alternative income on otherwise overgrazed or overused range and croplands. In terms of carbon sequestration, drylands simply lack the productive potential to support tree plantations comparable to those in "traditional" forestry carbon offset projects. Still, there is some potential for restorative tree-planting projects on arid lands that would also function as carbon offsets. Some pilot projects of this type have already been initiated.

One of these is the UNSO project in Benin mentioned above. Through this project—involving protection of woody savanna, establishment of nurseries, and planting of living fences—between 0.7 and 1 million tons of carbon may be sequestered on 20-30,000 ha (Global Environment Facility, 1992a). Total project cost is US $2.5 million.

Another project currently under development is Appropriate Technology International's (ATI) demonstration program in Senegal for the planting and commercial utilization of jatropha trees (ATI, 1995). Jatropha are fast-growing and drought-resistant, and are used primarily for windbreaks. The seed oil of jatropha can be used as a substitute for diesel fuel. This quality makes the trees exceptional for offsetting carbon, since they not only absorb carbon dioxide, but their seed oil can displace fossil fuel emissions. ATI estimates that its current project will offset approximately 135,000 metric tons of carbon over 5 years, at a cost of US $2 million, or $15 per ton of carbon. The project will also provide significant employment and income for tree planters and harvesters. Extensive opportunities exist for jatropha planting projects throughout the developing world (ATI, 1995).

A more questionable carbon offset option is represented by the so-called "Eden" project. This project is based on the use of a small solar-powered condensation and irrigation device to provide moisture to the root systems of individual trees (Global Warming Research and Development Corporation, 1994). The "Eden Device" can allegedly pull moisture out of the atmosphere to allow trees or other plants to grow in soils and climates where they otherwise would not survive. Though there would likely be numerous logistical (not to mention political and ecological) obstacles to this undertaking, backers of the device allege that atmospheric carbon dioxide levels could be stabilized through the planting of 2.3 billion trees at a cost of US $100 billion over the next 30 years (*Id.*). This estimate remains unsubstantiated.

Whatever the attractiveness of reforestation projects on drylands, however, opportunities for their implementation are likely to be less numerous than those of other anti-desertification measures.

## Relevant Carbon Offset Evaluation Criteria

There are many ways to offset carbon dioxide emissions, and therefore many options available to parties interested in offset projects. These options fall into three broad categories: $CO_2$ emissions reduction-based offsets; $CO_2$ sequestration-based offsets; and non-$CO_2$-based offsets. Emissions reduction-based offsets include energy sector activities such as improving supply-side efficiencies, demand side management, fuel-switching, and biomass energy projects. In addition to reducing $CO_2$ emissions, such projects can offer economic returns to investors. Emissions-reduction projects may also include efforts to prevent degradation of biological resources, such as forests. Many elements of dryland offset projects, notably rangeland protection and dune stabilization, would fall into this category.

Sequestration-based offsets include any type of project that takes advantage of natural photosynthetic processes to increase the amount of $CO_2$ being removed from the atmosphere. These projects are diverse, and include measures such as reforestation, afforestation, and soil management. Most offset projects envisioned on drylands (increasing crop residue retention, halophyte cultivation, tree planting) are sequestration projects. Energy crop projects span the two categories.

Finally, non-$CO_2$-based offsets involve activities that seek to reduce emissions of other greenhouse gases, such as methane, nitrous oxide ($N_2O$), or chlorofluorocarbons (CFCs). Examples of such activities include capturing methane from landfills, reducing the use of $N_2O$-emitting agricultural fertilizers, and reducing CFC emissions from air conditioners.

Because policy and implementation guidelines for carbon offsets are still in their infancy, there is no consistent definition of what constitutes a "good" offset. Whether projects on drylands become a target for private sector funding depends on how investors are likely to evaluate them. Private investors will want to minimize risk and maximize return. The return can include carbon or a combination of carbon and financial returns. Private sector evaluation criteria, however, may differ from those of conventional anti-desertification funding sources because private investors in carbon offsets are more likely to be held accountable for project long-term success (Trexler, 1995).

Minimizing risk will often mean looking for projects in areas familiar to investors where political and economic conditions are stable and conducive to private sector goals. Maximizing return means getting the most carbon per dollar invested. Investors will often seek out ways to share costs and to locate organizations capable of assisting in project implementation.

In addition, private investors are likely to use several related technical criteria in evaluating projects. These include project credibility, reliability, quantifiability, and cost-effectiveness (Trexler and Associates, 1995). Projects may also be evaluated for their ancillary benefits. These criteria are further developed below.

*A. Offset Credibility*

The criterion of credibility is the degree to which the basic concept on which a project's $CO_2$ benefits are based is both plausible and robust. This can be measured in several different ways. A primary measure would be how direct the link is between project activity and $CO_2$ benefits: the more direct the link, the more credible the project. Improving the efficiency with which a power plant produces electricity, for example, has a direct, measurable effect on its $CO_2$ emissions, and thus would be a highly credible offset.

The link between improved land management and carbon sequestration—where training in crop management will lead to soil preservation and eventual buildup of organic matter and carbon—may not be so direct. If achievement of $CO_2$ benefits depends on a complex chain of events, perhaps based on market or technology predictions, the project's credibility is weakened.

Another measure of credibility involves the potential for "leakage." Leakage refers to the loss of $CO_2$ benefits over time through market or other side effects. The most commonly cited example is a forest preservation project, where protecting one area simply results in destruction of another parcel. The number of potential leakage points in any particular project, as well as the likelihood and magnitude of the potential leakage, will affect the project's credibility.

Other credibility issues include how long project benefits persist (the longer the better) and whether project activities might have happened anyway (additionality). Projects that achieve nothing beyond what would be expected under the status quo obviously will lack climate change mitigation credibility. Many energy sector activities, for example, also make good economic sense; power companies might reasonably pursue them in the absence of emissions reduction incentives. As a result, their additional value as offsets is questionable. The extent to which a project's actions result in true reductions from projected emissions is often referred to as its "additionality."

*B. Offset Reliability*

Reliability refers to the likelihood that an offset project will be implemented as intended and that it will achieve the anticipated $CO_2$ benefits. While credibility concerns the project concept, reliability focuses more on the reality of implementation. Relevant reliability issues would include the degree of control that offset funders have over project implementation variables; susceptibility of the project to economic and political conditions; and susceptibility of the project to environmental disturbances, include climate change itself, drought, flood, and insect infestations. Project funders may also consider the experience of those who will implement the project, and whether the project's basic concept has a proven track record. Poor ratings in any of these categories could discourage investment.

*C. Offset Quantifiability*

The ability to obtain accurate estimates of $CO_2$ benefits is an increasingly important component to $CO_2$ offset projects. The easier it is to measure and quantify $CO_2$ benefits, the more the project's credibility and cost-effectiveness will be enhanced. It will also be easier to assess the project's reliability. Relevant issues surrounding quantifiability include whether a project's impacts can be measured empirically or whether they need to be modeled; whether unintended project impacts (including leakage) can be measured; and the range of error for empirical measurements.

*D. Offset Cost-Effectiveness*

It is often difficult to compare offset costs among different alternatives. The same offset project pursued in two distinct locations, for example, could require different funding levels. Discounting carbon benefits that accrue in the future can significantly affect the apparent per ton costs of carbon. Carbon monitoring and verification can add significantly to project costs. Beyond simple $/ton estimates, other variables can affect an investor's evaluation of project cost-effectiveness. For example, project investors are likely to consider the project's expandability and replicability (replicating and expanding a project could lower future per ton costs); the percentage of project costs that must go towards monitoring and verification; and how easily monitoring and verification activities could be amended over time, given new science and policy developments.

Most fundamentally, projects that do not stand the test of time from a regulatory and crediting standpoint will not prove to be cost-effective in the long run. Projects that meet high standards in credibility, reliability, additionality, and quantifiability will be more likely to qualify for favorable regulatory treatment in future years than weaker projects.

*E. Ancillary Benefits*

Though technically perhaps not as important as other evaluative criteria, the presence or absence of ancillary benefits is likely to play a role in deciding the merits of individual projects. The availability of non-$CO_2$ environmental, economic, and social benefits may make a project particularly attractive. For the private sector, ancillary benefits may provide significant public relations value. Private investors often will prefer projects with ancillary benefits in their own service territories or in regions where they may be seeking new business opportunities.

## Applying the Evaluation Criteria to Arid Land Carbon Offset Types

Though there are many ways to combat desertification on drylands, and drylands span many regions of the globe, it is still possible to assess generally how private sector investors may approach anti-desertification projects as possible carbon offsets. Applying the evaluative criteria outlined above highlights a number of issues that may make arid land projects seem less desirable to investors.

*A. Arid Land Offsets and Credibility*

The theoretical ability of arid land projects to mitigate climate change is clear. Whether such projects can constitute credible private sector carbon offsets is somewhat more problematic. Because the causes of desertification are so numerous and complex, finding workable and robust methods to combat them has proven to be far from simple. Anti-desertification programs are particularly susceptible to changing climate, population pressures, changing land use patterns, and political and economic pressures. Though there may be ample technical means for preventing or reversing land degradation, degradation may continue in the face of these pressures.

Almost all projects that aim to reform land management must do so primarily by changing people's behavior. Such programs are often complex, cover large land areas, and require a high degree of community coordination. Though some fraction of project activities may have direct carbon benefits—tree planting or dune revegetation, for example—many others will sequester carbon only indirectly.

Offsets based on agricultural reform would involve educating and training individuals in improved cultivation methods, who may then transfer their skills to other members of their community and region, which will hopefully lead to widespread soil conservation and significant accumulation of soil carbon. The carbon benefit of such programs is clearly several steps removed from the activities being funded. In addition, success may be highly contingent upon local politics, which could influence how widely the programs are accepted, and economics.

The duration of land management projects may be an issue as well. The proposed UNSO projects, for instance, assumed that the practices they initiated over a five-year period would be carried out into the indefinite future, and the resultant future (and undiscounted) carbon benefits figured heavily into the projects' cost-effectiveness.

Certain land management programs may suffer the same credibility issues faced by forest protection projects—namely, to what extent would the land in question have been degraded in the absence of program activities? In most cases, however, additionality probably would not be an issue; the advancement of desertification worldwide is clear and efforts to stop it have been too few and ineffectual. Likewise, carbon "leakage" is not likely be an issue for most arid land offset projects. To the extent that project activities take place outside the traditional economic sector, for example, there are unlikely to be many unexpected market side-effects.

Projects involving cultivation of halophytes or tree planting may fare better in terms of credibility than projects focused on land management, since these projects have fewer variables to control, carbon benefits are relatively more direct, and success may be tied directly to the project's economic success. Still, to the extent that they must be implemented over large areas and involve the coordination of many people in countries with unstable social and economic conditions, they may be perceived as lacking credibility.

Generally, the perceived social, economic, and political context surrounding many arid lands projects may detract most from their credibility. Many countries with the largest expanses of degraded and desertified lands face increasingly uncertain economic conditions, prospects of growing social upheaval, and political instability. This is especially true of Africa, where a third of world drylands are located (Dregne, *et al.*, 1991). These conditions may scare off private sector investors, who find them too risky.

*B. Arid Land Offsets and Reliability*

Any program that relies for its success on changing human behavior is likely to be less reliable than projects where controlling physical variables is more direct. On balance, carbon offset projects that try to address mismanagement of drylands may therefore be less reliable than other forestry or energy-based projects. The track record of anti-desertification programs to date seems to support this conclusion. Many programs, such as the U.N. Plan of Action to Combat Desertification, have failed for numerous reasons, ranging from lack of funding from the international community and local governments, to poor understanding of the problem by authorities in affected regions, to political instability and hostilities (Buonajuti, 1991). In any case, benefits of dryland offset projects are likely to be fragile compared to those of other options available to potential funders. The carbon gained through many projects could be suddenly lost at any point during a project.

The success or failure of dryland offset projects may depend to a significant extent on variables outside the control of offset funders. Dryland regions are particularly susceptible to adverse climatic conditions and drought, which may jeopardize otherwise successful programs. Crop management programs may be susceptible to insect infestations and other biological threats given the harsh growing conditions in these regions. Of equal concern is the economic, political, and social conditions in many areas. Even the best planned and best executed projects can break down in the face of economic hardship, political hostilities, and social upheaval. Unfortunately, the countries with the most desertified lands tend to be the ones with the strongest propensities for these conditions.

Lack of project control is often inherent in the implementation of anti-desertification projects as well. In most cases, these projects do not involve the purchase of land. The nature of land management reform projects, where education in agricultural techniques is assumed to spread throughout a community or to surrounding areas, makes it difficult to control the outcome of project activities, and would make establishing contract rights to sequestered carbon quite difficult. Halophyte cultivation and tree planting projects may be able to overcome some of these shortcomings, but these projects are still susceptible to larger environmental and social variables.

*C. Arid Land Offsets and Carbon Benefit Quantifiability*

Though it may be possible to empirically measure the success of dryland management reform projects in terms of halting degradation, carbon benefits will likely require extrapolation. Extrapolation could become complicated when one takes into account the combined effects of increasing crop residue retention, applying fertilizers, reducing fallow, and shifting to mixed crops on arid croplands, not to mention conservation measures on pasture and rangeland. Estimating the carbon benefit of these effects is possible given a fairly reliable baseline (though again, the baseline may be an issue as well), but empirical monitoring and verification will be difficult. The costs of monitoring could prove prohibitive.

Many dryland management projects presume that project activities will be carried on after they are initiated and that they will be replicated by surrounding communities. This assumption also presents a difficulty for quantification. How does one measure the degree to which such secondary effects take place? Successful carbon offset projects would have to resolve this issue.

Carbon sequestration from halophyte cultivation and tree planting is probably easier to empirically quantify. However, the range of error in empirical measurements may be significant. Measuring sequestration from halophytes, for example, would involve periodic soil carbon monitoring. For tree planting projects involving jatropha, quantifying the amount of fossil fuel displaced by burning seed oil may be difficult where there is poor access to fossil fuels. The carbon benefits of straight plantations of either halophytes or trees may be easiest to quantify, but such programs will also be more costly.

*D. Arid Land Offsets and Cost-Effectiveness*

Estimates of arid land carbon offset cost-effectiveness are generally still preliminary. Even those figures that do exist, such as UNEP's projected range of US $10.0 to US $22.4 billion per year for a global program to combat desertification, are hard to compare to the costs of other carbon offset options. If the actions envisioned under this program were to be carried out for the purposes of carbon offsets funded by the private sector, the total cost would likely be much higher. The reason is that UNEP's estimate does not include the costs of packaging and implementing projects as carbon offsets, nor does it include subsequent costs of monitoring and verification which would be necessary for these offsets to be credible.

Given the credibility and reliability issues outlined above, the costs of monitoring and verification could easily make up a significant percentage of total cost for these projects.

Cost estimates for halophyte cultivation and tree planting projects give a better idea of some more project-specific costs, although monitoring and risk are largely not factored in. Generally speaking, it is not clear if there are opportunities for cost-sharing in the case of dryland carbon offsets. Anti-desertification measures assuredly have economic benefits in the long run, but gestation periods are long and returns tend to be low (Kassas, *et al.*, 1991). It is unlikely that farmers benefiting from new forms of cultivation would be willing or able to contribute towards the cost of implementing them (unlike foresters, who in many cases may be willing to help cover tree planting costs if they are later able to harvest the trees). This may hurt the overall cost-effectiveness of dryland projects compared to other offset options.

Finally, it is not clear how easily monitoring and verification procedures could be updated in the light of new scientific or policy developments, adding another element of uncertainty to potential cost. And because land management programs are often geographically specific and tailored to relatively small groups of communities, they may not be as replicable or expandable as other offset projects. Economies of scale that could help future project cost-effectiveness are therefore unlikely (though this may be a less significant issue with halophyte cultivation or tree planting). Overall, arid land offset projects on average are unlikely to be at the bottom or even close to the bottom of a global carbon offset supply curve. This raises questions as to when such projects would be considered cost-effective for private sector investment funding.

*E. Arid Land Offsets and Ancillary Benefits*

Whatever possible shortcomings that offset projects in drylands may have in terms of more technical criteria, their ancillary benefits are unquestionable. Desertification is a serious global problem and threatens to undermine the economies and natural resource base of many developing countries. Tree planting and improved cultivation can assist local communities in bettering their livelihood and securing their economic future.

Halophyte cultivation can restore economic productivity to salinized lands. Whether private investors will want to capitalize on these benefits is another issue. Most investors will be looking for public relations benefits, and thus will prefer projects either in their own service territories or in regions where they are seeking to develop new business. This puts at a disadvantage the extensive dryland areas outside of Latin America and Asia.

### A General Comparison of Arid Land with Other Biotic and Energy-Based Offset Options From the Standpoint of Private Sector Funding Objectives and Priorities

While the theoretical potential exists to sequester sizable amounts of carbon on drylands, the attractiveness to private carbon offset investors of projects to accomplish this is somewhat suspect. If nothing else, these projects suffer from a perceptional disadvantage due to their primary location in areas outside where potential investors have customers or business interests and in countries with relatively unstable political and economic conditions. Moreover, in a general ranking of offset options, most arid land projects are likely to score relatively low.

Arid land offset projects face a number of potential problems with regard to the evaluative criteria likely to be used by offset investors. Given these problems, and the general perceptions that private investors may have of these projects, it seems unlikely that they will be preferred over other options. In the past several years, advocates of offset projects on arid lands have made numerous proposals, some with dubious credibility. As mentioned above, the "Eden Project" endeavors to plant enough trees to fully stabilize atmospheric $CO_2$ levels, an undertaking that supposedly would require blanketing drylands with 230 billion trees. Projects like these are often perceived as grandiose by project investors and have not helped the cause of those advocating more reasonable and well-thought-out projects.

In a general ranking of offset options, private investors are likely to look first to energy sector projects because of their direct credibility, robust benefits, and the perceived potential for financial return. Land use projects (including forestry) will often take a secondary position, because their carbon benefits are more fragile and subject to more intervening variables. Compared to other land use projects offsets on drylands are even more suspect. They suffer from greater complexity, less direct links between project activities and carbon benefits, more difficult measurability and verifiability, and greater or more uncertain costs. In addition, many arid land projects would necessarily take place in areas where the political context is less than ideal.

Private investors looking for offset investments that they can likely credit against a future regulatory regime will often shy away from projects on drylands. Many energy sector projects have well-defined costs and carbon benefits, and can in theory even pay for themselves. By contrast, the mechanisms and costs of many arid land offset projects are at this point poorly defined and developed. Though initial cost per ton estimates of dryland carbon offsets appear comparable to other offset options, most estimates proposed so far have been superficial. Adjusted for risk, discounting, and other variables, the costs of most arid land projects would probably be quite a bit higher than those for other available offset options. This fact makes it unlikely that large amounts of private sector carbon offset funds will be made available to help combat desertification.

### Designing Arid Land Projects in the Face of These Difficulties

Although the barriers are significant, there undoubtedly are limited opportunities for attracting private sector money to projects on arid lands. Halophyte and jatropha plantings show some promise, and may have less fragile benefits than many agricultural land management programs. To package these programs for private investors, however, they will have to be well thought out in terms of evaluative criteria, including having well-defined costing methodologies. In addition, success for these programs will likely mean identifying a specific target audience for funding. The most likely candidates would be power producers with specific interests in affected regions. For example, European companies are much more likely to be interested in Africa projects than U.S. companies.

During this period of policy development in the climate change mitigation arena, arid lands interest groups should attempt to structure projects with a high potential for success. Until more experience with offset projects has accumulated and methods of carbon accounting and program implementation are uniformly recognized by funders, regulators, and host countries, careful consideration must be given to the design and implementation of carbon offset and joint implementation projects. Such consideration will increase the likelihood that the proposed project will stand the test of time substantively, economically, and politically. Project developers are now proposing many ideas that fail to pass even a basic "laugh" test, and the offset integrity of even already funded projects varies widely (Trexler and McFall, 1993).

The following guidelines are representative of discussions held thus far on the legitimacy and implementation of future offset programs. These guidelines seem likely to form the foundation of a future regulatory system that credits offset projects against mandatory $CO_2$ controls or taxes (Trexler, Kosloff, and Gowen, 1994).

1. <u>Simplicity</u>: Project design should clearly favor offset projects in which the emissions reduction or carbon sequestration directly results from the proposed project, and is not a second or third-order effect of the project intervention itself. The more complex the set of linkages between initial actions and carbon sequestration, the less credible the approach may ultimately prove. Purchasing threatened tropical forest, for example, is a more direct intervention than promoting nearby agroforestry in the hope that the forest will be indirectly protected.
2. <u>Credibility</u>: Project design should clearly favor offset projects in which the political and economic context, as well as the track record of project proponents, show that prospects for long-term project success are as high as possible. For example, proposing a forestry extension project in a country with no effective infrastructure should be viewed with caution.
3. <u>Incremental Effect</u>: Project selection and design should favor projects in which a "but for" baseline can be securely established.
4. <u>Verifiability and Measurability</u>: Project selection and design should build in verification and measurement protocols and should consider the need for retrofitting more stringent protocols later.
5. <u>Positive Secondary Benefits</u>: Project selection and design should emphasize the identification and promotion of ancillary environmental and economic benefits to the region, particularly for joint implementation projects where property rights and other project safeguards may not be well-established. Given political charges of "carbon colonialism" and the

drawing of analogies between exporting carbon and exporting hazardous waste, project developers should be aware of the political environment in which a project will be reviewed. Aggressive promotion of non-carbon project benefits will likely garner goodwill in the project region, which will translate into an increase in the long-term reliability of the project's carbon offset and increased opportunities for future project expansion.

6. Supportive Political Context: Project selection should focus on countries with an expressed interest in hosting joint implementation projects. Even before formal crediting systems are put into place, project contracts signed by government representatives should provide for allocation of carbon credits to the funding entity.

7. Cost-Effectiveness: Project selection and design should emphasize project cost-effectiveness, considering the various carbon accounting frameworks that may ultimately be established. These include front-loaded, averaged, or strictly count-as-you-accrue carbon counting mechanisms. Project proponents, however, should be wary of schemes to significantly leverage their carbon offset investment so that the cost per ton is significantly reduced. Funding just 10% of the cost of a forest protection project, for example, may not ultimately entitle the funding entity to claim all of the carbon benefit, particularly if the balance of the funding is from governmental or other sources that may have a future interest in accruing their own carbon credits.

At the same time, it may be legitimate to pay just the incremental cost needed to make an otherwise economically marginal project financially viable. Project funders should evaluate the cost-effectiveness of a project under both optimistic and pessimistic scenarios of how much carbon credit they may ultimately be judged to be entitled to. Given the significant overhead costs of even small pilot projects, project selection should emphasize project expandability to promote long-term potential cost-effectiveness.

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# The U.S. electric utility industry and forest carbon management

**John D. Kinsman and Gary Kaster**

## Synopsis

This paper presents the outlook of the U.S. electric utility industry regarding controlling greenhouse gases via biotic (vegetation and soil) carbon management. Domestic and international policy regarding biotic carbon offsets is discussed. Two voluntary U.S. electric utility programs will be described: the Climate Challenge Program and the Utility Forest Carbon Management Program. The latter program is specifically designed for U.S. electric utilities to advance the state of knowledge regarding options for managing greenhouse gases via forestry; establish low-cost forestry options to manage greenhouse gases; promote environmental stewardship by the utility industry; and implement projects to manage greenhouse gases.

Key Points

1. Human activities related to energy production and land use are increasing the atmospheric concentration of greenhouse gases such as carbon dioxide ($CO_2$), which in turn may change the energy flux of the Earth and its atmosphere, possibly causing global warming and other changes in climate. The impacts of greenhouse gas emissions are very uncertain, however – the rate, magnitude and regional characteristics of human-induced climate change are difficult to predict and require much additional research. Regardless of the potential consequences of climate change, policies are being developed and programs are being implemented to adapt to or mitigate greenhouse gases and climate change.
2. Many options exist for managing greenhouse gas emissions and sinks. These include: increasing the efficiency of energy supply and use, including use of environmentally beneficial electrotechnologies; increased use of renewable and nuclear energy systems; fuel switching from coal and oil to natural gas; capturing and using methane from coal mines and landfills; and increased motor vehicle fuel economy. In addition, adaptation (e.g., planning for sea-level rise, or planting different crops) would lessen adverse impacts.
3. Sequestering $CO_2$ in "sinks" such as plant biomass is another option. For example, carbon might be sequestered in growing trees, in harvested trees (forest products), in halophytes (salt-tolerant plants), as organic matter in soil, in oceanic seaweed farms, or in microalgae in the ocean. Alternatively, biomass can be used as a substitute for fossil fuel to produce energy.
4. Trees are referred to as "carbon sinks," because they take $CO_2$ out of the air and sequester it in living plant tissue. About one-half of a tree is carbon. As a tree approaches maturity, its rate of net carbon uptake declines. Averaged over time, a mature forest is in near carbon equilibrium as the uptake via photosynthesis is balanced by releases via respiration and decay. One way to prolong the active role of forestry in sequestering carbon is to engage in periodic harvesting, thus setting the stage for a new growth cycle. To maintain the carbon benefit, however, the harvested wood must be used in long-lived products such as lumber in construction. Another option is to increase recycling of building material as well as paper products.

5. Dead plant tissue in and on top of soil also stores carbon until it decomposes. Forestry management practices can increase the uptake of carbon in biomass and soil, and can maintain existing stores of carbon out of the atmosphere. The technical potential for forest carbon management is great, able to counteract a meaningful portion of the 3 petagram (Pg, equal to 3 billion metric tons) carbon annual addition to the atmosphere. Forest carbon management is promoted in all domestic and international climate policies and treaties. Carbon can be managed through many different types of forestry activities.

6. Electric utilities are interested in all technically and economically feasible alternatives for managing greenhouse gases emissions. Most commonly, utilities will concentrate on energy supply and energy demand activities to manage emissions. Other activities include management of terrestrial (e.g., forest) carbon, recovery and use of landfill methane, joint implementation of projects in other nations, and transportation-sector reductions. When an emissions source manages its emissions indirectly by effecting a reduction at another source, it is said to be "offsetting" its emissions. The emission of a ton of $CO_2$ can be negated or offset by avoiding the release of a ton elsewhere, or by removing a ton of $CO_2$ from the atmosphere.

7. Hundreds of millions of hectares around the world that previously supported tree cover could do so again. Formerly sustainable slash and burn agricultural systems have become unsustainable under the pressures of growing populations; salinization, soil compaction and erosion have rendered agricultural land and pasture unproductive over many millions of hectares; and many millions of hectares have been degraded through logging, fuel wood collection, grazing, and fire. In each of these cases carbon storage on the land declines, often dramatically. While some degraded forests recover on their own, others do not -- for example, fuel wood pressures might be too intense, logging interventions might be too frequent, or heavy undergrowth and vines might impede tree regeneration. In many areas the control of wildfire would result in the regeneration of forest cover over large areas.

Key words. Carbon, carbon dioxide, forest, electric utilities, management

## Introduction

Human activities related to energy production and land use are increasing the atmospheric concentration of greenhouse gases such as carbon dioxide ($CO_2$), which in turn may change the energy flux of the Earth and its atmosphere, possibly causing global warming and other changes in climate. The impacts of greenhouse gas emissions are very uncertain, however -- the rate, magnitude and regional characteristics of human-induced climate change are difficult to predict and require much additional research. Regardless of the potential consequences of climate change, policies are being developed and programs are being implemented to adapt to or mitigate greenhouse gases and climate change.

The United Nations Conference on Environment and Development (UNCED), held in Rio de Janeiro in June 1992, was the forum for the signing of the Framework Convention on Climate Change by 154 countries. The Convention commits all parties to develop and make available national inventories of anthropogenic emissions by sources and removals by sinks for all greenhouse gases. The Convention also requires developed country parties, including Eastern European countries and the Newly Independent

States, to adopt policies and implement measures to mitigate climate change by limiting emissions of greenhouse gases and enhancing sinks and reservoirs.

The U.S. was the first industrialized nation to ratify the Convention, in October 1992. An important provision of the Convention is "joint implementation," which involves partnerships between parties in different nations to manage greenhouse gases. On Earth Day in April 1993, President Clinton committed the U.S. domestically to honor the greenhouse gas emission reduction goal of the Convention, "reducing our emissions of greenhouse gases to their 1990 levels by the year 2000." At the first meeting of the Conference of Parties (COP-1) to implement the Convention, held in Berlin during April 1995, a process was initiated to address future actions in 2005, 2010 and 2020, to culminate with a decision at COP-3 in 1997-98. A new aim for post-2000 greenhouse gas management may result.

In the U.S., the Climate Change Action Plan was announced in October 1993 with an emphasis on fostering partnerships between the government and the private sector by using voluntary programs to reduce greenhouse gas emissions. Such voluntary programs include the Climate Challenge, Climate Wise, Green Lights and the Motors Challenge. Section 1605 of the Energy Policy Act of 1992 established a program to develop guidelines for the voluntary collection and reporting of information on greenhouse gas emissions and reductions.

The list of "annual reductions of greenhouse gas emissions and carbon fixation achieved through any measures" includes fuel switching, forest management practices, tree planting, use of renewable energy and other actions. Final guidelines were published in October 1994 (U.S. Department of Energy, 1994) and reporting forms were released in July 1995 (Energy Information Administration, 1995).

Many options exist for managing greenhouse gas emissions and sinks. These include: increasing the efficiency of energy supply and use, including use of environmentally beneficial electrotechnologies; increased use of renewable and nuclear energy systems; fuel switching from coal and oil to natural gas; capturing and using methane from coal mines and landfills; and increased motor vehicle fuel economy. In addition, adaptation (e.g., planning for sea-level rise, or planting different crops) would lessen adverse impacts.

Another option is to sequester $CO_2$ in "sinks" such as plant biomass. For example, carbon might be sequestered in growing trees, in harvested trees (forest products), in halophytes (salt-tolerant plants), as organic matter in soil, in oceanic seaweed farms, or in microalgae in the ocean. Alternatively, biomass can be used as a substitute for fossil fuel to produce energy.

The subject of this article is the management of carbon in trees from the electric utility industry's perspective. The Climate Challenge Program and the Utility Forest Carbon Management Program, which are expanding utility industry interest in managing greenhouse gases via forestry, will be described.

## Electric Utilities, Greenhouse Gases and Their Management

U.S. electric utilities are a major source of $CO_2$ emissions, primarily from burning coal, oil and natural gas. For the United States, in 1990 electricity used to power residential energy uses resulted in $CO_2$ emissions of 162 million metric tons of carbon (MMTC), compared to 148 MMTC to power commercial sector energy needs and 166 MMTC to meet industrial sector energy requirements (Energy Information Administration, 1994). All told, U.S. electric utilities emitted 477 MMTC in 1990, or 34.7 of U.S. energy and industry $CO_2$ emissions. This equates to about 7.8% of global energy-related $CO_2$ emissions (Boden et al., 1994).

Electric utilities are interested in all technically and economically feasible alternatives for managing greenhouse gases emissions. Most commonly, utilities will concentrate on energy supply and energy demand activities to manage emissions. Other activities include management of terrestrial (e.g., forest) carbon, recovery and use of landfill methane, joint implementation of projects in other nations, and transportation-sector reductions. When an emissions source manages its emissions indirectly by effecting a reduction at another source, it is said to be "offsetting" its emissions. The emission of a ton of $CO_2$ can be negated or offset by avoiding the release of a ton elsewhere, or by removing a ton of $CO_2$ from the atmosphere.

The use of emissions offsets in environmental management is not new, having been applied in some cases for as long as 15 year in the U.S. Carbon dioxide offsets, however, exist within a different scientific and regulatory context. First, stack-based $CO_2$ controls are both extremely expensive and present severe practicality problems (Fluor Daniel, 1991). Second, $CO_2$ is long-lived in the atmosphere, mixes globally, and thus can be offset anywhere in the world. Third, $CO_2$ is different from other emissions, because it can be practically removed from the atmosphere after being emitted.

In the U.S. and elsewhere, electric utilities are managing greenhouse gases through voluntary, cost-effective actions and programs. Draper (1994) and De Michele (1994) believe that there are incentives for voluntary actions by the electric utility industry to reduce greenhouse gases:

- The U.S. is a signatory to the Framework Convention on Climate Change, and President Clinton has pledged action to limit the nation's greenhouse gases.
- Public discussion and action may not wait for final scientific conclusions.
- For the sake of customers and shareholders, utilities must pro-actively retain the operational flexibility to achieve greenhouse gas reductions using the most cost-effective methods. To ensure the opportunity to use market-based approaches and avoid costly and inefficient one-size-fits-all approaches, utilities must work with federal and state agencies.
- The utility industry has unique contributions to make in greenhouse gas management, possessing special competence in providing cost-effective customer service and in achieving environmental excellence through technical innovation, such as energy-efficient electrotechnologies; supply-side efficiencies through clean coal technologies, nuclear energy, natural gas, and renewable energy technologies; and demand-side management.

The Federal government has its own reasons for supporting a voluntary approach, including cost-effectiveness and the fact that a legislative/regulatory approach would take years to implement.

## The Climate Challenge

The Climate Challenge Program is a joint, voluntary partnership between the U.S. electric utility industry and DOE to reduce, avoid or sequester greenhouse gases. The Climate Challenge is the cornerstone of the electric utility industry's approach to managing greenhouse gases. As of late 1995, DOE reported that close to 600 utilities, through over one hundred signed participation accords, have committed to 47 million metric tons of carbon equivalent (MMTCE) of reductions. In other words, without these actions, emissions from these utilities in the year 2000 would be 47 MMTCE higher than in 1990.

Sturges and Hewitt (1994) surveyed 19 utilities regarding the Climate Challenge and concluded that the majority of the utilities interviewed are taking the program seriously, for the following key reasons:

- It is important for the company to provide support for the voluntary approach taken by the Administration.
- The program is consistent with corporate beliefs about appropriate environmental impact mitigation activities.
- Companies want to be pro-active and help shape policy responses and programs affecting the industry.
- Companies want to demonstrate cost-effective solutions to climate change.

The implementation of the Climate Challenge is based on these general principles:

- Utility activities will be voluntary and flexible and may cover any greenhouse gas.
- The program includes opportunities for industry-wide and utility-specific activities.
- The program recognizes that many activities have been or will be undertaken for business and other reasons, while also reducing, avoiding or sequestering greenhouse gases.
- Utility activities may be cross-sectoral in nature, i.e., they may occur outside of utility facilities or their impacts may occur outside of the utility sector. Examples of these types of cross-sectoral activities include: electrotechnology and electrification programs, which may result in net reductions in greenhouse gas emissions overall while increasing electric utility emissions; demand-side management programs; forestry and other sink-related activities; and international activities.
- Activities will be cost effective and take into consideration impacts on ratepayers and shareholders, competitive situations of the utilities with regard to costs and rates, resource planning, and regulatory mandates.
- It is recognized that utility-specific circumstances (e.g., growth requirements, fuel mix, geography, power supply resources, financial resources) vary and that the actions taken and results achieved may vary accordingly.
- Climate Challenge activities may overlap with activities from other programs, such as Climate Wise, Motor Challenge, Green Lights and Natural Gas Star.

- o DOE will work with other agencies of the federal government to assist the utility industry to the maximum extent possible in reducing or eliminating legal, institutional, economic, market, informational and other barriers to implementation of activities.
- o Each electric utility choosing to participate in the Climate Challenge Program shall sign an instrument with DOE. In that agreement, the utility will make one or more of the following types of commitments:
  1. Make a specified contribution to particular industry initiatives.
  2. Reduce greenhouse gas emissions by a specified amount below the utility's 1990 baseline level by the year 2000.
  3. Reduce greenhouse gas emissions to the utility's 1990 baseline level by the year 2000.
  4. Reduce greenhouse gas emissions by or to some other specified level.
  5. Reduce or limit the rate of greenhouse gas emissions to a particular level, expressed in terms of emissions per kiloWatt-hour generated or sold.
  6. Undertake specific projects or actions, or make specific expenditures on projects or actions, to reduce greenhouse gas emissions.
- o Participants will report annually on activities and achievements in a clear and understandable manner that is consistent with the guidelines adopted pursuant to subsection 1605(b) of the Energy Policy Act.
- o Participants will confer with DOE at reasonable intervals to jointly evaluate programs and discuss possible adjustments to voluntary commitments.

A Climate Challenge Options Workbook (U.S. Department of Energy and Electric Utility Industry, 1995) identifies approximately 50 different types of opportunities for greenhouse gas reductions, addressing barriers to implementation; possible solutions to overcome the barriers; potential partnerships that could be developed to implement the option; and case studies. Options are organized according to area of application: end use; renewable energy generation technologies; other generation technologies; transmission; distribution; transportation; and other (e.g., forest carbon management, methane management and use, and use of coal combustion byproducts).

For the Climate Challenge, the Edison Electric Institute has led or supported development of five major initiatives for the utility industry to support jointly. These initiatives address electrotechnologies and renewable energy; international energy projects; geothermal heat pumps; electric vehicles; and forest carbon management. The first four of these initiatives are described in the next paragraph.

Through the **EnviroTech ℠ Investment Fund**, the industry has established several limited partnership funds to invest in promising electric technologies and renewable energy technologies, managed by leading professional venture capital fund managers. As of late 1995, two funds have attracted more than $50 million in investment. The **International Utility Efficiency Partnerships** is an organization comprised of utility and non-utility organizations that will pursue the following goals: 1) further the concept of joint implementation; develop working relationships between parties; and 3) identify and participate in international energy projects. Through the **National Earth Comfort Program**, the Geothermal Heat Pump Consortium (an organization of electric utilities and their institutions; equipment manufacturers; other allies; and environmental organizations, including the U.S. Environmental Protection Agency), has a goal of increasing geothermal heat pump annual unit sales from 40,000 to 400,000, which is estimated to reduce annual greenhouse gas emissions by 1.5 million metric tons of carbon annually. Among the program objectives is to reduce the cost barrier to wide-scale customer acceptance of this technology by developing the infrastructure needed to reduce front-end costs. Finally, **EV America** is a partnership between the electric utility industry and governmental entities to demonstrate the commercial potential of electric vehicles (EV's) in the United States by placing electric vehicles in fleet applications in various regions of the country and demonstrating the environmental (reducing greenhouse gases and other major auto pollutants) and energy security benefits from using electric modes of transportation. The fifth initiative is the Utility Forest Carbon Management Program which is described below.

## Management of Biotic Carbon

Trees are referred to as "carbon sinks," because they take $CO_2$ out of the air and sequester it in living plant tissue. About one-half of a tree is carbon. Dead plant tissue in and on top of soil also stores carbon until it decomposes. Forestry management practices can increase the uptake of carbon in biomass and soil, and can maintain existing stores of carbon out of the atmosphere.

The technical potential for forest carbon management is great, able to counteract a meaningful portion of the 3 petagram (Pg, equal to 3 billion metric tons) carbon annual addition to the atmosphere (Sampson et al., 1993; Dixon et al., 1993). Forest carbon management is promoted in all domestic and international climate policies and treaties.

Carbon can be managed through many different types of forestry activities (Kinsman and Trexler, 1993):

**Forest Protection.** Protecting or managing standing forests can be an attractive means of implementing a carbon offset program. Tropical forests are being cleared for timber export, fuel wood, shifting cultivation, permanent agriculture, pasture, and urbanization and infrastructure (Postel and Heise, 1988). Millions of hectares (ha) of forest are cleared annually (Houghton et al., 1992). Deforestation in tropical latitudes is responsible for 10 to 30% of anthropogenic CO2 emissions.

**Forest Management.** Improved forest management practices (e.g., thinning) can lead to increased carbon uptake and also reduced carbon releases. For example, in Malaysia, the New England Electric System and Innoprise, a leading Malaysian forests products corporation, are involved in an offset project designed to reduce $CO_2$ released during the logging process on 1,400 ha. Elements of the project include improved siting of logging trails, directional felling of trees, and vine removal prior to harvest, all intended to decrease damage to undergrowth and unharvested trees during the logging process, thereby decreasing $CO_2$ released to the atmosphere as well as facilitating regeneration of the forest.

**Improved Management of Degraded Lands.** Hundreds of millions of hectares around the world that previously supported tree cover could do so again. Formerly sustainable slash and burn agricultural systems have become unsustainable under the pressures of growing populations; salinization, soil compaction and erosion have rendered agricultural land and pasture unproductive over many millions of hectares; and many millions of hectares have been degraded through logging, fuel wood collection, grazing, and fire. In each of these cases carbon storage on the land declines, often dramatically. While some degraded forests recover on their own, others do not -- for example, fuel wood pressures might be too intense, logging interventions might be too frequent, or heavy undergrowth and vines might impede tree regeneration. In many areas the control of wildfire would result in the regeneration of forest cover over large areas.

**Agroforestry.** Agroforestry, the incorporation of trees with agricultural and other practices, can play a significant role in carbon offset projects, particularly when combined with forest regeneration or protection. These types of projects can be consistent with the economic development goals and can help reduce pressures on surrounding forested areas. These projects require intensive involvement of local communities and site-specific tailoring and education.

**New Plantations.** Tree plantations on pasture, agricultural land or degraded forest sites can offer rapid growth rates over large areas of land, along with uniform management and quantifiable costs and benefits (Sedjo and Lyon, 1990).

**Wood Products.** As a tree approaches maturity, its rate of net carbon uptake declines. Averaged over time, a mature forest is in near carbon equilibrium as the uptake via photosynthesis is balanced by releases via respiration and decay. One way to prolong the active role of forestry in sequestering carbon is to engage in periodic harvesting, thus setting the stage for a new growth cycle. To maintain the carbon benefit, however, the harvested wood must be used in long-lived products such as lumber in construction. Another option is to increase recycling of building material as well as paper products.

**Soils.** Globally, soils contain about 150 to 300% as much carbon as above-ground biomass (Dixon and Turner, 1991). Some management practices (e.g., cultivation and intense prescribed fire) lead to soil carbon loss, while crop fertilization and reduced tillage can increase carbon storage (Johnson, 1992). In general, increased carbon accumulation in soil is associated with practices that promote cooler soils (e.g., mulch and shade), wetter soils (irrigation), more fertile soils and soils with reduced aeration (limited tillage). It may be difficult, however, to boost carbon storage in cultivated dryland soils by no-till farming (Executive Summary, this volume). Reversion of agricultural land to forest leads to significant soil carbon storage (Brown et al., 1992; Sedjo, 1992).

**Drylands.** Approximately 40% of the Earth's land surface is drylands (United Nations Environment Programme, 1992). Since world dryland soils contain 240 Gt of organic carbon, small increases in storage/small decreases in releases can be significant and vigorous efforts to control land degradation in these areas could result in a net sequestration of up to 1.0 Gt C per year (Executive Summary, this volume). In these systems, most techniques to mitigate desertification will increase storage of carbon in vegetation and soil: e.g., relaxing grazing intensity, fertilization and residue management in dryland crop management, enhanced bush encroachment in semi-arid savannas, introduction of legume trees into grass pastures, energy crops, increased biofuel use efficiency, agroforestry, improved pasture management, savanna fire control and woodland management (Executive Summary, this volume).

To this list should be added the use of drought-tolerant plant species, water-efficient crops, and soil conservation techniques. Perhaps even more so than in non-arid regions of developing nations, considerations of local inhabitants are paramount when designing dryland management, as under harsh conditions there can be extreme land use pressures. Programs of desertification control that provide sustainable land management lead to improved conditions (with carbon benefits if properly managed) for inhabitants through increased biodiversity, food production, woodland products, livestock feed, and reduced hazards such as floods and soil erosion (Executive Summary, this volume). However, the vagaries of climate, land use, political instability, severe poverty and lack of property rights (Mabogunje, 1995; Migongo-Bake, this volume) may make carbon management in drylands fairly risky. One particular carbon management option for drylands is addressed in detail by Glenn et al. (1992 a,b) -- carbon sequestration in halophytes (salt-tolerant desert plants). About 130 million of the world's 700 million ha of salt desert habitat could support halophytes, with carbon sequestration rates comparable to those of tree plantations, and halophytes could potentially sequester up to 0.7 Pg carbon per year (Glenn et al., 1992a). The harvested biomass could be stored in desert soils if its decomposition is slow, or it could be burned to produce energy. A key advantage of using halophytes is that they grow in saline soils that are useless for conventional agriculture and thus can be irrigated with saltwater, avoiding the use of fresh water.

**Trees for Energy Conservation.** Because of the replacement of soil and vegetation with concrete, asphalt and metal, many urban areas have experienced a heat island effect characterized by a several degree higher temperature than in nearby rural areas. The heat island effect has caused the need for an additional 1500 MW of electric power plants in Los Angeles (U.S. Environmental Protection Agency, 1992). Trees can counteract this heat island effect through the process of evapotranspiration -- a tree can transpire up to 100 gallons of water per day, equivalent to the cooling effect of 100 hours of air conditioners in a hot, dry climate (U.S. Environmental Protection Agency, 1992). In addition, shade trees can reduce the requirement for cooling residences and buildings, sometimes offsetting the need for fossil fuel use and reducing $CO_2$ emissions. Shade tree planting on the south and west sides of a home can reduce air conditioning needs by 10 to 50% (U.S. Environmental Protection Agency, 1992). These trees also sequester carbon from the atmosphere and can serve as windbreaks, reducing winter heating requirements. A guidebook has been published on strategic landscaping for energy conservation, describing tree planting and surface albedo modification (U.S. Environmental Protection Agency, 1992).

However, there are limitations to urban tree planting for energy conservation purposes. Little energy is used for cooling in many parts of the world and even in many parts of the U.S. Proper maintenance of urban trees is difficult, and urban trees live notoriously short lives, succumbing to many different stresses such as drought, vandalism and urban air pollution. Proper species selection and location are key -- the U.S. electric utility industry spends approximately $1.5 billion annually and considerable energy clearing trees, which are the number one cause of electricity outages, away from power lines. Nonetheless, in many situations urban tree planting makes environmental and economic sense.

**Biomass as an Energy Source.** Wood or other biomass can be turned into a carbon offset through its conversion to energy if it is used in place of fossil fuel. Net $CO_2$ emissions are practically zero for a system where $CO_2$ released during biomass combustion is simultaneously sequestered by the next energy crop. Fossil fuel inputs to facilitate such an essentially closed system can be very low under some circumstances. In the U.S. at present, the use of tree biomass as fuel by utilities is economical only for small power plants near sources of waste wood.

In the future, a 100 MW power plant, operating at 35% efficiency, would require slightly more that 40,000 ha of land, or about 2% of the area within 80 kilometers, for energy crop plantations (assuming a feedstock yield of 24 green tonnes carbon per ha each year) (Turnbull, 1993). Research and development activities are focusing on promising energy conversion technologies (such as whole tree burning, biomass gasification, and co-firing wood chips with coal), plus improving feedstock yield. With advances in energy conversion and crop yield, short-rotation trees grown on a 6- to 12-year rotations have been estimated to have the potential to reduce U.S. fossil fuel $CO_2$ emissions by 20% (Graham et al., 1992). Preliminary information indicates that co-firing of wood chips with coal is technically and economically feasible in many cases. In developing nations, biomass energy development leading to fossil fuel displacement can yield carbon offset opportunities, while at the same time providing needed energy. Nations such as the U.S. may benefit from exporting such technology. Agricultural policy, specifically food crop subsidies and land requirements, will be key factors in determining the prospects for increased biomass energy production, both domestically and abroad.

**Electric Utilities and Forest Management**

The electric utility industry has a long history of involvement with traditional forest management and tree-planting programs, through preserving forest lands for both recreational use and wildlife habitat, tree maintenance around power lines, education of homeowners on tree placement around power lines, and commercial forestry on utility-owned lands. In association with events such as Earth Day and Arbor Day, many utilities supply seedlings for employees, children and others to plant. The utility industry owns a large amount of land in order to house and surround its current and future generation, transmission and distribution facilities.

Utilities have also recently initiated numerous forestry projects specifically to conserve energy and to offset $CO_2$ emissions (Kinsman and Trexler, 1993 and 1995). A dozen or more utility companies are involved in urban forestry energy conservation programs such as American Forests' Global ReLeaf and the DOE/American Forests' Cool Communities. A few utility companies, such as the New England Electric System and PacifiCorp, have initiated forestry efforts targeted at managing carbon. The Southern Company is sponsoring research by the Smithsonian Tropical Research Institute to investigate carbon sequestration rates, the long-term benefits of standing tropical forests, and the role of rain forests in tropical economies. In addition, some utilities are using biomass as a fuel to produce electricity.

In early 1995, many electric utilities entered into voluntary accords under the Climate Challenge. Numerous of these voluntary commitments included forestry activities. For example, the American Electric Power Company and Detroit Edison Company committed to plant 15 million and 10 million trees, respectively, by the year 2000.

Some specific reasons for utilities to participate in forest carbon management include:

- There is a large technical potential for forest carbon management -- a project can offset millions of tons of carbon emissions.
  - Forestry options to manage carbon are cost effective in many cases -- e.g., a few dollars per ton of carbon offset. Forest carbon management opportunities can be among the most economical ways to address $CO_2$ emissions (Sedjo et al., 1995).
- Forestry carbon management adds flexibility, thus expanding the utility repertoire of options.
- Experience leads to improved future projects.
- Forestry projects yield positive public relations -- using forestry to manage $CO_2$ is well received by the public and environmental groups.
- Forestry efforts have positive secondary environmental and social benefits -- e.g., restoration of degraded lands and protection of biodiversity.
- International projects will help to demonstrate the effectiveness of joint implementation activities with other nations, which is a critical tool for economically addressing greenhouse gas issues.

## Utility Forest Carbon Management Program

The Utility Forest Carbon Management Program (UFCMP) is an initiative developed by the Edison Electric Institute for the Climate Challenge. The program is designed to expand utility industry efforts to manage $CO_2$ via forestry projects, both domestic and international. The UFCMP is a key component of making the Climate Challenge a success, which leads to 1) a greater likelihood of a reasonable Federal program for dealing with greenhouse gas emissions, and 2) proof that a voluntary approach to environmental protection can work. In addition, joint sponsorship of projects by many of the 55 participating utilities also means that risk is shared.

The goals of the program are to:

- Advance the state of knowledge regarding options for managing greenhouse gases via forestry.
- Establish low-cost forestry options to manage greenhouse gases.
- Implement projects to manage greenhouse gases.
- Promote environmental stewardship by the utility industry, including helping to demonstrate that a voluntary approach to environmental protection can work.

The UFCMP developed criteria and a process to review proposed projects and, subsequently, a request for proposals was issued to hundreds of individuals and organizations in February 1995. Thirty-two proposals were received in March 1995 and reviewed by the UFCMP committees, an outside consultant and, to a limited extent, by the UFCMP Advisory Council (representatives from nine non-utility organizations -- American Forests, Resources for the Future, Trees Forever, Society of American Foresters, Smithsonian Tropical Research Institute, U.S. Country Studies Program, U.S. Department of Energy, Oak Ridge National Laboratory, and USDA Forest Service). Proposed projects were located in the U.S., Central America, South America and Asia.

Technical criteria address project greenhouse gas calculations, monitoring, contingency plans, and non-greenhouse gas impacts, as well as project developer qualifications and experience. The full life cycle of project greenhouse gas emissions and emission reductions must be specified, addressing "leakage" and the fate of harvested biomass. The cost-effectiveness of the project in terms of $ per ton carbon managed is a key project characteristic.

After technical review was completed, projects were ranked and eventually a "pool" of six projects emerged as the final product for which utility sponsorship would be sought. Subsequently, UFCMP members and non-member utilities were asked to provide cooperative funding. A new non-profit corporation called the UtiliTree Carbon Company was established by over 40 utilities to sponsor the projects. The projects in the final pool represent a diverse mix of rural tree planting, forest preservation, forest management, urban tree planting and research efforts at both domestic (eastern and western U.S.) and international sites.

Total funding will exceed $2 million and carbon will be managed at a cost of under $5 per ton, including administrative expenses. Participants will share on a pro rata basis the reporting of carbon reductions into the voluntary Energy Policy Act section 1605(b) data base. In addition, a data base will be developed to report the greenhouse gas benefits of all utility forest management projects producing greenhouse gas benefits.

## Combating Desertification Via Use of Carbon Offsets

There may be a potential economic link between combating desertification and the amelioration of global warming/climate change by enhancing biotic carbon storage (Executive Summary, this volume). In other words, dryland management activities could gain funding if they can deliver carbon offsets. Squires and Glenn (this volume) assert that large fossil fuel producers and users may create a multi-billion dollar annual demand for biotic carbon offsets.

There are many reasons why a large market is unlikely to develop and why whatever market does develop will do so slowly. Thus, carbon offset projects should not be expected to supply a substantial portion of the UNEP estimate of over ten billions of dollars U.S. required for desertification control each year (Trexler and Broekhoff, this volume).

At present, it is clear that large $CO_2$ emitters such as electric utilities will not be investing large sums of money (billions of dollars) in greenhouse gas mitigation and controls:

- In the U.S., the electric utility industry is undertaking significant voluntary actions, more than any industry in any nation (see Section 3), and is hoping that these voluntary actions will suffice. Similarly, utilities in other nations such as Canada and The Netherlands are undertaking voluntary actions.

- o Large $CO_2$ emitters will not rush to implement greenhouse gas management, as the economics support the concept of "go slow;" the less the premature turnover of equipment and processes, the more economically efficient will be the response.
- o At least in the U.S. as of early 1996, there is no regulation or pending legislation that would force emitters to undertake any specific activities to manage greenhouse gases. The Climate Change Action Plan relies heavily on voluntary initiatives to meet the aim of the Framework Convention on Climate Change in the year 2000. In addition, the current U.S. Congress is unlikely to impose severe regulations on U.S. industry that would adversely affect national economic growth and international economic competitiveness.
- o There is at present no resolution of the issues related to joint implementation of projects. The concept of joint implementation, which involves partnerships between parties in different nations to manage greenhouse gases, will be a key factor in determining whether $CO_2$ emitters from developed nations become interested in developing nation dryland management for carbon offsets. The debate over "credits" and criteria, as well as the pilot phase of joint implementation agreed to at COP-1 in Berlin, will be critical. Some developing nations are quite wary of joint implementation, but others see such projects as a vehicle for providing numerous environmental benefits, including greenhouse gas management, as well as a source of money for development. Since participants from the developed nations will often be private businesses and industries, not governments, the money spent on joint implementation projects usually will be in addition to the direct governmental aid already received. It should be noted that, in general, U.S. electric utilities might be more interested in supporting joint implementation projects in countries that have expanding markets for electricity, where companies affiliated with a utility might develop energy projects.
- o The U.S. electric utility industry is entering an era of open competition from a typical condition of state-regulated, regional monopolies. Companies are carefully considering all costs to enhance their competitiveness within their regions of the country. Thus, utilities must balance current voluntary environmental actions against the need to keep costs low.
- o While deep cuts in U.S. electric utility $CO_2$ emissions could lead to annual costs to the industry in the billions of dollars, dryland carbon offsets would likely only be employed for a small portion of the $CO_2$ management effort.
    - If required to manage greenhouse gases, electric utilities would first prefer to take all cost-effective actions related to their own generation, transmission and distribution of electricity.
    - Next they would consider actions that are quite inexpensive (in terms of dollars per ton of $CO_2$ managed), with actions within their service territories preferred. The activities would include management of greenhouse gases other than $CO_2$, such as methane and nitrous oxide.
    - At present, the costs of dryland carbon offset projects are less well-known that costs of forest carbon management projects. In addition, some indicators of reliability suggest that dryland carbon offset projects might carry greater risks. Therefore, forest

carbon offset projects at present might be considered more attractive.

Change is certain to occur, however. The political situation with regard to how the U.S. meets its commitments under the Framework Convention on Climate Change may change and, over the next several years, the international deliberations over post-2000 greenhouse gas management and joint implementation may be resolved.

The outcome of the so-called Berlin mandate -- the process initiated at COP-1 to address future actions in 2005, 2010 and 2020 -- will culminate with a decision at COP-3 in 1997/98 and this will be a major driver of the future of dryland management for carbon offsets. If a more ambitious aim for post-2000 greenhouse gas reductions were to result, developed nations such as the U.S. may impose more demanding requirements on $CO_2$ emitters.

The link of joint implementation to stricter post-2000 emission management targets or requirements will be a key factor in the Berlin mandate discussions. Developed nations might not agree to stricter targets/requirements unless joint implementation is deemed an acceptable option for managing greenhouse gases. Joint implementation is so highly-valued by developed nation industries because the cost to control greenhouse gases is generally lower, or much lower, in developing nations. Energy efficiency gains come easier and forestry projects are less expensive in developing nations.

The current prospects for dryland carbon offset projects is not necessarily dim, however:

- Table 1 in the Executive Summary of this volume notes that some dryland management options might be able to manage carbon at a cost of $5 per ton C or less. Many forestry options for managing carbon are also in this range.
- U.S. electric utilities and others are evaluating offset projects now. Programs such as the Utility Forest Carbon Management Program/UtiliTree Carbon Company and the U.S. Federal government's U.S. Initiative on Joint Implementation are supportive, but for now the supply of offset projects greatly exceeds the demand for them.
- $CO_2$ emitters might be interested in obtaining the carbon offsets from dryland management projects if the $CO_2$ emitter would only be responsible for a portion of the total cost of the project. This might make the cost-effectiveness (in terms of $/ton C managed) attractive.

  It is, however, anticipated that at present relatively few dryland management projects would be attractive enough to go forward based only upon their greenhouse gas management laurels, without consideration of the additional benefits and perhaps the need for sponsors interested in other aspects of the project.

It is likely that several years will pass before there could be much money flowing into dryland carbon offset projects. It will be several years, at a minimum, before post-2000 greenhouse gas management targets or requirements are adopted at the international level, which would be followed by several years of national development of policies to meet the international targets/requirements. Then actions would be phased in over time. If ever an international market-based system were developed (with allocation of emission rights, banking and transfers of "allowances" such as for sulfur dioxide under the U.S. Clean Air Act Amendments of 1990), such a system would take many years to develop due to the tremendous complexity of establishing the "currency" of emissions via monitoring or calculations, combined with the tremendous numbers of different greenhouse gas emission sources, in different nations.

## Summary and Conclusions

Electric utilities are interested in all technically and economically feasible alternatives for managing greenhouse gases emissions. Utilities are being pro-active, through the Climate Challenge and other programs, to ensure operational flexibility to achieve greenhouse gas reductions using the most cost-effective methods. As of late 1995, DOE reported that, through over one hundred signed accords, close to 600 utilities have committed to substantial actions, without which emissions from these utilities in the year 2000 would be 47 MMTCE higher than in 1990. Energy Secretary Hazel O'Leary has termed the Climate Challenge a "win/win/win" situation for the environment, energy and the economy. The Climate Challenge and the industry-wide initiatives demonstrate that government and industry can work together, on a voluntary basis, to develop and implement cooperative approaches to public policy issues.

The utility industry has unique contributions to make in greenhouse gas management, possessing special competence in providing cost-effective customer service and in achieving environmental excellence through technical innovation, such as energy-efficient electrotechnologies; supply-side efficiencies through clean coal technologies, nuclear energy, natural gas, and renewable energy technologies; and demand-side management.

The technical potential for forest carbon management is great, able to counteract a meaningful portion of the 3 Pg carbon annual addition to the atmosphere. In addition, vigorous efforts to control land degradation in these areas could result in a net sequestration of up to one Pg carbon per year. Carbon offsets, properly documented and monitored, should be a major component of any such program to respond to greenhouse gas concerns.

Utilities have also recently initiated numerous forestry projects specifically to conserve energy and to offset $CO_2$ emissions. Fifty-five U.S. utilities are supporting the Utility Forest Carbon Management Program to expand utility industry efforts to manage $CO_2$ via forestry projects, both domestic and international. After reviewing 32 proposals, utilities have contributed more than $2 million toward a half dozen projects, which will manage carbon at a cost of under $5 per ton carbon (including administrative expenses).

There may be a potential economic link between combatting desertification and the amelioration of global warming/climate change by enhancing biotic carbon storage. However, there are many reasons why a <u>large</u> market (billions of dollars annually) is unlikely to develop anytime soon and why whatever market does develop will do so slowly. Thus, carbon offset projects should not be expected to supply a substantial portion of the tens of billions of dollars required for desertification control each year.

For there to be significant flow of dollars from $CO_2$ emitters to developing nation dryland carbon offset projects, there would have to be a strong international mandate for emissions controls by developed nations, which was uniformly accepted by all of such nations with credible plans for implementation of mitigation actions; joint implementation must be accepted and reasonable criteria must be developed for implementing projects and realizing carbon offsets; uncertainties regarding the quality and costs of the projects must be reduced; and costs to $CO_2$ emitters must be competitive with other options.

Acknowledgments

The authors appreciate the outstanding advice and support of the Utility Forest Carbon Management Program's Steering, Policy and technical committees. Steering Committee members and technical committee chairs deserve special recognition: Richard Chastain, Louis Coakley, William Edmonds, Jerry Golden, Lee Ann Kozak, Eric Kuhn, C.V. Mathai, Dennis O'Regan, Mary O'Toole and Tom Sullivan. Thanks also go to the program's Advisory Council for advice and leadership. Comments on a draft of this manuscript from Robert Beck, William Fang, Dale Heydlauff and Vic Squires are appreciated. Any errors or misrepresentations, however, are the sole responsibility of the authors.

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# European policy, land degradation and carbon sequestration in the Old World Mediterranean.

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Synopsis
This paper considers the following questions (i) What is the nature of the land degradation problem in southern Europe; (ii) what is the likely impact of global warming on the problem and (ii) how should it be approached through national and Community action ? The potential role for re-vegetation of degraded lands in the region is reviewed and discussed. Several issues are examined in more detail (i) the interaction between vegetation cover, land degradation and water resources to demonstrate that the control of the latter depends on the former: (ii) the current potential productivity of the vegetation cover to ask how far it can satisfy the needs of water and soil conservation and carbon sequestration; (iii) the likely Mediterranean scenarios for continued climate change and its impact on the plant community; (iv) the prospects for reclamation or even restoration of the plant cover and (v) the national, Community and international political dimensions of a reclamation programme.

Key Points
1. The highly-varied landscapes of the Old World Mediterranean have been subject to degradation, instability and change for at least 4000 years. Under pressures from growth, development and European Community and national agricultural policy, agriculture has intensified over the past forty years. At this moment, however, changes in global economic policy are leading to a shift to extensification and land abandonment. At the same time global warming is threatening to create more difficulties for agriculture and water resources in the region and there is international recognition of a state of desertification in southern Europe.
2. Most sites within the region have recorded significant reductions in rainfall in the past 30 years, especially since the mid-1960s. The models , field experiments and observations for semi-natural plant covers bear out the extremely strong reliance on rainfall and the special effects of drought on production of carbon. They also indicate strong spatial variations that are water-related. Restoration of the drylands within the region to their original pristine start is neither desirable nor practicable. The main constraints in the Mediterranean on land reclamation are water availability, grazing, lack of incentives and fire hazard..
3. The European Community has established a series of research programmes to understand better these changes and some results relevant to the land degradation and carbon sequestration from the MEDALUS programme are presented here. They show that (i) expected changes in temperature and precipitation are likely to affect semi-natural vegetation significantly; (ii) that land degradation has important implications for water resources, which is where the contemporary political, economic and environmental crisis lays, and (iii) that both set-aside policy and restoration and rehabilitation are logical courses of action that could bring carbon sequestration benefits in their wake.

Keywords: carbon sequestration, climate change, Mediterranean, European Union, land degradation.

*"Current research programmes pay little attention to the complex feedback mechanisms of how humans affect the environment, and how people are affected by and respond to global environmental change"* National Research Council Committee (Eos, 72(53) 1991 p.594)

## Background

The Old World Mediterranean basin comprises the land around the Mediterranean Sea that experiences Mediterranean climates. In the Koppen climate classification this is defined as one with a winter rainfall at least three times the summer rainfall. Strictly, the rainfall in the wettest month should be three times that in the driest month (Koppen, 1936). Palutikof, Goodess and Xhao (1995) have examined shifts of the northern boundary and find it may vary as much as 3-4° being northernmost in 1961-70 in the west and southernmost in 1971-80. A more stable index is the northern limit of the olive (in Europe) and of the date palm (in the Mahgreb). This paper is only concerned with the Mediterranean of the European Community i.e. Portugal, Spain, southern France, Italy and Greece, including the islands of Corsica and Sardinia.

This regions is one of interspersed mountains and plainlands, of complex geology and lithology and, within the overall Mediterranean climate, of strong variations in local climate, especially as controlled by relief. It is however well characterised by cool wet winters and hot dry summers and high inter-annual and inter-decadal fluctuations in rainfall. Here we are concerned essentially with the dry lands, i.e. areas with annual rainfall less than 600mm and annual potential evapotranspiration in excess of 1000mm. This includes most of the interior of southern Portugal, the Levant of Spain and interior basins such as the lower Ebro and La Mancha, most of coastal Italy south of Rome and the islands and a large part of Greece. In this region there is practically no natural vegetation and the last 4000 years has seen a complex succession of perturbed vegetation covers, most recently dominated by either *Quercus ilex* or *Pinus halepensis* woodland, *matorral* with bush vegetation such as *Juniperus sp., Arbutus sp.* and, much more commonly, degraded matorral (macchia, phrygana) with sclerophyllous species including *Pistacia lentiscus, Rosemarinus officinalis, Anthyllis cytisoides* and, as an introduced perennial grass, *Stipa tenacissima.*

As a result of a marginal climate, with high intensity rainfalls in autumn on parched soils, low soil moisture content, intensive grazing, repeated woodland clearance and a complex social evolution over long periods of time, many of the areas within the 'dry' Mediterranean have experienced severe erosion in the past, so that soils are thin, vegetation is poor and rapid runoff from intense storms leads to flooding. This simple statement, however, hides a complicated history in which erosion, which certainly was in existence several thousand years ago in Spain (Thornes and Gilman, 1985), has waxed and waned over the centuries. For example the policy of freedom from mortmain that was in effect in Spain over the past century led to the auctioning of 5.5 M ha of public forest and 2M ha of church lands most of which were felled by the buyers (Carrera et al., 1982) and almost certainly led to severe erosion (Thornes, 1965). This reflects climatic fluctuations as well as social history. There is also a high spatial variability in erosional intensity, with some areas having large amounts of erosion and others having very little. This is the 'erosional paradox' of the Mediterranean (Gilman and Thornes, 1983). The notion of several centuries of a relatively steady state resilient plant cover in some kind of dynamic equilibrium with a low density rural population, the stability by perturbation of Naveh (1982) does not bear detailed scrutiny, at least in the western Mediterranean. International trade was already established for minerals by Roman times, for wheat by the 12th century and for timber for boat building in the Middle Ages and there has been a constant struggle between environment and humanity since at least the Reconquest.

The pace of change has accelerated without doubt over the last forty years. The intensification of agriculture that commenced in the early 'sixties continued unabated until the late 'eighties. This resulted from application of fertilisers, mechanisation, introduction of new varieties and above all the expansion of irrigated agriculture (Grenon and Batisse, 1989). The nationally - led scramble of essentially agricultural countries to participate in the global economy was fuelled and sustained in Mediterranean Community members, as it was elsewhere, by heavy subsidies for agriculture under the EC programme of agricultural self sufficiency (Adger and Brown, 1994) and the result was an over-extension into marginal lands, excessive production of unwanted agricultural goods (especially cereals) in the Community and instability in the world markets as the subsidised products were dumped at unrealistically low prices.

At the same time there was a continuation and acceleration of the rural exodus as urbanisation took hold and as the move to the 'north' took place to cities such as Barcelona and Milan, or overseas as 'gastarbeiten' to Germany and as economic migrants to North America. There has always been a tradition of rural out-migration in hard times, but the pace was particularly strong in the years leading to the late 'eighties before return migration began to reverse the situation (Leontidou, 1996).

Co-incidentally a number of other important developments were taking place. First the environmental movement took on political strength in Mediterranean countries, though at a rate slower than in the rest of Europe. Secondly the threat of global warming led to a heightened awareness of their climatic marginality. Third a series of major droughts across the Mediterranean, especially of the early 'eighties in the western sector and in the 'nineties throughout the whole basin, heightened awareness of the critical condition of water resources. Fourth, a number of major floods in Mediterranean France, Spain and Italy drew attention to the inherited impacts of land degradation. Spain was already in the lead in trying to tackle the problems of land degradation.

Forest conservation was developed in the middle of the last century with the creation of the Forest Corps and in 1933 by the Patrimonio Forestal del Estado to be succeeded by ICONA (National Institute for Nature Conservation). The National Plan of Reforestation, the National Hydrological Plan and especially LUCDEME (Fight Against Desertification in South East Spain), created in the wake of the 1977 Nairobi Conference, all indicated a strong awareness of the national government to the land degradation problem in the south-east especially (Carrera et al. 1982).

In light of all these developments the European Community established at the Mytelene Conference in 1984 (Fantechi and Margaris, 1986) the need for a programme of research into Mediterranean Desertification. This commenced with the EPOCH programme in 1989 and continued into the Environment and Climate programme of the Third and Fourth Frameworks. The European Parliament also recognised the special case of Desertification in Southern Europe and the Fourth Annexe of the UNEP Convention on Desertification addressed the issue in the region, calling for special action at the regional (i.e. Community) level. This is a recognition that the problem here is different from that in Africa and India, that the resources to deal with it are available within the region and that the regional diversity calls for local rather than central action in planning mitigation. At the same time the potential shift of interest and resources to eastern Europe has highlighted the special position of the southern European states in relation to the Cohesion Fund and the need to ensure sustained subsidisation of the Mediterranean's special needs in respect to agriculture (Thornes, 1995).

In the early 'nineties the international community pressed, in the General Agreement on Tariff and Trade (GATT), for a relaxation of agricultural subsidies world wide and this has been approved by the Community. As a result there has been and continues to be a progressive withdrawal of subsidies that is likely to level out in the next few years. Coupled with this, under the MacSharry proposals there is an effort to accompany the reduction in subsidies with specific environmental measures. Although voluntary set-aside (see below) pre-dates GATT it was given teeth in the 1992 agricultural reforms. Whilst its objective is primarily to reduce over-production, and although it is only partially effective in this, it coincides with the abandonment of land in southern Europe and is in a direction that should reduce the offsite impacts of further land degradation.

## Objectives of this paper

Given the history described above, there is evidently a need to consider the following questions: (i) What is the nature of the land degradation problem in southern Europe; (ii) what is the likely impact of global warming on the problem and (ii) how should it be approached through national and Community action ? By co-incidence the recognition of the problem, the changes in agricultural, and the opportunity to offset global warming by carbon sequestration seem to converge on re-vegetation as the obvious mechanism to achieve this.

Whilst agreeing with Brown and Adger (1994) that land use alone cannot provide the sole mechanism for offsetting the carbon releases, even of the low energy consumption nations of southern Europe, there is the chance here of a double or even triple benefit. Land is being abandoned, land is being degraded, subsidies are available for environmental action and there is a call for proper management of these abandoned lands to prevent the fire hazard from increasing. Carbon sequestration coupled with erosion mitigation, associated water conservation and flood reduction seems to offer a possible solution.

This paper reviews the evidence available and discusses the extent to which revegetation is likely to meet some of the demand for a solution. Here I examine the following issues: (i) the interaction between vegetation cover , land degradation and water resources to demonstrate that the control of the latter depends on the former: (ii) the current potential productivity of the vegetation cover to ask how far it can satisfy the needs of water and soil conservation and carbon sequestration; (iii) the likely Mediterranean scenarios for continued climate change and its impact on the plant community; (iv) the prospects for reclamation or even restoration of the plant cover and (v) the national, Community and international political dimensions of a reclamation programme.

## The role of the vegetation cover

It is axiomatic among geomorphologists that erosion and runoff are determined by rainfall amount and intensity and plant cover. Repeatedly it has been shown experimentally, empirically in the field and theoretically that in practically all environments erosion as measured on bare ground decreases exponentially with vegetation cover, so that if 30% cover is reached, erosion has been reduced by some 70%, and by 70% cover it is practically zero (Elwell and Stocking, 1976; Francis and Thornes, 1990). It follows that revegetation is a necessary but not sufficient condition for the reduction of both runoff (and consequently flood hazard reduction) and sediment yield (and consequently off-site damage reduction including reservoir siltation).

In the MEDALUS programme we have generated two models, the MEDALUS Hillslope Model and the MEDRUSH catchment model, that deal with the strong interactions between rainfall, runoff, erosion and sediment yield. In both models climate drives rainfall and these in turn determine, together with vegetation, the partition between percolation and overland flow. The former drives plant growth together with atmospheric energy, and the latter hillslope erosion. While the MEDALUS model (Kirkby et al 1996) operates at a variable interval less than 1 hour, the catchment model typically requires daily or even monthly rainfall, from which storm events are simulated using a weather generator.

While the MEDALUS model operates for a single hillslope, MEDRUSH operates on a strip which represents the ensemble of all hillslopes in a small catchment and these are integrated up to give runoff and sediment yield for large catchments, with areas of 5000 $km^2$.

The results for the hillslope model (Thornes et al., in press) show that the decline of erosion with increased vegetation cover is well replicated, that experimental plot results can are well simulated by the model and that the reaction to reduced rainfall depends on how those reductions occur. With a reduction in the total amounts by loss of smaller rains, very little reduction of soil loss results and soil loss itself may even decrease. This is because most of the erosion and runoff occurs in one or two events per year and because ultimately there is insufficient rainfall to create erosion. These results have also been replicated in field experiments at the MEDALUS field sites.

The results from the model and for field experiments also show a significant impact of erosion on available organic matter and hence on plant growth as predicted by Thornes (1990). For example there is a strong depletion of organic matter with erosion as indicated by the enrichment of sediments with organic matter demonstrated at the Spata site (Kosmas, 1995) and indicated by runs of the model. At Spata enrichment ratios of 1.16-1.6 have been observed for organic matter.

All field sites report strong controls of land use on erosion. At Spata (Kosmas, Yassoglou, Moustakas et al., 1995) for example the figure for a single large event are as follows: bare soil 3445 kg $ha^{-1}$, bare stony soils 2344, annual herbs 904, vines with weed control 4638 and olives with understorey of turf 1.6 kg $ha^{-1}$. At Vale Formoso, Alentejo, Portugal (Roxo et al, 1995) for a number of plots over a variety of storms the proportion of erosion under different land uses was as follows: ploughed land 70.5%, wheat 12.1%, bare soil 11.7%, stubble 4.9%, Cistus 0.4% and natural grass pasture 0.3%.

Finally I note that forest does no better than matorral in stopping erosion. In rainfall simulation experiments Francis and Thornes (1990) showed that for a given cover the much higher biomass of *Pinus halepensis* woodland gave no more protection than a matorral of *Anthyllis cytisoides* with a biomass of about 3 Kg $m^{-2}$.

The impact of vegetation cover on soil moisture content is rather more complex. At the regional level, the Segura basin, one of the MEDALUS target areas in south-east Spain, has 58% of its losses in evaporation and evapotranspiration, making the soil and plant surface the most active and therefore critical part of the hydrological budget. In general in dry Mediterranean conditions plants use all available soil moisture in the growing seasons of March and April. At other times of year, and particularly in winter, recharge to the soil and ultimately to groundwater can occur. The role of the plant cover is then governed by re-evaporation of intercepted plant water and plant water consumption as indicated in the following section. The MEDALUS modelling studies and field work all underline the significance of high organic matter for infiltration and hence water retention.

What is generally absent is evidence of systematic amounts of through flow and hence a general catena controlled structure to plant-water relations. This observation is important not only for land degradation control but also for the issue of land reclamation for carbon sequestration.

## Plant biomass and productivity

### *Field observations at MEDALUS sites*

The significance of plants for both degradation and mitigation depends on the effectiveness with which cover and biomass, respectively, can be produced. There have been a number of studies on productivity in Mediterranean ecosystems, based on experimental plot studies, vegetation inventories, computer modelling and agricultural yield studies. Our experience is that there is not much data relating to the Old World Mediterranean and what there is still relatively incoherent. We should not be too surprised by this.

First there was until recently, with the exception of Montpellier and the CNRS, a strong tradition in forestry, plant phytosociology and soil studies, but a lower interest in plant eco-physiology and plant ecology. Second there is an enormous variety of habitats and a long history of change. We do not find here the homogeneity said to exist in other Mediterranean environments. Third, here above all, the role of human intervention, on the scale of months and years as well as centuries, means that even if there is such a thing as a steady state Mediterranean ecosystem, it has rarely the opportunity to be even approached let alone attained and may be largely irrelevant to debate on mitigation. Finally the water regime, which dominates growth in the regions of interest is a function of a whole suite of factors and not just climate.

Summarising the picture in 1981, H.A. Mooney showed that the three main types of cover, evergreen forest, evergreen scrub and subligneus communities in Mediterranean- type ecosystems had biomasses of about 33kg.$m^{-2}$ for forest and 3-4kg $m^{-2}$ for scrub and degraded matorral. Productivities were found to be between 100 and 200g $m^{-2}$ $yr^{-1}$ Francis and Thornes (1990) found similar values at sites the Province of Murcia, Spain where rainfall amounts to an average of 300mm $yr^{-1}$ and biomass for a uniform cover *of Anthyllis cytisoides* at 3kg $m^{-2}$. In this environment the strong seasonality of growth is well known and ecophysiological studies at Rambla Honda field site Almeria (Incoll, Puigdefabrigas, 1995) . and the confirm that this is the case.

However the studies have shown strong inter-annual variations in photosynthetic rates and therefore growth according to rainfall and strong inter-specific variations according to water use capacity. For example 1994 in south-east Spain was a particularly dry year and the dry season was especially prolonged and severe. Under these conditions *Stipa tenacissima* and *Anthyllis cytisoides* had virtually no flowers, whereas *Retama sphaerocarpa* , which was shown to obtain water from greater than 16m, flowered profusely. Even in retama, leaf shedding was exceptionally strong in 1994. Artificial watering (Incoll, 1995) also significantly enhanced photosynthetic rate. so that watered plants had photosynthetic rates in excess of 14 µmol m-2s-1 whereas in unwatered plants 2.0 µmol m-2s-1 was observed.

The observations reveal a lag in the impact of extreme drought or excess due to the mortality of seedlings, which is important in understanding the impact of past extreme events (Haase, Pugnaire and Incoll, in press). There is also a strong contrast between Anthyllis and Stipa in survival capacity under drought, the former showing significantly more death of above ground parts.

In addition to the total biomass, productivity and the variations in time, the spatial pattern of production is also important. Although the role of plants for erosion mitigation generally assumes an exponential decrease with cover, it has been shown that with island -type vegetation separated by bare soil, rates of runoff and erosion among bush islands with a higher overall biomass and cover can be higher than species with a uniform cover at lower biomasses (Abrahams and Parsons 1991). The mosaics of vegetated and bare patches at the Rambla Honda site are most strongly related to *Stipa tenacissima* and there is clear evidence that the plants are taking advantage of the runoff generated in the bare areas and that the spatial distribution and performance of annuals is influenced by that of perennials. (Puigdefabrigas, 1995).

Water exclusion experiments of both surface and subsurface water had significant effects, showing the importance of spatial cellular patterning. This is carried over to soil organic matter where the spatial contrast in vegetated patches and bare areas is strong. In Retama the contrast in organic matter between bush and bare is 4.7 and 1.3% respectively, in Anthyllis 4.6 and 1.7 and in Stipa 6.8 and 4.6. This is reflected in the hydraulic conductivity of the soils measured by the Almeria team. For bush and bare the values are (K, m/day) Retama 1.9 and 1.0, Anthyllis 0.7 and 0.3 and for Stipa, 0.8 and 0.3. All of these differences disappear below a few centimetres depth.

These controls all operate on sites affected by fire (Specht, 1981)and abandonment from agriculture. The latter is particularly important since large areas are being abandoned or left to long fallow both through regional economic change and through the impact of Community policy. Obando (1995) has shown a systematic logistic type increase in biomass on abandoned plots in the Guadalentin target area, with full cover being reached at about 12 years and 30% cover being reached in as little as 4-5 years.

There are accompanying changes in soil organic matter and diversity as revealed from the El Ardal Experimental Site by the Murcia team (Lopez-Bermudez et al., 1995). Root biomass stabilises after about 15 years at 2.5 $Kgm^{-2}$ and diversity after about 6 years, falling after about 30 years. The soil organic matter under *Rosemarinus officinalis* growing on abandoned land increases from very low values after agriculture to values comparable to the surrounding matorral (typically 3-4%) after about 20 years after which it continues to increase beyond values found in old scrubland (Martinez-Fernandez et al, 1994).

*Modelling temporal and spatial variations*

The spatial variation between surface runoff- fed (Stipa) and deep soil water-fed (Anthyllis) have been modelled by Thornes(1990). Here the species were regarded as r- and K- species, with the opportunistic r-species being able to mobilise runoff form individual events. These were allowed to compete on the basis of annual variations in the soil moisture and overland flow and the k-species typically show logistic response curves, squeezing out the r-species in runs of wet years, but collapsing in a non-linear response to prolonged low rainfall periods, such as the 'forties and the 'eighties.

Under these conditions it was found that individual deep drought years are enough to carry the k-species across a threshold. This modelling strategy and results seems to be born out by the field observations reported above and suggest that soil moisture and runoff provide the main key to plant productivity in areas of rainfall as low as this (300-400mm).

The Sheffield MEDALUS group (Woodward, Diamond and Sheehy, 1995) have recast the plant growth component of the MEDALUS hillslope model for MEDRUSH. This simplifies the relationships and provides a better coupling of the climate and plant components, because the layered soil model and a low losses from deeper soils in the hillslope model allowed too much root growth (Thornes et al, in press). The generation of assimilate is based on a Monteithian growth analysis (Russell, Jarvis and Monteith, 1989), partitioning according to phenology and respiration temperature controlled.

For evergreen shrub, Almeria typically produces 1200 g/m$^{-2}$, Athens 1400, Madrid 1700 and Naples 2400. The corresponding figures for annual grass being 50, 390, 650 and 1100 g m$^{-2}$. These figures closely reflect the climatic gradients between the sites. A more detailed analysis of variations within Spain (Woodward and Diamond, 1996) shows results that correlate very strongly with the length of the dry season in months. In this work they also show that the impact of $CO_2$ doubling is relatively small compared with variations in temperature (and therefore soil moisture), but most felt where the driest period is longest.

The models, the field experiments and observations for semi-natural plant covers bear out the extremely strong reliance on rainfall and the special effects of drought on production of carbon. They also indicate strong spatial variations at the hillslope, valley side and regional scale that are water-related. The same is true of cultivated plants. At the Spata (Greece) site rainfall exclusion points to a direct correlation between rainfall amounts and wheat productivity ( Kosmas et al., 1995). This is again well documented for dry Mediterranean environments. All the MEDALUS observations lead us to conclude that a good first approximation to above ground for the peak foliar standing crop (which occurs in May-June), soil productivity and soil organic carbon amounts can be obtained from rainfall amounts, its distribution throughout the year and constraints on total production. This could be improved by a knowledge of carry over from previous years, as indicated by Incoll et al (1995) for the Rambla Honda site. In other word rainfall is strongly the limiting factor in these dry Mediterranean environments, and length of dry month and proportion of runoff to percolation is important in determining the spatial variability at several different scales. This appears to be true of both the annuals and perennials observed and is in accord with results from other dry area (e.g. the work of Turner and Randall, 1989, in Southern Nevada, USA).

### The prospect of global warming and its impact in the Mediterranean

All MEDALUS sites and target areas report significant reductions in rainfall in the last 30 years, especially since the mid-'sixties. One implication of this is that estimates of productivity gathered over this period may not be appropriate for future estimates, and there seems to be no option but to attempt to model the impact of rainfall on production rather than use present values. This is well illustrated by the case of the Guadalentin, where long term records for daily rainfall go back 125 years. In the period 1894-1934 there was a reduction of almost 50% in the mean annual rainfall. Models suggest that this must have had a very notable impact on productivity (Thornes and Brandt, 1993). These fluctuations have therefore to be built into any estimates of the efficacy of carbon sequestration for mitigating the impact of global warming.

Both the changes to be expected and, even more, the impacts of greenhouse gas warming are highly speculative for well known reasons. Therefore we provide only a brief insight to the work being done in MEDALUS, and only so far as it might relate to the problem of carbon sequestration. The reader is referred to the paper by Palutikof, Goodess and Guo (1994) for details of the set of seasonal scenarios for the Mediterranean basin of the perturbation in mean water availability due to the greenhouse effect. Using the forecasts provided for the UK Meteorological (Hadley Centre, 1992) model and the Canadian Climate Centre model, Palutikof, Goodess and Xhao (1995) have attempted to provide scenarios for the Guadalentin valley in south-east Spain, which is one of the Target Areas for the MEDALUS programme and which is further discussed below.

Given the specific need for seasonal rainfall inputs to plant growth models, the problem is to provide, for any region, an assessment of the impact of the Mediterranean wide effects at the global scale. Palutikof, Goodess and Xhao attempt to do this by relating the incidence of rain to regional weather types and the latter to GCM parameters other than rainfall, which are more reliable in the long term scenarios, especially sea surface pressures. The weather types are distributed seasonally and the frequency of south and south-east weather types, which bring the 'gotas frias' to the Guadalentin valley, are well reproduced by the Hadley Centre UKTR model.

These 'gotas frias' have been known for a long time to be the main causes of erosion in the south-east of Spain, because they occur on dry ground in autumn with very high intensities. In winter westerly and south westerly air flows prevail and these tend to recharge the soil moisture ready for spring plant growth. Here neither model performs very well, tending to overestimate the frequency of this weather type in both autumn and winter.

The actual rainfall is estimated from a regionally based empirically developed weather generator. This is very good at simulating the number of rain days though the standard deviation of rain days and the longest wet and dry periods are consistently underestimated. When the future changes in weather type frequency is simulated, the largest and most consistent changes are in summer, but mainly of weather types that have little significance for the parameters of interest. When the weather types of the future are translated into raindays, both models predict fewer rain days for the Guadalentin during the growing season and the period of lowest evaporation. They also predict an increase in the number of rain days during the non-growing season and the highest period of evaporation, when the surface is most vulnerable to erosion.

Further investigation by the East Anglia team reveal quite significant changes in temperature extremes. And, subject to the caveats about uncertainty, they indicate a decrease in the number of cold days and, less intuitively a very large change in the number of very hot days (an average increase of 71 days per year above 35°C. These results indicate that conditions are likely to be less rather than more conducive to carbon production in the future.

As was noted earlier, Woodward and Diamond (1996) have compared the response of shrubs and annual herbs to a 1.5°C increase in temperature and a 15% fall in precipitation. This shows, for Spain, nearly parallel changes in shrubs and grasses, a stronger impact in drier areas (percentage wise) and only a small absolute effect in areas with less than 4 dry months per year. Similar results were obtained by Kirkby and Neale (1986). It appears that it is in the truly dry environments of the Mediterranean, where the seasonal component of rainfall is strongest, where the effects of change are likely to be significant for plant production.

Finally, the modelling of Obando and Thornes (in press) of abandoned land recuperation and its impact on erosion, shows that recovery rates and character are heavily dependent on rainfall history in time and on the pattern of abandonment in space. It matters greatly to sediment yield and runoff whether abandonment is in a wet or dry spell and whether it is in the upper or lower part of a catchment, for example. This has to be considered as we develop mitigation strategies from first principles or extrapolate them from empirical results of past practices.

## The prospects for reclamation

### *The need*

Allen (1995) has separated the goals of revegetation into restoration, reclamation and rehabilitation in a North American rangeland context . For her *restoration* means reproducing the ecosystem that existed prior to disturbance, assuming the site was a relatively undisturbed, late successional or otherwise desirable native ecosystem. A more colourful analogy is that provided by Aaronson et al. (1993) who liken restoration *sensu stricto* to the restoration of a renaissance painting that has deteriorated over time, but still retains the initial lines and colours.

*Reclamation* (rehabilitation to Aaaronson et al) means land restored to a functioning ecosystem of less complexity and involving exotic species at a lower level of diversity. *Rehabilitation* (reallocation) implies that the land has been made productive again, but that an alternative ecosystem has been created, with a different structure and functioning from the original system.

As far as the Mediterranean is concerned the prospect of restoring the dry lands to their original ecosystem may be neither desirable or practicable. At the same time there is little likelihood that the mountain lands will ever be in great demand for agriculture. The clocks are unlikely to be turned back again despite the traditional use of the land as the buffer against hard times in the Mediterranean. On the other hand the heavily degraded land enhances runoff and flooding, and sediment yield and reservoir siltation and other off-site effects. It seems to follow that, in the most degraded land at least, revegetation as a form of rehabilitation or reallocation should have a high priority.

The immediate need is to ensure the conservation of water resource, to limit flooding and to prevent reservoir siltation. At the same time there is the prospect of multiple benefits. These include recreation, progressive soil improvement, a grazing nutrient bank for the future and, of course the possibility for carbon sequestration. Therefore in this context in working out the gains we have to see carbon sequestration as part of a package of benefits, in working out its potential. Moreover we have to reckon with the existence of potentially multiply stable states and thresholds of stability without calling for restoration of the original ecosystems. Modern ecosystems theory does not allow the luxury of an ultimate stable sate or goal to which the system is operating but does provide us with non-linearities that have to be known and manoeuvred to our benefit (Thornes, 1987).

*Constraints*

The main constraints in the Mediterranean on land reclamation are water availability, grazing, incentives and fire. We have shown in this paper that biomass production in the dry lands of the Mediterranean depend on water. Under unconstrained growth with adequate water (say the average of 300mm) weeds, herbs and bushes can accumulate to a 30% cover at most sites in about 3-4 years and provide a full bush cover with typically about 3 KG $m^{-2}$ in about 12 years in the driest lands of south-east Spain provided that there is some soil to settle in. This is without the application of fertilisers, water or even management. The future prospect for biomass will depend on the future prospects for rain, and these are not good.

Management is required because (a) the fire risk increases with standing dead is a well known function of time (Specht, 1981) and an increase in fire is both a perceived and real hazard throughout the Community nations of the northern Mediterranean, for a variety of complex reasons; (b) it is practically difficult to inhibit grazing now, even though the number of livestock is diminishing rapidly in the western Mediterranean and the pattern of herding is changing. The distribution of herding sheep and goats over the canadas (long transhumance routes in Spain), for example, has almost entirely died out.

The role of grazing itself is somewhat ambivalent. MEDALUS studies in Nios (Margaris et al, 1995) show that the major impact of grazing is to *increase* fire. As the pressure increases the available standing crop decreases and pastoralists set fires to encourage young growth. Others have claimed that maintaining grazing will be necessary to control the likelihood of fire by reducing the amount of standing dead. In grazing exclusion experiments in Lesbos showed an increase in cover, diversity of woody species, absolute number of plants within the exclosure and , at the Galani site, substantial reductions in erosion rates without grazing.

A budget model for the Nios case reveals that it would be far cheaper to provide feed for the animals and reduce fires, allowing grazing to continue, than to expend money either on a fire fighting service or stopping grazing and importing food from other islands (Koutsidou, 1994). These are not the only concerns of a policy that would simply abandoned land. Work by Kosmas (1995b) indicates that the increase in organic matter with age since abandonment is not necessarily associated with an increase in infiltrability. As abandonment after tillage increases there is an overall increase in bulk density and in the long-term, if bush vegetation establishes and stabilises, there will be an increase in runoff due to the reduction in annual's unable to compete. The prospects for removing the grazing constraint therefore depends on the development of alternative strategies for fire management.

The third constraint is the question of incentives. Unless financial incentives are provided the management of land will not take place. In the European Community, the set-aside programme has so far been designed to reduce agricultural production. Farmers are not allowed to claim subsidies unless 15% of land is set-aside from agricultural use. In reality farmers set-aside land of poorest quality so that the economic aims are not being satisfied, while farmers in northern Europe claim that to recover this land in the future will be very costly.

In the Mediterranean, forecasts show a more or less contiguous decline in upland agriculture for the foreseeable future (Grenon and Batisse, 1989). It follows that to be paid to abandon land is a positive incentive. If at the same time a carbon dividend can be provided, then the prospects both for carbon sequestration and land management are positive.

## Discussion and Conclusions

The following observations can be drawn from what has been said above:

(I) That the Mediterranean environment is very complex in space, has had a complex cultural history and has a rainfall that is highly variable in time. Under these circumstances sweeping statements about the potential for carbon sequestration and its advantages are to be treated with circumspection and even scepticism.

(ii) Land abandonment, long fallow, the shift from the countryside and the decrease in agricultural subsidies following the GATT agreement means that the returns from poorer quality land (that most common in the dry areas) land are falling, a trend that seems likely to be sustained, at least on the poorest land.

(iii) As a consequence land management policies, including set aside and the actions accompanying the GATT agreement outcome, are being developed to shift the land use from production to conservation.

(iv) Under these circumstances, although carbon sequestration is not the prime cause of land reversion from tillage to scrub, it could be a significant added value that could substantiate the argument for allowing this policy to continue and even in encouraging it. The carbon gains so achieved, at low or zero cost, could be used to offset carbon dioxide production from the Mediterranean countries, or could attract investment in management, fire control or recreational facilities from carbon offsets by producing countries. Even better, those offsets could be more directly used to apply specific erosion control measures.

(v) Arguments about the specific net benefits of carbon sequestration are dependent on a detailed economic, social and cultural analysis of the costs and benefits. Moreover they are very sensitive to discounting rates, areas under transformation and actual estimates of carbon captured and below ground biomass, undecomposed litter, soil organic carbon content and rates of production. Whilst the areas of Mediterranean bush and scrub are reasonably well known for Mediterranean countries, the estimates of the carbon pools are still tentative, despite the accumulated evidence from the MEDALUS project. A detailed analysis of these will be undertaken in MEDALUS III, but with a fully developed bush cover *under present conditions*, first indications are that the benefit will be significant, and of the order of 4-6 tonnes/ha in successful years.

(vi) Unfortunately the indications from MEDALUS climate change studies reported above suggest that rainfall could become less, temperatures higher and rainfall more intense, so that net primary productivity, which is closely coupled to rainfall is expected to fall relatively, marginally in the wetter areas and significantly in the dries areas. Also the indications are that the doubling of $CO_2$ will not notably increase production.
(vii) The general balance therefore seems to be that the prospects for sequestration are good from a social, political and economic point of view, even if the action is only to leave the abandoned land in scrub. However the coupling of productivity to rainfall means that the uncertainty in quantity of sequestered carbon will be high and the long term climatic prospects is for a worsening of the average productivity in the lands most in need of reclamation.
(viii) Sequestration should probably therefore be viewed as an important desirable beneficial output of contemporary desertification-mitigation policy and action rather than an ultimate goal in its own right in these environments and under current cultural, political and socio-economic and future climatic conditions. This view may need to be revised as more data becomes available, but on its own, carbon sequestration is unlikely to be a solution to the problems of land degradation in southern Europe.

## Acknowledgements

This MEDALUS programme is supported by the European Commission DGXII under the Framework III Climate and Environment Programme. I am especially indebted to my colleagues in the research project for permission to quote their work and for their continuing effort in producing high quality research. I hope I have not misquoted them here. I am also indebted to my research assistants Julie Shannon and Paola Mairota for help and advice in the preparation of the text.

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# Towards incentive-compatible mechanisms for enhanced carbon storage in the African drylands

**Brent M. Swallow**

Synopsis

This paper considers the economic prospects for reducing carbon emissions and increasing carbon storage in the habitable drylands of Africa. Particular attention is paid to the semi-arid and drier sub-humid agro-ecological zones of sub-Saharan Africa. Crop and animal agriculture are the main sources of rural employment in those areas; cropping, livestock grazing, wood collecting and foraging are the main human uses of the natural environment.

The paper begins with a conceptual model of "net carbon benefit" for the habitable drylands of sub-Saharan Africa. The model distinguishes the impacts of different types of human activities, particularly crop agriculture, livestock grazing. crop/livestock production. fuel wood harvesting and tree planting on carbon emission and storage. Several propositions are then presented about the relationships between those activities, the environment, the variable climatic conditions that affect the region, the structure of economic institutions and incentives, and the adoption of technologies and land management practices.

Key points

1. The habitable drylands of Africa cover about 12,86 million km$^2$ (about 44% of the land surface). Of these lands 5.04 million km$^2$ is arid, another 5.14 million km$^2$ is semi-arid and the remaining 2.69 million km$^2$ is dry sub-humid. These regions have a human population of about 245,500,000 , approximately 189,000,000 of whom live in sub-Saharan Africa.

2. It is estimated that Africa's human population is increasing at an annual rate of 2.9%. This growing population needs more energy for basic needs such as cooking and heating and economic activities such as agro-industrial processing. Fuelwood is the source of 80-90% of rural energy in Africa. As population densities rise people will begin to remove live trees and diminish the density of standing woody biomass. Not only is the burning of fuelwood releasing carbon to the atmosphere, it is also eliminating important carbon sinks.

3. Africa's drylands are sinks and sources of carbon. Improvements in the global balances of greenhouse gases could occur through reduced emissions of carbon dioxide or increased storage of carbon as live plant matter or soil organic matter. The habitable drylands of Africa have the potential to store large amounts of carbon and thus mitigate some of the effects of increased atmospheric carbon dioxide concentration. Grasses, trees and shrubs can store carbon in the above-ground biomass for periods of several years, more importantly, the carbon contained in plant roots can be stored in soils for extended periods of time.

4. Carbon dioxide emissions can be reduced by reducing the amount of land burnt each year. (savannah land and fallow land for crop agriculture), while carbon storage can be enhanced by greater production of plant biomass (crops, trees. grasses). and greater storage of that biomass in the soil.

5. However, restoration and regeneration of African drylands must be compatible with the interests of the individuals and groups of people who

eke out their lives in these areas. In other words, projects and programs must be incentive-compatible to be successful: they must satisfy the carbon storage objective while appealing to the needs and motivations of Africa's farmers and livestock keepers.

Key words: dryland degradation, restoration, sub-Saharan, agro-ecosystems, pastoralism, cropping. fuelwood harvesting, demography

## Introduction

The habitable drylands of Africa present interesting opportunities and compelling needs to the global community. On one hand, the drylands of Africa have the potential to store large amounts of carbon and thus mitigate some of the carbon dioxide -- an important 'greenhouse gas' -- that is emitted through the combustion of fossil fuels and changes in land-use around the globe. Grasses, shrubs and trees can store carbon in above-ground biomass for periods of several years; more importantly, the carbon contained in plant roots can be stored in soils for extended periods of time (Wisniewski et al., 1993). On the other hand, the people who earn their livelihood from use of the African drylands have urgent and compelling basic needs. Thirty percent of the people of sub-Saharan Africa are classified as extremely poor (less than $US 275 annual consumption in 1985).

Most of the statistics on land degradation and economic stagnation indicate a 'lose-lose' situation for the global community and the majority of rural Africans. Cleaver and Schreiber (1994, p.1) begin their exploration of the nexus between population, agriculture and the environment with the following statement: "Over the last thirty years, most of Sub-Saharan Africa (SSA) has experienced very rapid population growth, sluggish agricultural growth, and severe environmental degradation." Can these trends be reversed? Can increased production of deep-rooted trees and grasses be compatible with improved livelihood for rural Africans? How? Could carbon offset programmes make a significant contribution to the achievement of that 'win-win' solution? In this paper I make a modest attempt to address these daunting questions.

At the 1992 'Earth Summit' in Rio de Janeiro, 155 countries pledged to work together to reduce emissions of carbon gases (the most important being carbon dioxide and carbon monoxide) to 1990 levels by the year 2000. Representatives of over 100 countries convened in Berlin in April 1995 to re-consider that goal and explore how this might be achieved. One alternative is through international 'carbon offsets' in which polluting countries would receive credit for enhancing carbon storage in other countries. To date the few carbon offset projects that have been initiated have involved utility companies in the United States financing forestry projects in particular sites of the humid tropics (*Time*, April 3, 1995; Trexler and Meganck, 1993). The carbon benefit generated by a new plantation in Ecuador or improved logging practices at a small logging site in Malaysia can be monitored and valued assigned. But carbon offsets involving less carbon emission and more carbon storage in the African drylands would need to cover large tracts of land. Could large areas be set-aside into permanent vegetation reserves?

While some current conservation areas and national forests might be better managed for sustainable fuelwood production or carbon storage, it is unlikely that African governments would be either willing or capable of excluding people from substantial amounts of additional land no matter how badly degraded or sparsely settled. Most governments have found it difficult to exclude people from national parks and national forests. Even if people could be displaced from certain areas, it is quite possible that those same people would cause as much environmental havoc in other areas. And many people in the donor nations would find it repugnant to exclude poverty-stricken pastoralists and farmers from their customary lands in exchange for money flowing into the coffers of African governments.

One of the maintained hypotheses on which this paper rests, therefore, is that restoration and regeneration of African drylands must be compatible with the interests of the *individuals and groups* of people who eke out their lives in those areas. In other words, projects and programmes must be incentive-compatible to be successful: they must satisfy the carbon objective while appealing to the needs and motivations of Africa's farmers and livestock keepers. In this paper I attempt to identify such incentive-compatible policies and programmes, but do not attempt to attach a price tag to their implementation. An agency such as UNEP or the World Bank might consider that a subject for follow-up study.

This paper focuses on the habitable drylands of Africa. According to the UNEP (1992) *World Atlas of Desertification*, in Africa the habitable drylands cover about 12.86 million km$^2$ (44%), inhabitable hyper-dry areas cover about 6.7 million km$^2$ (23%) and more humid areas cover the remaining 10.1 million km$^2$ (34%) of the total land area of about 29.66 million km$^2$. Of the habitable drylands, 5.04 million km$^2$ is arid, another 5.14 million km$^2$ is semi-arid, and the remaining 2.69 million km$^2$ is dry sub-humid. Most of the habitable African drylands are located in three regions: 1.51 million km$^2$ in north Africa, 8.02 million km$^2$ in the Sahel (including East and West African Sahel), and 2.95 million km$^2$ in southern Africa (Table 1). An overlay of the UNEP (1992) dryland boundaries with the 1994 human population data layer developed by Deichman (1994) results in the following estimates of the number of people inhabiting the drylands as of 1994: all of Africa – 245,480,520; sub-Saharan Africa – 189,310,509 (GIS analysis contributed by Russ Kruska, ILRI, September 1995) (See Figure 1).

Table 1 Land area in Africa by aridity zone (millions of km$^2$)

| | Region | | | | |
|---|---|---|---|---|---|
| | North | Sahel | South | Others | Total |
| hyperarid | 385.4 | 276.4 | 8.2 | 0.0 | 670.0 |
| arid | 98.1 | 348.6 | 54.1 | 2.7 | 503.5 |
| semiarid | 37.4 | 303.7 | 159.4 | 13.3 | 513.8 |
| dry subhumid | 15.1 | 150.1 | 81.5 | 22.0 | 268.7 |
| humid | 9.3 | 260.0 | 127.7 | 612.6 | 1009.6 |
| total | 545.3 | 1338.8 | 430.9 | 650.6 | 2965.5 |

Source: UNEP, 1992, Table 1

## Carbon sources and sinks in the African drylands

African drylands are sources and sinks of carbon(Table 2). Improvements in the global balances of greenhouse gases could occur through reduced emissions of CO$^2$ or increased storage of carbon as live plant matter or soil organic matter. This section presents a brief discussion of the various sources and sinks with the aims of: (a) quantifying their importance, (b) distinguishing those that are subject to human manipulation from those that are not, and (c) predicting how different variables might change as a result of changes in human activities. The main implications are summarized in a number of eleven propositions. The propositions are summarized in Table 5.

Table 2: Sources and sinks of atmospheric carbon in the African drylands

| Sources of atmospheric carbon | Sinks of atmospheric carbon |
|---|---|
| Grazing and over-grazing | Perennial fodder crops and planted pastures |
| Removal of trees and consumption of fuelwood for domestic energy needs | Trees in forests and woodlots |
| Land clearing for agriculture | Agro-forestry |
| Mining soil organic matter | |
| Savanna burning | |

2.1 Source: Grazing and over-grazing

Domesticated livestock have long been targeted as a major source of Africa's environmental ills. This mainstream or conventional view is supported by two facts and three assumptions often taken as fact. The facts are that: (i) the majority of the habitable drylands of Africa is either fallow or permanent pasture, and (ii) that livestock grazing is the most visible and one of the most important uses of those areas. The three assumptions are: (i) African livestock owners are driven to maximize the numbers of animals they hold to enhance their personal prestige; (ii) the regimes that govern access and use of African rangelands fail to provide the incentives necessary for proper resource management; and (iii) the herbaceous and woody vegetation is sensitive to the intensity and manner in which it is grazed. The implication drawn from these facts and assumptions is that livestock owners, left to their own devices, will stock too many animals for short-term profit maximization or for long-term environmental sustainability. In short, the 'tragedy of the commons' is seen as inevitable (Hardin, 1968) unless livestock owners are convinced that they should sell more animals and are given individual rights to specific areas of rangeland so that they consider the full costs of their actions.

With few exceptions, however, the livestock and range management projects that were based on these assumptions were dismal failures (see section 3.2 below). Project experience and scientific evidence has since culminated in a re-appraisal of the bases of the assumptions. Research on pastoralists' behavior led to a re-assessment of assumption (i). In the most arid areas, pastoralists' behavior is necessarily dominated by the need to cope with extreme environmental and market risks. Without access to insurance or alternative income prospects, pastoralists adopt strategies to self-insure or pool risks with others (Swallow, 1994).

One self-insurance strategy is to maximize the reproductive capacity of their herds: productive females for pastoralists primarily concerned with milk and progeny; adult males for crop producers primarily concerned with animal traction (see Webb et al., 1992 for examples from Ethiopia). Detailed studies of the property institutions that govern the use of rangelands indicate show that the tragedy predicted by Hardin does occur but is far from ubiquitous. Several examples of common property regimes have shown to effectively govern the collective use of natural resources (see reviews by Swallow and Bromley, 1994, 1995).

The ecological significance of livestock owners' behavior and the regimes that govern that behavior depends upon the dynamic relationships between rangelands and livestock. Recent research on the ecology of African rangelands has challenged the general validity of assumption (iii). Research conducted at both the aggregate and field levels indicate that the productivity of arid rangelands is primarily determined by climatic conditions. Hulme and Kelly (1993) showed that 83% of the variation in areal extent of the Sahara between 1980 and 1989 was explained by variations in annual rainfall.

At the field level, research conducted in the Turkana area of Kenya (Ellis and Swift, 1988), around artesian bore-holes in Senegal (Hanan et al., 1991) and in the Gourma of Mali (Hiernaux, 1993) found that while grazing intensity reduced standing herbage biomass had very little impact on the long-term structure and productivity of herbaceous vegetation. Rather, it was found that herbage yields were almost completely dependent upon rainfall. Given the episodic and variable rainfall in those areas, rangelands and livestock populations are continually in flux. Non-equilibrial, or state-transition, models are more appropriate than equilibrial or successional models for depicting rangeland dynamics in such circumstances.

The scientific evidence supporting the non-equilibrial model was gathered at sites in arid areas: average annual rainfall is about 300-400 mm in southern Turkana and 150-350 mm in the Gourma. Coppock (1993, 1994) argues that rangelands receiving higher amounts of rainfall, such as the Borana Plateau of southern Ethiopia (400-700 mm), exhibit some elements of the equilibrial model of rangeland dynamics. First, current grazing patterns do affect the future structure of vegetation. In other words, vegetative structure is density dependent. Second, the probability of drought is relatively low. De Leeuw and Reid (1995) argue that the main difference between 'equilibrial' and 'non-equilibrial' rangelands is the type of grasses that dominate (which in turn is a function of soil moisture and type).

Non-equilibrial areas are dominated by annual species that regenerate from seed and produce large stocks of seed under severe environmental conditions. The bi-modal rainfall pattern in East Africa means that annual grasses dominate in areas receiving less than 400 mm of annual rainfall; the unimodal rainfall pattern in West Africa means that annuals dominate in areas receiving up to 800 mm of annual rainfall. The implication, therefore, is that livestock grazing patterns are not important determinants of herbaceous vegetation in those areas of East and West Africa.

In areas receiving higher amounts of rainfall where perennial species are the main source of forage, livestock grazing can have more impacts on the structure and functioning of ecosystems. Abel (1993) discusses four types of impacts of heavy grazing or browsing on herbaceous vegetation and soils: a) the removal of vegetation results in a direct increase in soil erosion through the exposure of more soil to wind and rain; b) selective grazing pressure results in a decline in the abundance of palatable perennial grasses and an increase in the abundance of less palatable grasses, weeds and bushes; c) a change in the composition of the vegetation that exposes more soil will result in an increase in soil erosion; and d) an increase in soil erosion can have negative effects on vegetative structure (although in fact there are many other impacts).

With data collected in the central region of Botswana in the 1980s, Abel (1993) used a series of models to examine those impacts. His results indicate that high stocking rates have: a) small direct impacts on soil erosion, b) significant impacts on species composition of the herbaceous vegetation; and c) very small secondary impacts on soil erosion. As long as vegetative cover remains over 40%, the rate of soil loss is relatively insensitive to percent vegetative cover.

Woody vegetation also plays central roles in the structure and function of ecosystems and the composition of livestock diets in arid areas. Trees are important sources of feed, especially for camels and small ruminants and especially during dry seasons when the quality of herbaceous forage is low. De Leeuw and Tothill (1990) report that browse comprised about 4%, 34% and 87% of the annual diets of cattle, sheep and goats in a crop-livestock system in semi-arid Mali. In south Turkana shrubs and trees comprised 79% of the wet season and 19% of the dry-season diet of goats and 98% of the wet-season and 95% of the dry-season diet of camels. Goats also relied heavily on seeds and seedpods (29%) of *Acacia spp.* during the dry season when other nitrogen-rich feed sources are scarce (Coppock et al., 1987).

Most of the trees found in the drylands are deep-rooted drought-resistant species that are less susceptible to changes in climatic conditions than annual or perennial grasses. The dynamic relationships between livestock and trees may be very complex; at very high stocking rates livestock browsing may retard tree growth and contribute to deforestation around areas of high human and livestock population density such as wells and villages. On the other hand, Reid and Ellis (1996) show that livestock and livestock management practices can play very positive functions in tree growth. They found that the consumption of *Acacia tortilis* seedpods by Turkana livestock, disposition of intact seeds in richly-manured livestock corrals, and frequent movement of corrals by the Turkana people was responsible for most of the tree cover.

> *(P1) In low rainfall areas (i.e. < 400 mm annual rainfall in East Africa and < 800 mm annual rainfall in West Africa), livestock grazing has little impact on the structure of herbaceous vegetation. Livestock grazing can have negative or positive impacts on the structure of woody vegetation in those areas.*

> *(P2) In higher rainfall areas, livestock grazing can affect the composition of herbaceous vegetation with the effect of reducing the abundance of the most palatable grasses and thus the potential to produce secondary outputs. Vegetative cover usually decreases and increased soil erosion might occur if vegetative cover reaches low levels. The woody species that replace the perennial grasses may be of more or less value for carbon storage than the grasses they replace.*

> (P3) There are complex relationships between livestock and trees in many dryland areas. Very heavy browsing can retard tree growth. Alternatively, livestock can also play important roles in mediating tree growth.

2.2 Source: Removal of trees and consumption of fuelwood for domestic energy needs

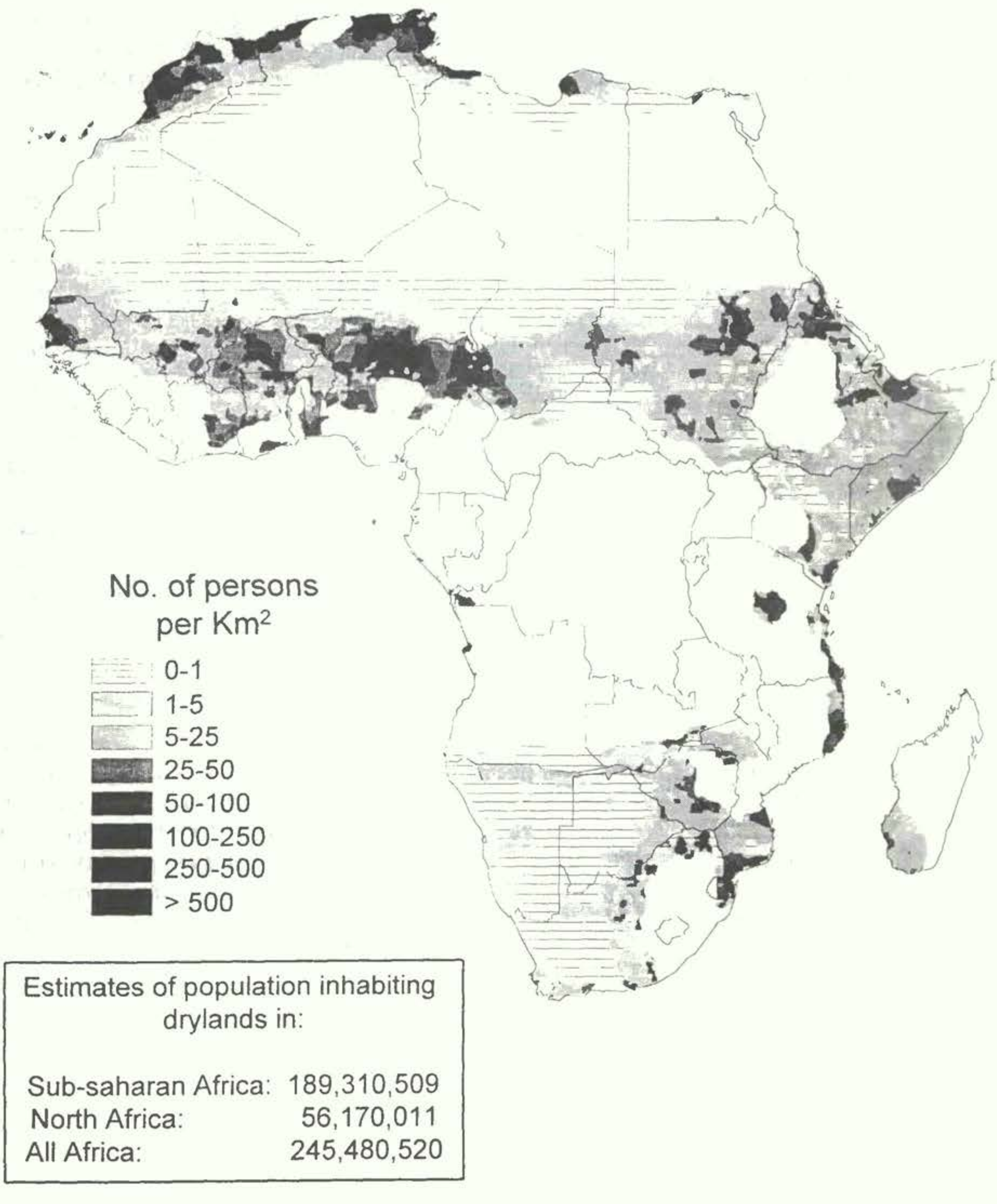
It is estimated that Africa's human population is increasing at an annual rate of 2.9%. This growing population needs more energy for basic needs such as cooking and heating and economic activities such as agro-industrial processing. Fuelwood is the source of 80-90% of the energy consumed in rural Africa. For example, fuelwood contributes between 93% and 97% of the total rural and urban energy consumed in Mali (Foley, 1987). Andreae (1991) assumes that the average fuelwood consumption in Africa is 475 kg per person per year. Six studies conducted at various sites in the African drylands indicate per capita consumption of between 354 (Amboseli in Kenya) and 555 kg (Mali) per year (Benjaminsen, 1993, p.404).

Simple multiplication of population levels by per capita use statistics indicates fuelwood shortages in semi-arid areas carrying more than about 40 persons per $km^2$ (DeLeeuw and Reid, 1995). At higher population densities, it is predicted that people will begin to remove live trees and diminish the density of standing woody biomass. It is further predicted that tree removal will be even more rapid around urban centres where trees are removed and converted into charcoal for sale to urban consumers. By these calculations, much of the African drylands, and most of the areas around towns, are being deforested at an alarming rate (see Figure 1). Not only is the burning of fuelwood releasing carbon to the atmosphere, it is also eliminating important carbon sinks.
While some variant of this scenario is occurring in many locations, it is far from ubiquitous. In fact people usually respond to energy shortages in complex ways. First, per capita consumption of fuelwood declines with the size of the consuming unit. In an analysis of fuelwood consumption in Kano, Nigeria, per capita consumption of fuelwood was 456 kg per annum for consuming units of 1-2 persons and 120 kg per annum for consuming units of over 20 persons. Second, two trends exploit these economies of scale: most young male migrants attach themselves to large established urban households and thus do not cook for themselves; many large households cook for others as a petty commercial activity (Cline-Cole et al., 1990).

Third, there is evidence that increasing population density, and thus cropping intensity, is associated with increasing or constant, rather than decreasing, woody cover. Of three study areas around the city of Kano, Cline-Cole et al. (1990) found that the area with the highest density of trees was the one that supported the highest rural population density (500 persons/km2) and was closest to the city. The same area recorded a 2.3% annual increase in tree density between 1972 and 1981 as people planted more trees for a variety of purposes.

Figure 1: Map of drylands with population densities super-imposed

Human Population Density in the Drylands of Africa (1994)



In an area of 322,000 km$^2$ of northern Nigeria including Kano, RIM (1991, p.22) found that cultivated areas had greater wood densities than either grasslands or shrublands (see Table 3). In the Gourma area of Mali, people relied exclusively on deadwood for fuel and thus fuelwood collection did not contribute to deforestation (Benjaminsen, 1993). (Although drought conditions in the late 1980s and early 1990s contributed to the death of an exceptionally large number of trees in that area and thus to a large amount of dead wood.)

Table 3 Wood densities in northern Nigeria, 1990 (Source: RIM, 1991, p.22)

| Stratum | Wood densities (cubic metres per hectare) (without baobab) (a) |
|---|---|
| Grassland | 4.1 |
| Shrubland | 6.1 |
| Cultivation | 7.5 |
| Shrub / grassland | 8.6 |
| Woodland | 20.3 |
| Dense Woodlands | 47.4 |

[a] Baobab is not used for fuelwood.

In an area of high population density in Kenya, Scherr (1995) found that the density of native trees is indeed lower at higher population densities, but that the density and variety of planted and transplanted trees is much higher at higher population densities. Planted tree density is directly related to population density and inversely related to the availability of fallow or common lands. Farmers reported 16 different uses of trees, the most important of which were building poles, fuelwood and fuelwood (Table 4).

Table 4 Number and primary use of trees in Nyanza District, western Kenya
Source: Scherr, 1995, Table 5

| | Number of trees | % of trees |
|---|---|---|
| Total | 171,554 | |
| Mean per farm | 512 | |
| Primary use | | |
| fuelwood | 28,302 | 16 |
| fruit | 12,599 | 7 |
| fencing | 15,187 | 9 |
| building poles | 62,867 | 37 |
| timber | 15,425 | 9 |
| green manure | 14,255 | 8 |
| other | 22,919 | 13 |

This should not imply that there are no fuelwood problems in Africa. Reid (1993) found that there was no successful regeneration of trees around towns in Turkana. Benjaminsen (1993) found that local scarcities of deadwood mean that rural people are spending more time collecting fuelwood and urban residents are spending significant proportions of their incomes to purchase fuelwood and charcoal. Fuelwood scarcities in the most populated areas of Ethiopia have caused people to use dung and crop residues as fuel. This contributes to the problem of soil mining discussed below.

*(P4) Many parts of dryland Africa face a 'fuelwood crisis' of high demand relative to available supplies. People respond to those shortages in different ways in different circumstances. Those responses may lead to the removal of native trees, planting of new trees, or soil mining as people shift from wood to crop residues and manure as fuel sources. People may bear large costs in terms of increased collection times, increased energy costs or reduced nutritional status. Fuelwood markets should be made more efficient so that potential suppliers are able to respond to the high demands.*

## 2.3 Source: Land clearing for agriculture

Clearing of forest and savanna land for agriculture is postulated to be one of the main sources of carbon emissions (Andreae, 1991). Between 1979-81 and 1989-91, it is estimated that the total area of cropland in the continent increased by 5% (WRI, 1994), an average increase of 0.5% or 9,050 km$^2$ per year. Expansions in cultivated area have been particularly large in some countries and some localities. Burkina Faso (28%), Malawi (25%), Swaziland (18%) and Uganda (18) had the highest percentage increases between 1979-81 and 1989-91. In Sudan, the amount of land in rainfed mechanized farming increased from nil in the 1930s to about 30,000 km$^2$ in 1985. A large reduction in the area of Acacia senegal woodland resulted (Elnagheeb and Bromley, 1992).

Because crop cultivation tends to expand onto key grazing resources such as dry-season grazing reserves, agricultural expansion can lead to severe conflicts between agriculturalists and livestock producers (e.g. Little, 1987; Lane, 1991; Bassett, 1993; van den Brink, 1995), although conflicts tend to be muted when crops and livestock are elements of mixed crop-livestock systems. In some circumstances crops can increase overall feed supplies by reducing the fraction of biomass that is removed by fires and raising the output of crop residues (de Leeuw and Reid, 1995). Conflicts also arise when crop cultivation expands into areas containing important wildlife resources. For example, the expansion of settled agriculture into the Zambezi Valley is a matter of passionate national debate in Zimbabwe (*The Herald*, February 14, 1995).

A much larger area of land is cleared by farmers who use fire to clear land as they shift cultivation from one plot of land to another previously fallow plot. If shifting cultivation is practiced on one-half of the land, and farmers cultivate plots for 2 years before letting them revert to fallow, the amount of land cleared each year is about 450,000 km$^2$ or 2% of the habitable area of the continent. Shifting cultivation can be a sustainable land-use system when fallow periods are long enough to restore the productive capacities of the land and there is enough labour available to clear trees from fallow land. In densely populated areas, fallow periods have become very short, resulting in soil erosion, increased weed populations and decreased soil productivity. In areas with low population densities, many farmers lack the labour necessary to clear land that has long been fallow. This results in the repeated use of the same land and eliminates the restorative fallows.

*(P5) Land clearing for agriculture is a particularly important problem in certain countries and localities. It may be possible to design policies and programmes to mitigate some local problems.*

*(P6) The periodic clearing of land that characterizes shifting cultivation is a more serious continental problem but one that may be less amenable to solution by changes in policy or programme. The amount of land involved in shifting cultivation will gradually decline as population densities increase.*

2.4 Source: Mining soil organic matter

Shorter fallow periods and longer cropping periods can only be sustained if farmers adopt new ways to conserve and replenish soil nutrients and organic matter. The situation in many parts of Africa, however, is one of continuous cultivation, burning or removal of crop residues, and low input of inorganic or organic fertilizers. The results are reductions in soil nutrients and organic matter and emissions of carbon into the atmosphere.

Van der Pol (1992) calls this soil mining. He compiles evidence from a number of studies conducted in southern Mali to show that the cultivation of all of the traditional food crops (millet, sorghum, maize and rice) and cash crops (cotton, groundnuts) generate large net reductions in soil nitrogen, potassium and magnesium. He estimates that replacing the lost nutrients with inorganic fertilizers would cost an average of about $US 59 per hectare per year across all of the cultivated and fallow areas of southern Mali.

Soil mining can be mitigated or reversed through the use of organic or inorganic fertilizers, crop rotations including legumes, and better management of organic matter, nutrients and water. There are many encouraging examples of farmer adoption of such techniques in the African drylands. Sanders et al. (1990) note that farmers in the semi-arid areas of Burkina Faso tend to adopt water conservation techniques such as tied ridges (25% of farmers in sample villages), earthen dikes (about 600 $km^2$ of land in the central Plateau) and stone dikes (65 $km^2$ of land in the central Plateau), while farmers in the dry sub-humid areas tend to adopt soil fertility techniques such as inorganic fertilizers.

Most farmers now use fertilizers and ox ploughs in the most populous areas of northern Nigeria (750 - 1,250 mm ave. annual rainfall) (Goldman and Smith, 1995). In the last 20 years farmers in The Gambia (700-1000 mm ave. annual rainfall) have rapidly adopted animal traction packages that combine speed – using horses and donkeys for draught in addition to oxen -- with minimum and no tillage cultivation – seeders and toolbars rather than ploughs. As of 1987/8, 70% of all farm units owned at least one draft animal, 89% of those owned at least one horse or donkey used for traction and 94% owned a seeder and/or a toolbar (Sumberg and Gilbert, 1992). The Kofyar farmers who live on and around the Jos Plateau in northern Nigeria have constructed tied ridges and terraces and apply livestock manure and crop residues to maintain soil fertility (Netting, 1993). Farmers in Machokos District of Kenya have constructed *fanya juu* bench terraces, drains and dams, adopted animal traction, stored and stall fed crop residues and planted trees (Tiffen et al., 1994).

*(P7) Crop production and management practices that mine soil nutrients emit carbon and contribute to low yields of crops and crop residues. Under the right conditions, farmers will adopt new techniques that conserve water, organic matter and nutrients. Incentive-compatible programmes for reduced carbon emissions will need to identify and build upon those conditions.*

### 2.5 Source: Savanna burning

The major source of carbon dioxide and other greenhouse gases in the African drylands is burning of savanna grasslands. Andreae (1991) estimates that 3690 million tonnes of dry matter are burnt each year on the world's savannas, releasing 1660 million tonnes of carbon to the atmosphere, and that two-thirds of the savannas burnt are in Africa (removing 2430 million tonnes of drymatter and 1094 million tonnes of carbon). By these estimates, burning of African savannas contributes about 32% of all carbon released to the atmosphere by biomass burning in the tropics. However, other estimates indicate much lower amounts of drymatter burnt and carbon released (De Leeuw, 1992; Manaut et al., 1991).

Burning of savanna grasslands is sometimes a deliberate range management technique, frequently is an accidental bi-product of burning for land clearing, and perhaps most often is an incidental consequence of small grass fires set to control pests and flush game. Andreae (1991, p. 268) argues that burning is an appropriate technique for stopping shrubs and bushes from over-whelming the grassy vegetation that is the preferred grazing of most domesticated livestock. Fires tend to be less prevalent in areas with higher population densities and more intensive land use (de Leeuw, 1992).

*(P8) Savanna burning is such a widespread phenomena in the African drylands that farmers must perceive important benefits. A good understanding of the benefits and costs to farmers would be necessary for planning any intervention for reducing burning.*

### 2.6 Sink: Perennial fodder crops and planted pastures

Several authors have written about the potential for deep-rooted perennial grasses to store carbon. Fisher et al. (1994) report that about 350,000 km2 have been sown to *Andropogon gayanus* and *Brachiaria humidicola* in high rainfall areas of South America. Results from two sites in Columbia indicate that pastures planted to *A. gayanus* and *B. humidicola* store significantly more carbon than natural savannas. Even greater differences are found between natural savannas and natural pastures sown to *B. humidicola* and the legume *Arachis pintoi.* Woomer et al. (1994, p.10) propose that "pasture systems seem to be among the more suitable agroecosystems in terms of soil carbon storage and conservation."

The patchy data available on the areas planted to fodder crops and planted pastures in Africa, indicate that the total area is very small. In 1979 there was about 2,400 $km^2$ of land planted to fodder crops in Kenya, mostly napier grass (Stotz, 1983) and in 1985 there was at least 120 $km^2$ under planted pasture in Tanzania (Lwoga and Urio, 1987). Most of this was outside of the drylands. In the late 1980s about 200 $km^2$ of cropland was planted to fodder crops in the drylands of francophone West Africa (Lhoste, 1990 as cited in de Leeuw 1994). Between 1987 and 1991 637 fodder banks of the leguminous forage *Stylosanthes hamata* were established in northern Nigeria. At an average of 4 hectares per fodderbank, the total area covered is only 2.8 $km^2$ (Ajileye et al., 1994).

According to Ajileye et al. (1994) the main constraints to greater adoption of fodder banks in Nigeria are: (a) insecure property rights that pastoralists have to land; (b) the large amount of labour that is required to start and maintain a fodder bank; (c) lack of suitable fodder species for the semi-arid areas; (d) the high costs of fences recommended to protect the fodder banks; and (e) the risk associated with committing land to a particular crop for a number of years (also see section 2.8 below).

Mohammed-Saleem and de Leeuw (1994) assessed the land suitabilities for *Stylosanthes hamata* cv Verano in ten countries of West Africa. The results indicate that none of the semi-arid or arid areas are 'very suitable' for Verano and that all of the very suitable and much of the suitable areas are in the sub-humid zone. McIntire et al. (1992) argue that the adoption of planted forages is affected by: (i) mobility – more mobile production strategies provide fewer possibilities for forage production; (ii) labour supply – forage production and feeding are labour-intensive; (iii) insecure or temporary tenure; and (iv) supply of feed from other sources – e.g. agro-industrial by-products. Temporary or insecure tenure may also constrain the adoption of perennial forages. Planted forages are likely to be adopted where livestock products are of particularly high value (e.g. milk in peri-urban areas) and where high population density reduces the availability of other feeds.

> *(P9) A variety of perennial grasses and legumes are available 'on the shelf' to African farmers. The greatest potential for increased adoption of those grasses is in sub-humid areas where there is high demand for livestock products and relatively few alternative sources of feed supply.*

## 2.7 Sink: Trees in forests and woodlots

Trees are the classic carbon sink: they store significant quantities of carbon in above-ground woody biomass and sometimes even larger amounts as root biomass. Large amounts of carbon may accumulate over time as new forests are established. Afforestation has therefore been a particular focus of carbon off-set projects undertaken to date (see above). Nilsson and Schopfhauser (1995, Table V) estimate that 270,000 $km^2$ could be available for plantations in all of Africa, an area about 15% as large as the total area of cropland (1.8 million $km^2$) on the continent (WRI, 1994).

> *(P10) There may be some potential to enhance the carbon stored in state-managed plantation forests or forest reserves. Another alternative is communal forests managed for sustainable production of fuelwood and other subsistence products; compatible with the collective interests of the local population (see section 3.2 below).*

## 2.8 Sink: Agro-forestry

Unruh et al. (1993) and Schroeder (1993) argue that an under-estimated sink of carbon is agro-forestry – systems in which trees are grown with crops, livestock, or both. These systems are potentially important for three reasons: (i) the area currently used for crop and livestock production is very large; (ii) the woody biomass produced in agro-forestry systems could provide a renewable substitute for non-renewable fossil fuels; and (iii) agro-forestry contributes to the intensification of agriculture and thus reduces the need for shifting cultivation or additional land clearance. Unruh et al. (1993) use GIS analysis to estimate the areas currently available for different types of agro-forestry and total above-ground biomass potentially stored in those different types.

Evidence on tree densities and tree planting indicate that farmers have already exploited some of this potential. As discussed in sub-section 2.2 above, the evidence generally indicates that native trees are thinned as the density of human population and land-use intensity increase. Small-holder agriculturalists tend to conserve certain types of trees such as the karite (*Butyrosperum parkii*) and acacia senegal that produce valuable products (Benjaminsen, 1995), while large-scale mechanized agriculture tends to clear almost all trees (Larson and Bromley, 1990). The evidence also indicates that farmers transplant and plant selected trees around their homesteads and along boundaries between fields. The longer people remain settled in a confined area, the greater the variety of trees they plant. The species of trees that are chosen depend upon those that are available, the market demand for different tree products, and the farmers' perceptions of the scarcity of different tree products (e.g. Scherr, 1995).

Nonetheless there is concern about the rate of adoption of certain types of agro-forestry practices. A technique that has attracted tens of millions of dollars for research and promotion during the last 15 years is variously known as alley cropping, alley farming, and hedgerow intercropping. All three terms refer to systems in which fast-growing leguminous trees and shrubs (such as *Leucaena leucocephala* and *Gliricidia sepium*) are established in hedgerows with food crops cultivated in the alleys between the rows. Trees and shrubs are regularly pruned to prevent them from shading out the crops and prunings are either used as a mulch for food crops or fed to livestock with the manure returned to the alleys. On-station trials produced very promising results. For example, Ehui and Spencer (1993) showed that the alley-farming system they examined was more sustainable and more economically-viable than the farmers' traditional cropping systems.

Alley farming has not proven to be popular with farmers. Insecure land tenure was initially identified as the major constraint (Francis, 1987) and prompted ILCA (the International Livestock Centre for Africa) to commission land tenure studies in Cameroon, Nigeria and Togo. Those studies showed that a significant portion of the land in each country was under a secure form of tenure and thus, "land tenure is not likely to be a major constraint to the adoption of alley cropping" (Lawry et al., 1994, p.186). More recent research indicates that adoption is also constrained by several other technical and socio-economic factors: (i) inappropriate tree species for certain conditions, (ii) more competition between hedgerows and crops than originally thought, (iii) alley farming demands high labour inputs and is inflexible in the timing of its labour requirements, and (iv) the main benefits of alley farming are delayed until 3-4 years after the alleys are established (Carter, 1995).

*(P11) Agro-forestry has a large untapped potential for carbon storage and income generation. Agro-forestry extension and promotion programmes need to appeal to farmers' needs and incentives.*

Table 5 Summary of propositions on carbon sources and sinks in the African dryland

**(P1)** In low rainfall areas, livestock grazing has little impact on the structure of herbaceous vegetation and either negative or positive impacts on woody vegetation.
**(P2)** In higher rainfall areas, overgrazing can reduce the abundance of the most palatable grasses and thus the potential to produce secondary outputs. Vegetative cover usually decreases and increased soil erosion might occur if cover is very low. The woody species that replace the perennial grasses may be of more or less value for carbon storage.
**(P3)** There are complex relationships between livestock and trees in dryland areas. While very heavy browsing can retard tree growth, livestock can play important roles in mediating tree growth.
**(P4)** Many parts of dryland Africa face a 'fuelwood crisis' of excess demand. People respond by removing trees, planting new trees, or using crop residue and manure as fuels. People may bear large costs in terms of increased collection times, increased energy costs and reduced nutritional status. Fuelwood markets should be made more efficient so that potential suppliers are able to respond to the high demands.
**(P5)** Land clearing for agriculture is a particularly important problem in certain countries and localities. Policies and programmes may be designed to mitigate some local problems.
**(P6)** Shifting cultivation is a serious continental problem but one that may not be amenable to solution by changes in policy or programme. The amount of land involved in shifting cultivation will gradually decline as population densities increase.
**(P7)** Crop production that mines soil nutrients also emits carbon and contributes to low crop yields. Under the right conditions, farmers will adopt new techniques to conserve water, organic matter and nutrients. Programmes for reduced carbon emissions will need to identify and build upon those conditions.
**(P8)** Savanna burning is such a widespread phenomena in the African drylands that farmers must perceive important benefits. A good understanding of the benefits and costs to farmers would be necessary for planning any intervention.
**(P9)** A variety of perennial grasses and legumes are available to African farmers. The greatest potential for increased adoption of those grasses is in sub-humid areas where there is high demand for livestock products and few alternative sources of feed supply.
**(P10)** There is some potential for enhanced carbon storage in state-managed forests. Another alternative is communal forests managed for sustainable production of fuelwood and other products, compatible with the interests of the local population.
**(P11)** Agro-forestry has a large potential for carbon storage and income generation. Extension and promotion programmes need to appeal to farmers' needs and incentives.

## 3. DEVELOPMENT AND CHANGE IN THE AFRICAN DRYLANDS

The propositions presented above begin to point the way toward programmes and policies that could reduce carbon emissions and / or enhance carbon storage in the African drylands. To be successful, however, programmes and policies should be compatible with the individual and collective interests of Africa's farmers and livestock owners. And to be sustainable, they need to be consistent with demographic, economic and political trends. This section of the paper is divided into two subsections: sub-section 3.1 gives some general comments about the current situation in rural Africa; sub-section 3.2 describes some important trends shaping agricultural development and resource management on the continent. As above, the main implications are summarized in several propositions.

### 3.1 The base situation

*Human welfare and economic conditions.* Quantitative measures of economic performance and human welfare depict a discouraging situation for Africa as a whole. In terms of GDP per capita, 23 of the world's 25 poorest countries are in Africa. In terms of the UNDP human development index (a composite of statistics on life expectancy, health, nutrition, education, income and equity), 21 of the world's 25 least developed countries are in Africa (UNDP, 1992). Agriculture continues to be the main source of employment – employing 68% of all workers in Africa (WRI, 1994), and a major source of GDP -- comprising 31% of GDP for all of sub-Saharan Africa (Cleaver and Schreiber, 1994). Strong linkages between farm and non-farm sectors mean that agriculture "can be a dynamic lead sector in rural growth strategies" (Hopkins et al., 1994, p.10).

*Risk.* An over-arching characteristic of agricultural production in the African drylands is risk. Crop and livestock farmers are exposed to environmental, property and market risks. Environmental risks are high due to the high degree of spatial and temporal variability in rainfall and to diseases, pests and interactions with wildlife. Rainfall risks tend to be relatively high in drier areas (where the coefficient of variation in inter-annual rainfall can be as high as 60%), while disease and pest risks tend to be relatively high in more humid areas (e.g. trypanosomiasis). A property risk entails lack of secure expectations about future access to or control over a resource. Property risks are especially high in areas of civil war, unrest and banditry.

Extensive livestock production is exposed to high property risks regarding livestock, water and forage resources. Other risks are associated with the markets on which households seek to obtain agricultural inputs, foodstuffs and other consumables in exchange for their surplus agricultural products or labour. An important characteristic of markets for livestock and crops in Africa is the correlation between environmental and market risks; terms of trade between crops and livestock move against livestock as drought conditions require livestock be sold or die (Swallow, 1994).

Farmers adopt complex strategies for dealing with those risks. Crop farmers intercrop, stagger the planting of different crops and varieties with different maturation periods, and disperse plots up and down hill slopes (e.g. Webb et al., 1992). Livestock farmers diversify the species and breeds of livestock they hold, and undertake mobility, ranging from daily herd movements to seasonal transhumance to migration. If possible, most farmers diversify their sources of income to include crops, livestock, wage labour and self-employment.

*Missing and imperfect markets.* Among the most important constraints to increased agricultural production in Africa are markets. Markets for land are constrained by national laws in countries such as Ethiopia and Tanzania and by customary rules on the use and transfer in all parts of Africa. Even where land titling systems have been implemented, as in Kenya, people must bear high transaction costs to register land transfers. Most land transactions, especially bequests, rentals and leases, therefore occur with little if any reference to the formal system.

Although land tenure is the frequent scape-goat for poor agricultural performance in Africa, studies conducted by the World Bank, Land Tenure Centre and ICRISAT in Kenya, Somalia, Burkina Faso, Ghana and Rwanda do not show any relationship between land tenure status, agricultural production, or investments in land improvements (Place and Hazell, 1993; Bruce and Migot-Adholla, 1994). The evidence rather indicates that increases in agricultural production are constrained by missing and mal-functioning markets for many agricultural inputs and outputs (e.g. Carter, et al., 1994). Markets for insurance and credit are largely unavailable to most farmers (Bromley and Chavas, 1989; Platteau, 1988), so they adopt a variety of risk-spreading strategies and are conservative in their approach to new technologies (see above). Markets for imported inputs such as fertilizers, pesticides and vaccines are highly variable. Parastatals in countries such as Malawi and Zimbabwe have been reputed to be quite effective in distributing inputs for crop production; cotton producers in francophone West Africa have also benefitted from relatively steady supplies. In other countries, such as Ethiopia, it has been difficult for most farmers to get any imported inputs.

Rural labour markets are also restricted in many areas (e.g. Ethiopia, Benin); even where they do exist they often are constrained by high costs of labour monitoring and lack of information. Farmers therefore rely primarily on labour available within their households or through communal work parties (Ade Freeman, 1994b). Finally, the 'market' for information about new agricultural techniques and management practices is limited by poor linkages between farmers, extension workers and agricultural researchers (Osborn, 1995).

*Infrastructure and transaction costs.* A principle reason for missing and mal-functioning markets is the poor state of the rural transport and communication infrastructure. Spencer (1994) reports that the density of rural roads in the humid and sub-humid areas of Africa is only 63 km/1000 km$^2$, about half of which requires substantial rehabilitation. With 97 km /1000 km$^2$, Nigeria has the highest density of rural roads in humid or sub-humid Africa; in 1950 India had a similar density of population to present-day Nigeria but a road density of 718 km /1000 km$^2$. Research across several Asian and African countries has shown the importance of rural roads for expansions of agricultural production and rural economic growth (Spencer, 1994). Poor transport infrastructure also exacerbates drought-induced famine in Ethiopia (Webb et al., 1992).

*Policy.* In this context of poverty, risk, imperfect markets and poor infrastructure, African governments have implemented, or attempted to implement, a variety of policy instruments to promote their avowed objectives of food security, economic growth and environmental protection. Most of these policies, at least until the 1980s, reflected a belief in the advantages of command and control and the disadvantages of private enterprise. State marketing boards were established allegedly to protect farmers from the vagaries of the international market and exploitation by greedy capitalists. In fact, marketing boards were often used as instruments for providing cheap food to urban consumers and extracting economic surplus from sale of export crops (Bates, 1981). The colonial trend toward centralization of authority over resource use and management continued in the form of 'state ownership' of land in most of west and central Africa (Riddell and Dickerman, 1986). Rigid resource use rules such as the forest codes of West Africa (Lawry, 1989) or the grazing regulations of Lesotho (Swallow and Bromley, 1995) were implemented to govern individual use of 'state property'.

*(P12) The overall results of these risks, multiple-market failures, poor rural infrastructure and government policies were:*

*(i) continued reliance on civil institutions that help farmers to pool risks but offer few potential gains from trade (Bromley and Chavas, 1989);*

*(ii) farmers diversify their income sources rather than specialize and exploit economies of scale in production and distribution;*

*(iii) dampened incentives to agriculture as a whole and export crops in particular (Byerlee and Sain, 1986; Williams, 1990);*

*(iv) farmers adopt 'safety-first' rather than profit-maximizing strategies (Wiebe, 1991);*

*(v) farmers are very conservative in their adoption of new techniques and tend to adopt new techniques on a step-wise basis; and*

*(vi) many farmers remain exposed to high levels of environmental and market risks despite the number of self-insurance and group insurance mechanisms they employ.*

3.2 Trends

*Overall economic conditions.* Africa experienced poor economic growth and slow agricultural development during the 1970s and 1980s. The countries of sub-Saharan Africa experienced annual growth in per capita GNP of 1.5% between 1965 and 1980 and -1.7% between 1980 and 1989 (UNDP, 1992, Table 24). Agricultural GNP per capita decreased by 1.2% between 1970-80 and by 1.3% between 1980 and 1991. Between 1979-81 and 1989-90, per capita food production fell by 6%. Food imports increased by 185% between 1974 and 1990, food aid increased by 295% (Cleaver and Schreiber, 1994).

The growth of Africa's agriculture was in part constrained by government policies that dampened incentives to farmers. The 1980s also happened to be a period of declining prices for the major agricultural commodities traded on the world market. World markets for livestock products were destabilized by surplus production of beef and milk in Europe and other high-income regions of the world. Some of the surplus was dumped on African markets at very low prices. The meat markets of coastal West Africa (especially Cote d'Ivoire), which had been a major source of income for traditional exporters such as Niger, Mali and Burkina Faso, were undermined by dumping by cheap imports from the countries of the European Union.

*Structural adjustment.* The 1980s was also the period when 34 African governments launched economic adjustment programmes with support from the World Bank and International Monetary Fund. Reforms were undertaken to stabilize macro-economic conditions, improve financial markets, devalue exchange rates, streamline government bureaucracies, liberalize markets for agricultural inputs and outputs, and expand private involvement in the supply and finance for public goods. The evidence indicates that the countries that have sustained structural adjustment programmes over several years have begun to enjoy some benefits in terms of renewed economic growth (Veit et al., 1995). Using pooled data for 28 Africa countries for four time periods between 1960 and 1987, Savvides (1995) found that GNP growth was positively related to: (i) accumulation of physical capital, (ii) growth of the financial sector, and (iii) political freedom, and negatively related to: (iv) inflation and growth of the government sector. As a group the countries of the CFA Franc Zone (with an over-valued currency supported by France) did not grow as fast as other African countries (Savvides, 1995). Twenty-one sub-Saharan African countries achieved positive growth in GNP per capita between 1988 and 1993, over half grew by 5% or more. 1994 appears to have been a particularly good year with

12 countries growing by more than 5% and only 11 experiencing negative growth (down from 17 in 1993) (Veit et al., 1995).

After the first 4-5 years of structural adjustment it became apparent that economic reform and recovery are slow and complex processes; it takes time for government rhetoric to translate into concrete actions (e.g. Leonard, 1992) and still more time for government employees, consumers and private entrepreneurs to develop new expectations about the role of government. Meanwhile, the processes of adjustment often result in at least temporary increases in poverty and social problems as government employment contracts, government subsidies and transfers are reduced, and markets adjust to national and international economic forces. Liberalization of the markets for foreign exchange typically result in increases in the prices of tradables relative to non-tradables. Many urban dwellers lose because they are consumers of food and producers of non-tradables (Ferroni and Gootaert, 1993).

For rural residents, the initial effects of structural adjustment depend upon whether they are large users of subsidized fertilizers, net buyers or sellers of food, and whether their surplus products are tradables or non-tradables. Asuming-Brempong (1994) found that liberalization of the exchange rate and removal of fertilizer subsidies in Ghana resulted in decreases in production and lower financial returns per hectare planted to cereals (esp. maize and rice). Ade Freeman (1994a, 1994b) predicted that the removal of the fertilizer subsidy will also reduce returns to maize production in Nigeria and Benin. His linear programming model further predicts that farmers will shift from maize, which requires high fertilizer inputs, to sorghum / millet intercropping and leguminous crops such as groundnuts, bambara nut, and cowpeas which use relatively small amounts of fertilizer. Negative income effects will be balanced against the positive environmental effects of these more sustainable cropping patterns.

*Ten responses to population pressure.* Boserup's (1965) classic book, *Conditions of Agricultural Growth,* still guides most studies of the processes of agricultural intensification. In simple terms, population growth increases the amount of food and fiber needed for subsistence. Farmers initially respond by cultivating existing cropland more frequently and bringing new land, some of which had formerly been reserved for collective uses such as dry-season grazing and fuelwood collection, into crop production. Resources that remain as common property (e.g. native trees) are exploited more heavily and possibly over-utilized. Farmers and communities can respond to this situation in at least ten ways:

(i) continue the degenerative practices and mine the soil nutrients until yields become so low that farming is unprofitable;

(ii) migrate away from the areas of high population density and land degradation;

(iii)) supplement household and on-farm resources with hired labour and / or purchased fertilizers;

(iv) augment household inputs with animal traction (oxen or equines) and mechanical cultivation devices;

(v) integrate crops and livestock through animal traction and nutrient transfer and cycling (McIntire et al., 1993);

(vi) make investments in land improvements for erosion control and / or water retention;

(vii) bring more leguminous crops such as cowpeas and pigeon peas into cropping rotations (*African Farming,* May/June 1995, p.16);

(viii) make the short-duration fallow more productive by planting forage legumes or perennial grasses (see section 2.6 above);
(ix) undertake intensive specialized production of outputs that are particularly profitable; and / or
(x) undertake intensive diversified production strategies (e.g. crop - livestock - trees).

The evidence suggests that different combinations of these responses have been employed in different locations.

* In Mali and elsewhere in semi-arid West Africa, farmers graze animals on crop residues and apply manure to some of their fields. Unfortunately the number of animals and pasture available to supply nutrients are insufficient to meet nutrient needs and thus soils continue to be mined. Fields close to the households receive most of the available manure (Spiers and Olsen, 1992; Gavian, 1993).
* In the densely-populated central Plateau of Burkina Faso, some farmers have made substantial investments in water retention structures, while many have simply moved to the less-populated areas in south-west Burkina Faso or northern Cote d'Ivoire (Sanders et al., 1990).
* In the densely-populated regions of northern Nigeria, farmers use oxen ploughing (RIM, 1992), employ hired labourers for timely control of weeds, apply relatively large amounts of inorganic fertilizers (Ade Freeman, 1994a) and rear large numbers of livestock in crop-livestock production systems (RIM, 1992).
* Farmers in northern Benin use less fertilizer, employ fewer hired labourers and keep fewer livestock than farmers in northern Nigeria, but plant more leguminous crops (Ade Freeman, 1994b).
* In the Machakos area of Kenya, farmers have used off-farm sources of income to invest in bench terraces that protect the soil against degradation (Tiffen et al., 1994).
* Farmers have specialized in dairy in the Kenyan highlands, sorghum in Sudan, cotton in many parts of West Africa.

Inter-regional differences in intensification clearly depend upon differences in agro-ecological conditions. Market conditions and government policies, at the sectoral and macro-levels, also have large influences on development pathways. Larson and Bromley (1991) studied how the policies of the Sudanese government undermined the profitability of gum arabic and contributed to the destruction of acacia senegal trees. Ade Freeman (1994b) used his linear programming model to predict farmer responses to population-driven agricultural development (e.g. Benin) versus market-driven agricultural development (e.g. northern Nigeria). In population-driven systems livestock are largely treated as inputs into crop production; in market-driven systems, meat and milk may be regarded as important products in their own right.

*Differential access to non-farm sources of rural income.* One reason for the discrepancy between the percentage of people working in agriculture (68%) and the contribution of agriculture to gross domestic product (31%) is that many people who farm also derive large percentages of their income from non-farm sources. For example, non-farm sources comprised about 50% of total income in Machakos, Kenya, in the mid-1970s (Tiffen et al., 1994, p.161). A study conducted in Rwanda in 1985/6 found that off-farm income comprised 57.5% of total income, and reached 80.1% in the smallest farm-size quartile of the population (von Braun et al., 1991). Wage labour can be particularly important during times of stress. For example, in a study of Taita households (Kenya), Fleuret (1989) found that 39% of households engaged in migrant labour in a normal year compared to 53% in a drought year. Ring (1990) and Sperling (1987) found that Dinka (Sudan) and Samburu (Kenya) pastoralists use income from wage labour to initially construct and reconstruct their herds after drought. Restricted access to non-agricultural sources of income exacerbated the effects of the Ethiopian famine of the mid-1980s (Alemneh Dejene, 1990).

*Environmental conditions and their impacts.* Besides distorted local markets and unfavourable terms of trade on the international markets, the drylands of West Africa also suffered from unfavourable rainfall conditions in the 1970s and 1980s. Thomas and Middleton (1994, Figure 7.3) show that that the Sahel experienced abnormally low rainfall every year (with one exception) between 1967 and 1991 (Thomas and Middleton, 199x, Figure 7.3). Fortunately, rainfall conditions were excellent in most of the Sahel in 1994 and after a dry June, appear fairly to be fairly good this July (FEWS Bulletin, July 21 1995).

Poor rainfall conditions stimulated massive migration throughout the drylands of west and central Africa. People and their animals moved from drought-affected areas of Mauritania, Senegal, Mali, Burkina Faso, Mali and Chad into more humid areas of those countries and to neighbouring countries to the south. The results of international migration of livestock keepers is reflected in national statistics. Between 1975 and 1987, cattle populations decreased in Senegal (-1.8% per annum), stayed constant in Mauritania, and increased slowly in Mali (1.6% per annum), Niger (2.3% per annum) and Chad (1.3% per annum), while they increased rapidly in Cameroon (4.1% per annum), Central African Republic (8.7% per annum) and Cote d'Ivoire (5.8%) (ILCA, 1993). Control of trypanosomiasis through chemotherapy (Central African Republic) and tsetse control (Cote d'Ivoire) has helped to foster this southward movement of livestock.

*Constraints on flexibility and opportunism.* Extensive livestock production is one of the most appropriate types of land-use in dryland Africa because of its adaptability to the highly variable environmental conditions. Animals can be regularly moved from location to another to follow seasonal climatic patterns (e.g. seasonal transhumance in Lesotho) or within a particular location to track local variability in the quality and quantity of forage (van den Brink et al., 1995). Certain patches or key resources can be reserved for particular times. Additional flexibility is possible if livestock owners split their herds into small groups, keep several livestock species, or have access to good markets on which to sell animals when forage is in short supply and buy new animals when conditions improve (Swallow, 1994; Scoones, 1994).

During the last 20 years, many extensive livestock producers have experienced a decrease in their ability to track forage resources across the landscape. There are several constraints on flexibility and mobility. Shortages of knowledgeable and skilled labour is one constraint. For example, White (1990) noted that migration of young males among the WoDaaBe of central Niger has led to shorter herd movements, less herd splitting, poorer disease management, and greater reliance on boreholes than dug wells. Increases in absentee ownership of livestock is a second constraint. The third and greatest constraint across much of Africa is restricted access to land resources. For example, the pastoral Fulani of Mali who once managed the Niger Delta in Mali as a dry-season grazing reserve now have no access to some areas and restricted access to others. Creation of national parks and conservation areas in some countries (e.g. Tanzania, Burkina Faso, Kenya) and state-sponsored farms in others (e.g. Tanzania, Sudan) have eliminated dry-season grazing reserves.

Expansion of agriculture, while not concerned with large areas of land on a continental basis, often affects key grazing resources and migration routes. And the risks of theft and violence restrict use of many dryland areas. In 1988 as much as half of the Turkana highland grazing areas were ceclared to be 'no-go' areas by the Kenyan authorities (Lane and Swift, 1989).

*Policies and programmes for livestock and range management.* De Haan (1994) discusses the way that the World Bank has modified its approach to pastoral development over the last 30 years. He outlines four overlapping phases: (i) the ranching phase (mid 1960s to mid 1980s) – transfer of ranching technology to tropical areas with heavy capital investments on parastatal ranches; (ii) the range / livestock phase (mid 1970s to late 1980s) – development of communal areas through construction of infrastructure and adjudication of land rights to pastoral groups; (iii) the pastoral association phase (ongoing since mid-1980s) -- empower pastoral associations to organize public goods such as wells and services such as animal health; and (iv) integrated natural resource management (ongoing since early 1990s) – comprehensive attention to natural resource management with consideration for the interests of various stakeholders. At the same time, there has been a strong downward trend in donor support for extensive livestock and pastoral development with the World Bank cutting its support for programmes in sub-Saharan Africa from about US$150 million per year in the 1980s to about US$25 million per year at present (de Haan, 1994). USAID essentially ceased all support for livestock / range development projects in the mid-1980s and now supports a limited number of community resource management projects, some of which cover dryland or pastoral areas.

Based on a review of World Bank projects in Senegal, Mauritania, Mali and Niger, Vedeld (1992) supports de Haan's contention that World Bank projects are tending toward greater participation by the intended beneficiaries. All of the projects have limitations, however, in terms of: (i) formal recognition of pastoralists' land and water rights, (ii) contingency plans for droughts, (iii) social science support for development of new organizations, and (iv) definition of intended beneficiaries.

*Policies and programmes for participatory natural resource management.* Since the mid-1980s there has been a gradual change in donor programmes and government policies toward natural resource management. In the francophone countries of Sahelian West Africa the approach of *gestion des terriors villageois* or *amenagement des terriors villageois* has become fashionable. The concept is that agrarian communities should exercise authority over natural resources within the areas they exploit and that governments should support local communities by providing institutional, technical, financial and political support. Donors and governments have adopted similar approaches of community resource management in eastern and southern Africa, but perhaps with more focus on particular resources.

Grazing associations in Lesotho have focused on management of mountain rangelands (Lawry, 1989); the CAMPFIRE programme in Zimbabwe has focused on management of wildlife resources (King, 1994).

While the terrior approach is an improvement over the rule-based approach that it replaces, it has yet to be a resounding or widespread success. One criticism is that the village terriors are too confining for flexible livestock production for those resident within the areas or for transhumant livestock producers who would normally cross several terriors (Benjaminsen, 1995). Degnbol (1995) argues further that the current top-down approach to *gestion des terriors* in Mali is fundamentally inconsistent with its avowed aims of decentralization and empowerment of local residents and community organizations.

*Participatory research and development.* One of the themes of the latest generation of pastoral development and natural resource management projects is the need for active participation by the intended beneficiaries. In fact participation is now regarded as essential for research, extension, development and resource management. Researchers have contributed to this new emphasis with their interests in indigenous knowledge and their recognition of the need to have new techniques adopted in order to have impact. Donors and governments have contributed because they need to develop alternative sources of finance and delivery of goods and services that were previously supplied by government agencies. Non-governmental organizations have contributed because they deal intimately with the practical problems of improving human welfare.

*(P13) Programmes intended to improve the management of natural resources are most likely to succeed if they are based on three principles: co-management, subsidiarity and over-lapping rights. Co-management regimes are cooperative management arrangements between state and local organizations in which states assign group rights, establish guidelines for inter-group interactions, and help create positive environments for local organizations to operate. The latter then organize and mobilize local participation (Swallow and Bromley, 1995). Subsidiarity is the principle that "tasks should be carried out as near to the level of actual users of resources or beneficiaries ... as is compatible with efficiency and accountability" (Swift, 1994). Overlapping rights refers to the fact that most resources in the African drylands are used by many users, often for different types of uses.*

*(P14) To be successful, programmes designed to increase vegetative cover to African drylands must take account of national policies at the macro and sectoral level and local variations in agro-ecology, market conditions, off-farm employment opportunities and land-use intensity. Programmes also need to be flexible enough to account for the radically-different environmental conditions that might be experienced in an area at different times during the programme.*

*(P15) Sectoral policies and programmes need to be consistent with market liberalization and privatization in order to be sustainable.*

Table 6 Summary of propositions on development and change in the African drylands

(P12) The overall results of the risks, multiple-market failures, poor rural infra-structure and distorting government policies affecting agriculture in sub-Saharan Africa were:

(i) continued reliance on civil institutions that help farmers to pool risks but offer few potential gains from trade (Bromley and Chavas, 1989);

(ii) farmers diversify their income sources rather than specialize and exploit economies of scale in production and distribution;

(iii) dampened incentives to agriculture as a whole and export crops in particular (Byerlee and Sain, 1986; Williams, 1990);

(iv) farmers adopt 'safety-first' rather than profit-maximizing strategies (Wiebe, 1991);

(v) farmers are very conservative in their adoption of new techniques and tend to adopt new techniques on a step-wise basis; and

(vi) many farmers remain exposed to high levels of environmental and market risks despite the number of self-insurance and group insurance mechanisms they employ.

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(P14) To be successful, programmes designed to increase vegetative cover to African drylands must take account of national policies at the macro and sectoral level and local variations in agro-ecology, market conditions, off-farm employment opportunities and land-use intensity. Programmes also need to be flexible enough to account for the radically-different environmental conditions that might be experienced in an area at different times during the programme.

(P15) Sectoral policies and programmes need to be consistent with market liberalization and privatization in order to be sustainable.

## 4. INCENTIVE-COMPATIBLE POLICIES AND PROGRAMMES

The previous sections reviewed some of the evidence on the sources and sinks of carbon and the processes of development and change shaping agriculture and the environment in rural Africa. In this final section I take the next steps toward workable carbon off-set programmes by suggesting some considerations for programme design and some sinks and sources that might receive priority attention. The penultimate sub-section is a list of some needs for additional research.

### 4.1 Considerations for programme design

*Labour-intensive public works* programmes can be effective mechanisms for directly reducing rural poverty and generating public goods such as physical infrastructure (e.g. roads, water management infrastructure) or community forests. Tesfaye Teklu (1994) has found that labour-intensive public works programs for constructing roads in Botswana and Kenya were effective in increasing the incomes of the working poor.

*Environmental cross-compliance* should be considered as a means for simultaneously alleviating constraints to agricultural production and advancing environmental objectives. For example, a public institution could provide operating credit at a favourable rates to dairy farmers who plant deep-rooted perennial grasses on certain areas of land. In the U.S., cross-compliance has been used to link commodity programmes and support payments to compliance with certain conservation standards. Red-ticket cross compliance makes participation in a support programme contingent upon a farmer attaining a certain environmental standard; green-ticket cross-compliance makes the level of a support payment contingent upon the farmer's performance vis-a-vis the environmental standard (Russell and Fraser, 1995).

*The CAMPFIRE approach* to management of wildlife in Zimbabwe could be extended to provide incentives for other types of collective action. For example, communities that establish and manage community woodlots could be given 'carbon dividends' for achieving certain standards of performance. Dividends could take the form of public infrastructure such as schools or clinics or cash dividends to all community members.

The *gestion des terriors approach* to natural resource management in francophone West Africa can be extended and modified to better satisfy the principles of co-management, subsidiarity and over-lapping rights. Co-management implies a true partnership between civil organizations and government agencies; subsidiarity implies that different types of land use will be managed by different levels of authority; over-lapping rights implies that the multiple uses of cropland, fallow land and woodland will be recognized and protected.

*Non-governmental or collective-action organizations* can be effective in bridging the gaps between public sector objectives and private sector actions.

*Recognition of the incentives and disincentives of different stakeholders* is important for understanding how the different agencies involved in natural resource management – farmers, livestock owners, community groups, non-governmental organizations, government agencies – might support or impede improved natural resource management. The international interests in carbon offsets creates another set of stakeholders (Arnould, 1990; Grimble et al., 1995).

**Table 7 A classification of trade-offs and conflicts.** (*Source: Grimble et al., 1995, Table 2*)

| Level | Trade-off | Conflicts of interest |
|---|---|---|
| Macro-macro | between policy objectives | between national institutions or depts. |
| Macro-micro | between national and local interest | between national institutions and local people |
| Micro-macro | between internal actions and externalities | between local people and larger society |
| Micro-micro | on-farm resource allocation | between different sets of local people |

A *long and adaptable time horizon* must be built into project plans. As discussed above, rural Africans necessarily react very cautiously and often slowly to new innovations. Community groups may move even more slowly due to the transaction costs of groups and their often political nature. Besides, the variable climate conditions in the African drylands makes it difficult to specify particular environmental goals that must be met at particular times.

*Programmes for rural development and technology dissemination* should recognize that farmers are very heterogeneous in their resources, constraints and objectives and offer a menu of proven, low-risk, technological alternatives (e.g. trees, forages, cultivation devices, water retention devices). Programmes should allow for flexibility and adaptation to the various environmental conditions that may prevail.

### 4.2 Priority sources and sinks

The sources and sinks that should receive priority attention in carbon off-set programmes for the African drylands should fulfill the following criteria: (i) there are clear linkages between human action and increased carbon storage or reduced carbon emissions; (ii) they are important at the national or international levels and very important in particular localities: pilot programmes could focus on such localities; and (iii) it is possible to design incentive-compatible mechanisms to achieve measurable carbon improvements.

With those criteria, and the discussion presented in sections 2 and 3, I propose the following priorities:

(a) Establish and manage state and local forests for the dual objectives of sustainable wood production and carbon storage (e.g. Property rights and responsibility assigned following subsidiarity; trees planted using labour-intensive public works; use and sell products in response to local needs and market signals; distribute revenue to co-owners following a variant of the CAMPFIRE model.)

(b) Support development of markets and market infrastructure for fuelwood, other tree products and livestock feeds. The relative scarcities of different commodities should be reflected in the prices paid to the producers of those commodities. Community groups should be able to exert ownership rights over resources that often are *de jure* state property but *de facto* open access. Such an approach has been successful in Niger where 40 villages have been granted ownership rights to wood resources and implemented plans for managing those resources (Mahamane et al., 1995).

(c) Promote policies and programmes for more active management of fire (e.g. labour-intensive public works for fire breaks, co-management of fire setting.)

(d) Stop land clearing for agriculture that replaces other land-use practices that are economically viable and ecologically sustainable. Examples may be mechanized rainfed agriculture replacing gum arabic production in Sudan or small-scale farming replacing wildlife habitat in some areas of Zimbabwe, Tanzania, Kenya and Botswana.

(e) Identify priorities for promotion of different types of agro-forestry, planted forages, water retention devices and leguminous crops using models that incorporate agro-ecological (rainfall, soils, relief) and socio-economic factors (prices, availability of substitutes). Identify constraints and opportunities for technique adoption in those areas and offer farmers a menu of alternatives that fit within those constraints and opportunities.

### 4.3 Research needs

Carbon off-set programmes could also focus on the public goods of improved knowledge from research. Research on any of the following topics should be conducted by multi-disciplinary teams:

(a) complementarity between global objectives of biological diversity and increased carbon storage -- this information would help determine the potential of government forests and conservation reserves for involvement in carbon offset programmes;

(b) studies of different types of rangelands (by soil type, rainfall, condition) to determine the compatibility of range management for livestock production versus land management for carbon storage;

(c) study farmers' perceptions of the advantages and disadvantages of fire;

(d) conduct experiments with trees, perennial grasses and leguminous crops such as pigeon peas and cowpeas to determine their potential contribution to more sustainable land use and carbon storage; and

(e) identify the factors that constrain the adoption of alternative energy sources such as solar and wind energy and fuelwood / charcoal conservation methods such as improved stoves.

### 4.4 Final words

To re-iterate, carbon offset programmes must be incentive compatible to be effective. They must be consistent with the trends toward market liberalization and subsidiarity to be sustainable. They must build up from the actions of individual farmers rather than the words of bureaucrats. To borrow a phrase from the environmental movement: the designers of carbon offset programmes will have to 'think globally and act locally'.

### Acknowledgements

The author acknowledges the financial support of UNEP and the helpful comments by Peter de Leeuw, Robin Reid, M.A. Mohammed-Saleem and Mark Stafford-Smith on an earlier draft of the paper. Russ Kruska performed the geographic analysis necessary to estimate the populations in the African drylands. Opinions and errors are the responsibility of the author.

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# Integrated approaches to assessing sequestration opportunities for carbon in rangelands

**Mark Stafford Smith, Dennis Ojima and John Carter**

Synopsis

This paper outlines an iterative process for determining whether the productivity of rangelands can be manipulated to achieve better carbon storage at a regional and project scale. Possible interventions are evaluated against technical, socio-economic and cultural constraints. The various steps are elaborated and illustrated with case studies from Australia, which show that different manipulations in a single landscape type may have quite different feasibility and value.

Key Points

1.The question of whether a significant amount of carbon can be captured in drylands through combating the degradation of those lands cannot, in practice, be answered by a simple assessment of the maximum storage potential of those lands. This potential is affected by the area currently degraded or at risk, management goals for different drylands, the socio-economic and cultural feasibility of possible carbon sequestering activities, and the sustainability of investment in such activities. These factors must be evaluated at a regional to local level.

2. There are three stages to determining whether sequestration opportunities in drylands are real. First, the broad potential at a global scale must be demonstrated. Second, a realistic approach to assessing opportunities at the regional level must be developed and applied. Third, mechanisms must be defined for linking donors or investors in sequestration with credible and sustainable projects by the land managers concerned. This paper concentrates on the second stage.

3. Regional projects for carbon sequestration in drylands could concentrate on the maintenance or preservation of existing carbon stocks (ie. Avoiding degradation), or on the capture of new stocks (ie. reversing degradation or otherwise altering a system to hold more carbon).

4. A comprehensive approach to assessing the regional feasibility of storing carbon must integrate three types of factors - biophysical options, socioeconomic opportunities and cultural constraints. An efficient, iterative analysis first requires a rough summary of possible ecosystem states in a region. The potential for transitions between these to increase or decrease carbon stocks is then compared with the biophysical ease and the socio-economic implications of each transition. For transitions which are economically possible, the cultural dimensions of resistance to change are then assessed. A first pass through this procedure can be done as a desktop study very easily, quickly narrowing down the alternatives. A more sophisticated analysis can then be applied to promising options in subsequent iterations of the process, drawing on an increasing level of participation by local managers who might be involved in projects. The paper gives examples of this process in Australia.

5. In many parts of the world, sufficient information already exists to make a first pass through this procedure. Such an exercise could be carried out

quickly and be extremely informative in terms of identifying regions, ecosystems and cultures where different project approaches might be applicable.

Key words: Australia, *Acacia,* management strategies, socio-economic, greenhouse gas, cost/benefit analysis

## INTRODUCTION

There is a growing interest in using drylands to sequester carbon and thereby reduce or offset the releases of greenhouse active gases into the atmosphere (Squires and Glenn, this volume). Management of drylands which sequesters carbon will not necessarily coincide with management of land degradation, but where such coincidence can be engineered in a socially acceptable context, there could be valuable synergistic reasons for action.

In an associated paper (Ojima *et al.,* 1995), we examine how simulation can be used for the generalised assessment of sequestration opportunities in relation to land management. Here we are more concerned with the process of implementing an assessment scheme which might identify opportunities at a sufficiently local level that they can be directly related to land use and land users. We argue that there are at least three stages in determining whether sequestration opportunities in drylands are real. First, we need to be convinced at a global scale that there is a sufficiently broad potential that it is worthwhile pursuing more detailed options; many other contributions to this volume are concerned with this assessment. If the answer to this first question is yes, though, we then need to have two further strategies in place. One is a realistic approach to assessing opportunities at the regional to local level at which they would eventually be implemented, and the other is a mechanism for linking donors or investors in sequestration with the land managers concerned. Other papers in this volume (eg. Trexler, Kinsman) deal with the latter of these issues; we focus on the former.

## A STRATEGY FOR ASSESSING THE VALUE OF MANAGEMENT

The value of directed management of drylands in sequestering carbon could come from one of two general effects.

· First, management might be instrumental in avoiding changes which are deleterious to carbon storage; we will term this '*maintenance*', and it depends on understanding the 'vulnerability' of the system to change.

· Second, management might be directed at actively sequestering more carbon than at present; we will term this '*capture*', and it depends on understanding how to change system states and what the value of changes would be.

Since these changes will often (but not necessarily) correspond to recovering from past degradation, both *maintenance* and *capture* may be related to managing current or past desertification. However, *capture* may also involve novel ecosystem manipulations, so that the more general term is useful. For many purposes, the two forms of activity depend on similar knowledge. However, the distinction is particularly important in assessing the social acceptability of novel ecosystem states.

**Fig.1 An iterative framework of steps for formally assessing mitigation opportunities linking carbon storage and degradation alleviation.**

**Assessing Mitigation Opportunities Related to Degradation**

| *Maintenance* | *Capture* |
|---|---|
| Avoid C losses | Seek C gains |

**Identify biophysical options**

- What different states of each ecosystem can exist?
- What levels of C storage/fluxes occur in different states?
- What biological conditions control changes between states?

| Maintenance | Capture |
|---|---|
| Identify the biological vulnerability of the system to change | Identify whether positive changes of state are biologically feasible |

**Identify socioeconomic opportunities**

- What is the (rational) cost/benefit of the management needed?
- What is the extent of regional occurrence of the ecosystem states?
- What are the maximum regional benefits?

| Maintenance | Capture |
|---|---|
| Identify the economic costs/benefits of avoiding degradation | Identify the cost of creating different changes |

**Identify cultural constraints**

- Is the management acceptable and socially feasible?
- Is the managed state socially desirable?
- What are the net realistic regional benefits?

| Maintenance | Capture |
|---|---|
| Identify the social pressures contributing to degradation | Determine whether the new state is socially useful, and its benefits |
| Determine whether financial inputs will deter damaging management | Determine whether there are cultural constraints on the required management |

**Compare with alternative investments**

- What is the regional/national context?
- What option has the highest net realistic benefits?

**Table 1. General questions related to mitigation opportunities.**

| | MITIGATION OPPORTUNITIES RELATED TO DEGRADATION | |
|---|---|---|
| *Issues* | ***Maintenance*** Avoid losses of C | ***Capture*** Seek gains of C |
| Physical | Do changes in system state cause loss of C? | Is there a system change resulting in C sequestration? |
| Ecological | How vulnerable to change is the system? Are changes irreversible? | Is the system change biologically achievable? |
| Socioeconomic | Are there pressures for management changes which will degrade the current system? | What cost is there to changing the system, and is the outcome acceptable socially? |
| Cultural | Is the current management culturally acceptable? | Is the management needed for the change culturally feasible? |

The question of whether a significant amount of carbon can be captured in drylands through combating the degradation of those lands cannot be answered by a simple assessment of the maximum potential storage capacity of those lands per unit area. This potential interacts with the areas currently degraded, management goals for different drylands, the social feasibility of possible sequestering activities and the sustainability of investment in such activities (Table 1). Figure 1 summarises the sequence of actions that we regard as the minimum set necessary to assess the value of manipulating drylands for the purpose of combating climate change and desertification. We expand on these steps below, but first we must comment on two matters - order and iteration. Some people would probably challenge the suggestion that the order of assessment should be biophysical, then socio-economic and only then cultural and political; others, of course, would challenge the reverse order. We specifically intend this sequence to be used iteratively in order to obviate such concerns. A rapid pass through the sequence with minimal data will allow all three categories of constraint to be weighted equally, with greater depth of understanding achieved with more information subsequently. Thus although we talk of a single sequence, an iterative approach is fundamental to its integrity, as well as leading to greater efficiency of use of scarce information since some options can be quickly screened out without further work.

Finally, before embarking on any detailed analyses it is useful to identify whether activities are generally in the maintenance or capture categories since this often puts a very different slant on how options are viewed socially and culturally.

In general a more sanguine view is taken of the maintenance of the *status quo*, or the rehabilitation of a past state, than of a novel manipulation of the system. In many steps, however, we can discuss the two together, just noting a few key differences of emphasis.

## Step 1. Identify biophysical constraints and opportunities

In this step, without initially worrying about cost or social constraints, we wish to identify what ecosystem changes are possible and what their implications are for carbon storage. The state and transition model of Westoby *et al.* (1989) is a useful foundation concept for this.

### *Step 1a . Identify different ecosystem states*

First, we must know what system states and system changes are possible in a region. In a first iteration, this should focus on extensive or highly productive systems, and note changes which have occurred in the past or which are in progress now. In northern Australia, for example, most of the major vegetation communities have been classified to their key states, with varying levels of understanding as to the rate and ease of transitions between these states (see papers in *Tropical Grasslands* volume 28, *cf.* Jones and Burrows 1994). These states are usually defined in terms of ecosystem state, but here we may adopt a broader concept which incorporates different types of management and mixtures of management; for example, these may be different stocking rates (with different economic outcomes) or the return or otherwise of manure to a small-holder cropping system (*cf.* Ojima et al., 1995). Thus the 'states' should be regarded as any realistic landuse/ecosystem state combination.

The different states also do not need to be defined in terms of 'degradation'. However, it will eventually be necessary to coordinate these results with estimates of the areal extent of different states, such as those provided by the World Atlas of Desertification (UNEP 1992), so standardised definitions may be useful. Often though it will eventuate that considerably greater resolution in ecosystem and condition definition is needed than these definitions. At this stage, there is no value in debating the social implications of these definitions, providing the data for the next two steps are available.

*Step 1b. Characterise basic carbon pools and fluxes*

To make any assessment of dryland storage capacity for carbon, it is necessary to have a reasonable knowledge of the likely above and below-ground fluxes and pools of carbon in the range of different soil types and climates represented in the region, in their various states. The level of accuracy achieved does not need to be very high for a first iteration through these steps, since many options may be readily ruled out on economic and social grounds. Increasingly there is a global database of sufficient initial accuracy for this purpose, although more work is needed to characterise the various states. This data is primarily sought on a point basis - that is, carbon storage and fluxes per unit area. However, sometimes the loss of storage in one patch may be a complete loss; in other cases spatial redistribution by water and wind may result in some or all of that storage capacity being transferred to another part of the landscape (*cf.* Pickup 1985). Thus these point estimates need to allow for possible landscape-scale implications.

*Step 1c. Determine what controls changes between states*

Next we need to know how to cause or prevent transitions between states. Westoby *et al.* (1989) provide a standardised way of describing these transitions, although their concept does not readily handle the landscape links mentioned above (Stafford Smith and Pickup, 1993). Given the suite of possible states and transitions, we can classify the transitions into ones causing loss, no change or gain in carbon storage capacity.

In the case of *maintenance*, we are principally concerned with avoiding change and hence understanding the vulnerability of the system to change to a state with less storage capacity. Examples are avoiding the loss of soil carbon through erosion or loss of standing biomass through tree clearing. Other transitions such as changes in species composition may cause no carbon storage loss; this would be acceptable from the point of view of carbon storage, but perhaps undesirable for management.

For *capture*, the question is how much investment is needed to cause a positive change. Some transitions may be biologically impossible on reasonable timescales, especially if there has previously been substantial loss of soil. However, for most transitions sufficient investment can overcome these constraints, if necessary by importing nutrients or carrying out expensive mechanical works. Whether these are economically-justified depends on the production response, including the value for carbon storage.

Given that knowledge is not perfect, it is also important to ascribe a level of riskiness to the possible benefits - this could be in terms of a mean and standard deviation. For example, for benefits estimated at (say) 1000 kg C $ha^{-1}$, there is a great difference between a standard deviation on the estimate of 10 kg C $ha^{-1}$ and 1000 kg C $ha^{-1}$.

Step 2. Identify socioeconomic constraints and opportunities

Step 1 provides information on carbon pools and fluxes for ecosystems in different states, and biological feasibility of transitions. Its principal outcome is to identify which transitions may be worth pursuing further at all. The next step is to assess the economics of causing or preventing particular transitions, firstly from the point of view of the landuse involved, and secondly at an aggregated regional scale at which political intervention might be worthwhile.

*Step 2a. Cost/benefit analysis of management for storage*

This step targets the land user and their enterprise or household scale. The principal question is whether states with lower carbon storage can be converted into states with higher storage, or whether states with higher carbon storage can be prevented from conversion to lower storage states. There will almost always be some level of investment which will permit a conversion to be made or prevented. The issue is how much this impinges on economic productivity, both in terms of the management needed to cause or prevent the transition, and in terms of the productivity of the target state. We consider this firstly from an economically rational point of view - on economic grounds alone, what external investment would be needed to make the activity worthwhile, presuming that people will then rationally consider this choice compared to current management?

On a first iteration this step can be very approximate. Eventually, though, it needs a reasonably sophisticated analysis of productivity which is appropriate to the land use and user concerned (ie. very different for a subsistence farmer and a commercial rancher). Since most rangelands operate under variable climate and external market risks, a dynamic assessment will always be preferable. We give an example later.

We are now in a position to classify the transitions according economic and biological desirability. The four broad areas of Fig.2 have very different implications for local action, as indicated. In particular, cases in area 1 should require very little support, while those in area 4 will be very difficult to manage. For changes in state in areas 2 and 3, the first action should be to seek other changes in the same ecosystem which are more positive for economic benefit and, if they exist, promote these. In short, if the investment needed is exceeded by the financial losses to production, then the option will be dubious in the absence of substantial other inputs such as carbon credit payments.

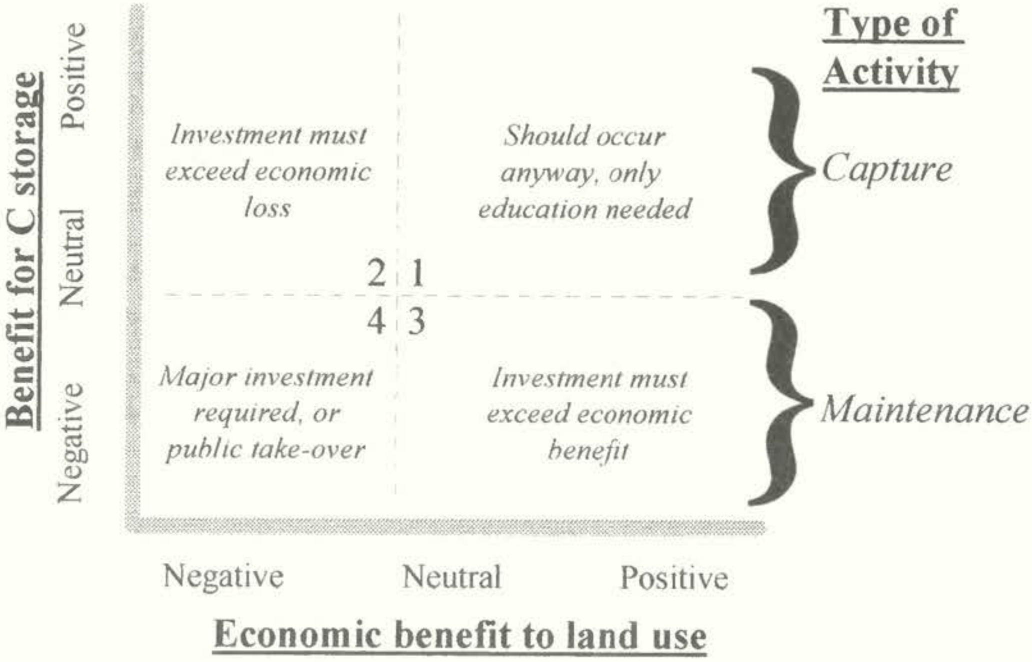
*Step 2b. Regional occurrence of different system states*

Given an assessment of what changes in storage might be possible at what cost and with what consequences on a per unit area basis, the next step is to assess what potential this actually represents in different regions. To do this, we require a knowledge of the spatial extent of land of different types in different states by region. Preferably this knowledge should be spatially explicit (eg. on a GIS), since its pattern of occurrence (eg. patchiness and distribution) locally may affect the ease and cost of management. Rough assessments early on may help to rapidly narrow down the range of options worth considering. Any systems which are of minor occurrence can be ignored unless extremely productive, although they may experience some synergistic benefits from general changes in management such as lowering stocking rates, without requiring explicitly-targeted measures.

*Step 2c. Calculate maximum regional benefits*

Combining the previous steps provides us with a prediction of the maximum level of carbon storage that may be possible in different regions, as well as its cost and riskiness in financial terms. If this is modest then further detail may not be needed. Meanwhile the first iteration will provide an indication between regions of where future analysis effort is most likely to be profitably spent.

**Fig.2 Possible categories into which changes in ecosystem state may fall, given the assessment only of benefits to carbon storage and land use.**




Step 3. Identify cultural opportunities and constraints

Step 2 dealt entirely with the local or enterprise-level economic constraints, and their regional implications. Finally it is vital to assess the cultural implications. These may be local or society-wide.

*Step 3a. Acceptability and feasibility of management*

It is quite possible to have management interventions or outcomes which are economically rational at a local level but which are culturally unacceptable, at least without a long period of acclimation. For example, if the manipulation involved removing all stock from a landscape, a pastoral society might be very unwilling to agree even in exchange for quite large subsidies. Other management activities might depend on a certain level of education in managers which requires many years of lead time to develop. Furthermore, there is diversity among land users - even with the perfect option it is unlikely that there would be total adoption immediately, so that there will be lags in uptake and inaccessible reservoirs of non-adopters to reduce the maximum possible regional benefit by a substantial margin. In general commercial land uses may be more amenable to land use change based on economic imperatives, but even here there may be substantial social disruption. The same is possibly more true of subsistence land uses.

*Step 3b. Cultural desirability of managed state*

Even when the local land users may be quite happy with a particular management activity and its implications, society at large may find it culturally and politically unacceptable. People are becoming increasingly concerned with the conservation of biological diversity and with 'naturalness'. Any manipulative management can be seen as out of place from this point of view, but the homogenisation of species, the introduction of exotic species, or the manipulation of species-rich areas are all likely to be viewed with scepticism. Since many suggested actions, especially for *capture*, involve all three of these elements, it would not be surprising to encounter cultural resistance. This also highlights how important it is to distinguish *capture* activities which are in fact rehabilitation from those which are novel manipulations. The latter are more likely to be unacceptable. We may need to be concerned with principles such as reversability and controllability in these circumstances. For example, one might introduce a tree species which cannot establish without human intervention, thereby ensuring that the impact can be reversed if so desired at some time in the future. *Maintenance* should usually be more acceptable.

There are other, more indirect, ways in which cultural and political constraints may arise. Ojima *et al.* (1995) describe an example in Ethiopia where it would be very beneficial to have manure returned to the ecosystem instead of using it for fuel; however, this cannot happen until the villagers have alternative forms of power. In Australia, deliberate political policies such as those of closer settlement and drought subsidies have often distorted expectations among land users in ways which are counter-productive in the long-term. Thus there may be an extensive set of issues to consider here.

*Step 3c. Net realistic regional benefits*

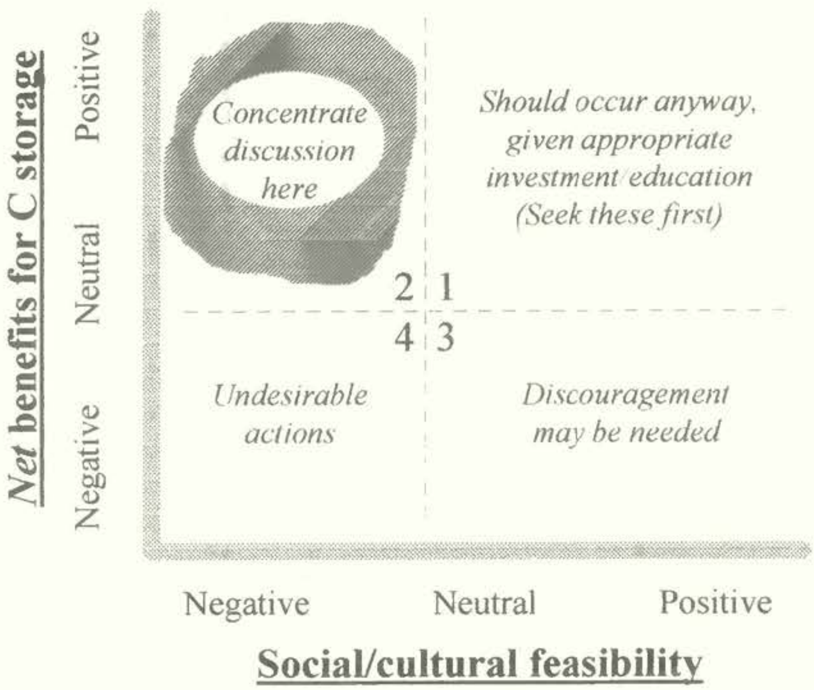
We can combine Steps 1 and 2 (the two axes of Fig.2) into one, assuming that some acceptable economic value is assigned to the carbon storage change. Further insights may be gained by plotting this result against cultural feasibility (Fig.3), an axis which is less amenable to economic conversion. In this, activities in area 1 should occur with minimal intervention, but will obtain big returns from a small amount of encouragement and awareness-raising. Area 4 should be readily avoided. In area 3, discouragement and education may be needed; in fact many government policies in the past have created a cultural will to carry out management which has been non-sustainable, such as drought subsidies (Tapp *et al.* 1995). Here the removal of inappropriate disincentives by government may be all that is needed. It is in area 2 where disputes are most likely to arise, however, between those who believe in engineering systems and others who support the primacy of human culture. Different societies will endorse more or less enforcement here. Where less is acceptable, a generation or more may have to pass before the activities become culturally acceptable.

It is hard to convert these issues precisely to financial values, but an assessment which combines non-adoption rates, management complexity, and local and national cultural feasibility allows a more realistic net regional cost-benefit to be calculated.

Step 4. Compare with alternative investments, and iterate

Given all the preceding calculations, it is still not good enough to simply ask whether the net benefit is positive. For efficiency's sake, it should also be *better* than other options. These options may be other ecosystems or natural resource management, or they might be direct investment in education, welfare or empowerment of women, for example (these issues are discussed further in other chapters in this volume - Trexler, Kinsman, Swallow, etc). It is also necessary to consider whether there is an investor who is prepared to support the management activities (if they do not fall into area 1 of Fig.3), often because of having some local or regional interest which can be filled. Once this task has been carried out, roughly on the first iteration, further iterations can provide more details on options which look hopeful.

**Fig.3 Possible categories into which changes in ecosystem state may fall, given net benefits for carbon storage after an appropriate level of investment to overcome any economic losses caused to land use (ie. collapsing the axes of Fig.2 into one), but allowing also for social and cultural constraints.**




One question which may be considered in the context of Fig.3 is, what form of investment is most appropriate for the different areas of the graph? Private investors presumably want low risk, high return investments, which probably fall into the graph's area 1. Yet these may represent activities where public extension services or the natural course of time will be the major cause of change. One might argue that these should remain in the public domain, therefore. On the other hand, private investors will be very unwilling to invest in the more risky options, which also presumably will end up in the public domain. It might be reasonable to require that acceptable private investments must fall into area 2 in Fig.3 and areas 2 or 3 of Fig.2, but this will require further debate.

Finally it is worth noting that improved management in a region over several ecosystems may have synergistic effects in terms of the investment necessary to achieve change. The level of education or infrastructure required for one system may trigger spontaneous changes in management elsewhere. Stocking rates on average are being reduced across the Australian rangelands for a variety of reasons, for example, and this may already be contributing to carbon uptake. Such regional synergies may be hard to assess, but may make investments all the more attractive.

### Australian Case Studies

To follow through the procedure laid out earlier and consider a variety of system changes, we will examine some rangelands regions of Australia. Although cases in less developed countries may be more important in the long run, Australian examples serve to illustrate data needs and responses where there is relatively good information.

The examples come from Queensland, where the best GIS database exists, but have been chosen to be representative of some of the issues which face inland Australia. Summaries of the pasture types and condition of the northern half of Australia may be found in Tothill and Gillies (1992), where some seventy pasture types are identified. At the level of resolution required here, however, these could be considerably reduced.

**Case 1: northern Mitchell grass plains, western Queensland, Australia.**

The northern Mitchell grass plains are based on black, cracking clays, and occupy an area of 0.22 m $km^2$. Comparable soil types cover about 10% (0.55 m $km^2$) of arid and semi-arid Australia, and are among the most productive of rangelands ecosystems for pastoral production. The northern Mitchell grasslands region in Queensland carries about 1.7 m average beef equivalents at a mean stocking rate of 10-15 ha $head^{-1}$ (Tothill and Gillies 1992) on about 350 pastoral properties, and produced a total value of agricultural commodities of about $160m in 1994. Production has been dominated by sheep but with low wool prices in recent years there is a steady move towards beef cattle. By comparison with the second case study, the region is regarded as relatively stable financially (Newman 1994), but some major ecosystem changes are occurring. We will now take two iterations through the procedure laid out above to exemplify how simple information allows us to narrow the field of options, and then how increasing information can clarify the results.

*Iteration 1.*

The northern Mitchell grasslands have three predominant biological states (Fig.4 - simplified from McArthur *et al.* [1994] by omitting a number of unstable states). Changes between States C-I and C-II are relatively reversible, with a decline in reliability of production in State C-II. However, these lands are rapidly being invaded by an exotic shrub species, *Acacia nilotica*, which can greatly reduce production (Carter 1994) but probably increase carbon storage. Tables 2 and 3 contain a low resolution assessment of the value of the different states to carbon storage, the factors which drive changes between states, and the likely production implications.

Fig.4: Key states and transition for the northern Mitchell grasslands in western Queensland, simplified from McArthur *et al.* (1994).

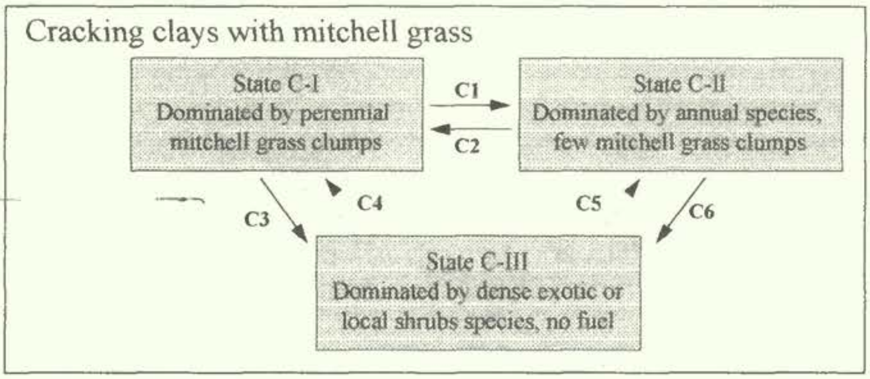


**Table 2: First assessment of key characteristics of each simplified state of northern Mitchell grasslands in western Queensland.**

| State | Extent in region | Carbon storage | Productivity |
|---|---|---|---|
| C-I | Moderate | Moderate | Moderate |
| C-II | Moderate | Lower | Slightly higher than C-I but less stable over time |
| C-III | Rapidly increasing | Higher, mainly as Acacias which fix N | Low |

**Table 3: Key characteristics of each transition for northern Mitchell grasslands in western Queensland.**

| Transition | Desirability for C storage | Desirability for production | Conditions for occurrence |
|---|---|---|---|
| C1 | + | - | Heavy grazing and droughts |
| C2 | - | + | Lighter grazing and summer rains |
| C3 | ++ | -- | Seed sources, grazing and no fire |
| C4 | -- | ++ | Very difficult without herbicide or mechanical intervention, then fire |
| C5 | -- | ++ | Very difficult without herbicide or mechanical intervention, then fire |
| C6 | ++ | -- | Seed sources, grazing and no fire |

The conversion C1 has often been caused by a lack of appreciation of the value of stable production; today, reduced stocking rates and an increase in awareness of land conservation values is already encouraging some pastoralists to manage for the transition C2. However, there are also problems with unmanaged herbivores, particularly kangaroos, in the region. Constraints on the control of kangaroos are a perceived disincentive to lower stocking rates, since the kangaroos simply use the forage instead of sheep or cattle.

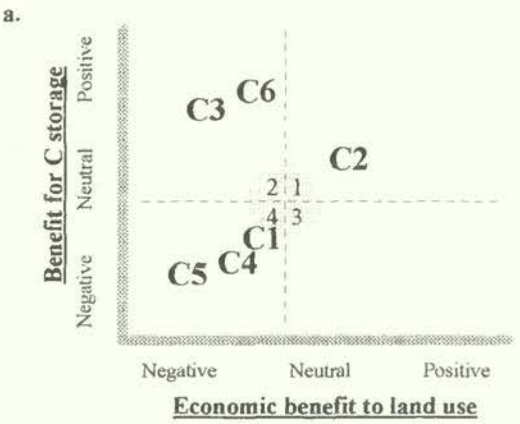
Pastoralists initially viewed changes into State C-III positively since *Ac.nilotica* fixes nitrogen and provides a source of extra protein in dry years, as well as providing shade around water where it was first planted. However, as it has become evident that it is very difficult to maintain a low density of the shrub, it has become difficult to muster stock and the invasion is increasingly viewed negatively. Furthermore, although awareness is low in the broader Australian society, *Ac.nilotica* is regarded as an exotic weed, so that deliberate attempts to convert the grasslands into a woodland with this species would need to demonstrate large benefits to be acceptable.

This simple assessment of the three Steps of the assessment framework allows us to plot the likely implications of different changes on the assessment graphs of Figs.2 and 3. Transition C2 ought to be beneficial for both storage and economics, is reasonably easily achieved and has no great cultural constraints. Investment in awareness along the lines of existing extension activities, perhaps enhanced by indicating that managers are be supporting the nation's carbon storage, should be sufficient here. On the other hand, transitions C3 and C6 fall in the realm of potential conflicts of interests, and are therefore worth examining more closely in a second iteration.

*Iteration 2.*

A second iteration must serve to bring greater *resolution* to the problem, more *precise data* on the values of different states, a better *economic analysis* and further insight into cultural issues. A closer examination indicates that State C-III may develop in different ways. The ideal utilisation rates for State C-I are around 20% of the longterm mean annual forage production (Partridge 1992). As there is increasing invasion by *Ac.nilotica*, the shrub competes with the forage layer and reduces forage productivity. Managers may respond to this in a number of ways, which may be characterised as lying along the following continuum; at one extreme, the reducing forage production is recognised and stocking rates lowered to maintain approximately the same 20% utilisation level; at the other extreme, the manager attempts to continue to run the same numbers of stock, resulting in utilisation rates which are close to 100%. *Ac.nilotica* at its highest density approximately halves the forage production possible in the perennial grassland (Burrows *et al.* 1990). In the former case, stocking levels must therefore approximately halve. In the latter case, stock manage in good years, but rapidly run out of feed in average to poor years, so that high stock deaths and low reproduction and wool growth rates are common. Some of the latter effect is offset by the increased protein availability from the leaves and pods of the shrub in dry times. In practice, of course, most pastoralists adopt a strategy somewhere between these extremes, but we will consider two new sub-states - one high stocking, C-III(high), and the other lower stocking, C-III(low).

**Fig 5: First iteration simplified assessment of (a) the carbon storage benefits and economic costs and (b) the net carbon storage benefits and cultural feasibility of different ecosystem changes in the northern Mitchell grasslands, Queensland (*cf.* Figs. 2 and 3).**



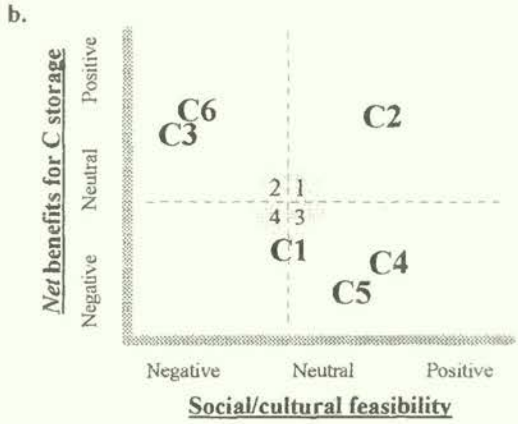




**Table 4: estimates of soil organic carbon from two Sides (A and B) of fenceline contrast near Julia Creek, Queensland (living roots mainly excluded)**

| | Site A | Site $B_{tree}$ | Site $B_{open}$ |
|---|---|---|---|
| Mean surface soil organic carbon (%) | 0.42 (±0.045) | 0.44 (±0.055) | 0.36 (±0.056) |
| Surface organic carbon (tonne/ha)[a] | 4.2 | 4.4 | 3.6 |
| % area of Side | 100 | 30 | 70 |
| OC on Side (tonne/ha) | 4.2 | 3.84 | |

[a] the bulk density of this self-mulching topsoil is close to 1.0.

With this greater resolution of the problem, we can examine further data. Organic carbon was measured at five adjacent sites in the northern Mitchell grasslands near Julia Creek (J.Carter unpubl.data.). Side A of the fenceline was in state C-I with about 20% long-term grazing utilisation; the other side of the fence, Side B, has been invaded by *Acacia nilotica* over the past 16 years and is now believed to be close to stable in state C-III(high); Side B was stratified into areas under tree canopies (site $B_{tree}$) and in between (site $B_{open}$). The net forage productivity of Side B is less than half of Side A, yet the pastoralist has attempted to maintain similar stock numbers to the other side, resulting in considerable overgrazing. From each of five sites in each category, five soil samples (0-10cm, each bulked from 20 sub-samples) were analysed for organic carbon (Walkely and Black 1934), and estimates of above ground vegetative biomass were made. These estimates all have large error terms associated with them, but serve to illustrate the approach needed.

From these data, it is possible to estimate the carbon storage possibilities of each state more quantitatively (Table 5). Additionally, Queensland's Climate Impact and Spatial Systems centre has detailed maps of the extent of the ecosystem, with resolution down to 5 and 1 km grid cells; for most of the region, there is some information about current shrub cover, although this is still being completed. Although about 7% of the area is already infested, no single property has yet more than about a third of its area densely shrubbed. It is thus possible to estimate the regional benefit of management.

**Table 5: estimates of total carbon storage in States C-I and C-III of northern Mitchell grasslands, and of fluxes during transitions, given different grazing strategies.**

| | State C-I (20% utilisation) | State C-III(high) (~100% utilisation) | State C-III(low) (20% utilisation)[a] |
|---|---|---|---|
| **Soil Organic Carbon** (non-living - from Table 1) | 4.2 | 3.84 | 4.26[b] |
| **Grass/annual biomass** (t/ha)<br>- aboveground | 2 (±1) | 0.4 (±0.2) | 1 (±0.5) |
| - tussocks & belowground | 29 (±9) | 0.4 (±0.2) | 14.5 (±4.5) |
| **Tree biomass**[c] (t/ha)<br>- aboveground | – | 15 – 25 (±15) | 12 – 20 (±12) |
| - belowground | – | 6 – 10 (±3) | 5 – 8 (±3) |
| **Living biomass** (t/ha) | 31 | 21.8 – 35.8 | 32.5 – 43.5 |
| **Estimated total carbon storage**[d] (t/ha) | 18.5 | 13.9 – 20.1 | 19.2 – 24.3 |
| Change from State C-I over ca.15 years | – | -4.6 – 1.6 | 0.7 – 5.8 |
| Sequestration rate per year (t/ha/yr) | – | -0.31 – 0.11 | 0.05 – 0.39 |
| Maximum[e] regional sequestration (Mt/yr) | | -5.6 - 2.0 | 0.9 – 7.1 |

[a] assuming similar tree cover but 50% of the perennial grass cover at 20% utilisation (ie. ca. half the stock numbers)

[b] from Table 1, assuming that Side A levels of soil carbon can be maintained in inter-tree spaces

[c] assuming 2.5 $m^2$ basal area $ha^{-1}$ of trees aboveground (Carter [1994] recorded up to 3.5 $m^2$ basal area $ha^{-1}$ but we take this to be an unsustainable peak level; at 25 $m^2$ basal area $ha^{-1}$ pasture production is suppressed by 50%) reducing to 2.0 $m^2$ basal area $ha^{-1}$ when there is a healthy grass sward; we convert this to biomass as 6-10 tonnes $ha^{-1}$ per $m^2$ basal area $ha^{-1}$ (the

range depends on canopy age structure), and assume tree roots are about 40% of aboveground biomass (J.Carter unpubl.data.)

[d] assuming a mean 46% C content in living tissue and wood

[e] this assumes all managers in the region (0.22 m $km^2$) begin growing *Ac.nilotica* now on all northern Mitchell grasslands, excluding country already infested (ca.7% - Carter 1994) and another 10% of country that is inappropriate for trees; the effect persists only for about 15 years; note that these figures equate to no more than the production levels of a single large strip mine in Queensland, or transport produced 14.2 Mt C in Queensland in 1991; Australia's net emissions for all energy use was 282 Mt $yr^{-1}$, 68 Mt $yr^{-1}$ of which was for transport (NGGIC 1994)

Since the invasion of *Ac.nilotica* takes about 15 years (10-20) to reach stability, the changes in pools only persist for this long, resulting in net annual gains as indicated for this period. There are few long-term biological limitations on a change to either C-III state. However, it is not feasible to get seeds everywhere instantly, nor to ensure a germination opportunity since the appropriate rainfall conditions are quite episodic, so the rate estimate is very much a maximum even given immediate managerial support.

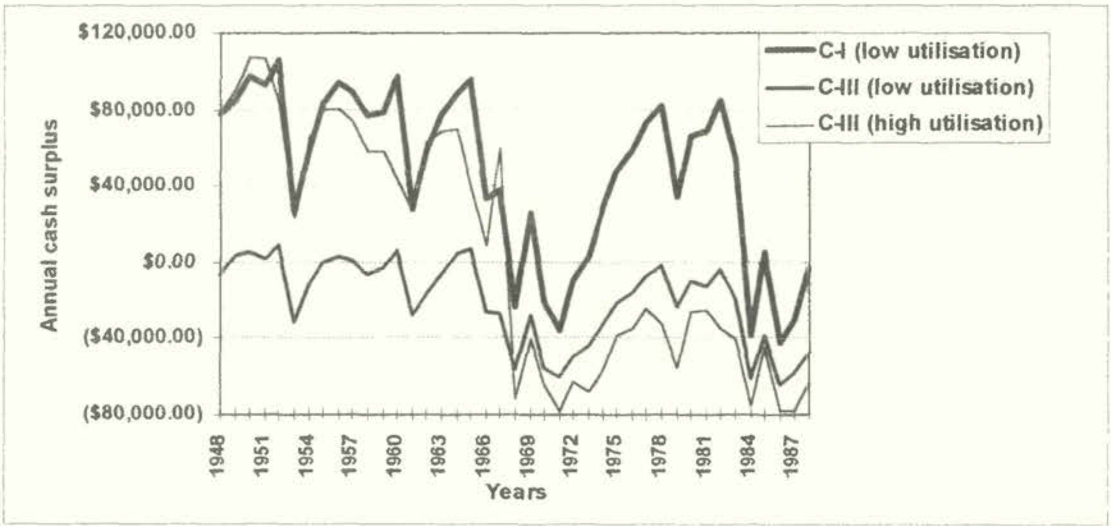
We can now examine the socio-economic implications of the states more closely. the management strategy of maintaining perennial grasses for State C-III(low) requires a cut in stocking rates to approximately half their C-I levels. Attempting to maintain high stocking levels in State C-III(high) results in an annual forage understorey, which means that frequent feeding or de-stocking is needed, or large stock losses must be accepted from time to time (the system becomes much less stable). The profitability of the three options given above can be modelled using RANGEPACK Herd-Econ (see Stafford Smith and Foran 1990 for details on this package) parameterised from economic survey work with managers (Buxton *et al.* 1995). The base projection for State C-1 is derived from a real property in the region. For State C-III(low), we manage for half the stock numbers and maintain C-I growth, mortality and birth rates; there is the appropriate reduction in variable costs, but we assume that fixed costs do not drop significantly, due to increased mustering costs and other problems brought on by dense shrubs. For State C-III(high), the same stock numbers are targeted as in C-I, but there are increased mortalities and reduced wool growth and lambing rates in all but good years; we retain the same variable and fixed costs which, given reductions in wool quality and increased mustering difficulties, is probably generous. (These rates and costs should be further refined by a visit to the region if it seemed necessary. For example, lambing rates may rise thanks to the availability of shade; however, this effect may be offset by other factors such as reduced continuity of feed). Fig.6 illustrates the predicted traces of annual cash flow for the three options over 40 years of typical climatic variability, all based on today's costs and prices, and Table 6 summarises some key statistics.

While the variability inherent in the C-III(high) strategy considerably reduces cash flow, the much lower stocking rates necessary for C-III(low) results in a $67,500 reduction in mean returns, even though a more stable number of animals is maintained. This is only a cashflow result, and would be adjusted in a full financial analysis. Note, however, that this decline considerably exceeds the misleading figure which would be obtained on the basis of a partial budget of reduced production per hectare, since it incorporates the full enterprise costs. These could be reduced a little by increased enterprise scale, for example by combining two properties into one (*cf.* Passmore and Brown 1992), but this would incur other costs; this option has not been explored here but would need to be allowed for if the option looked promising. For a property of 10,000 ha, this would represent a reduction of \$6.75 $ha^{-1}$; for the most optimistic carbon gain of 0.39 tonnes $ha^{-1}$ $yr^{-1}$, this represents about \$17 $tonne^{-1}$ C during sequestration. However, this is permanent loss of production, so this input would be smaller during the shrub invasion period of 15 years, but must then be maintained indefinitely into the future after the carbon gain has ceased.

**Table 6: Summary statistics of RANGEPACK Herd-Econ simulations of the three property management strategies (in equilibrium) for a northern Mitchell grassland sheep property when run through 40 years of realistic climatic variability (all financial values in 1995 Australian dollars).**

| | C-I | C-III(high) | C-III(low) |
|---:|---|---|---|
| **Annual cashflow (\$'000)** | 47.0 | 4.6 | -20.5 |
| **40-yr mean & sd** | 44 | 62 | 22 |
| **Animals (Dry Sheep Equivalents)** | 11900 | 8180 | 5900 |
| **40-yr mean & sd** | 2460 | 3790 | 1230 |

**Fig.6: Annual cash flow on a northern Mitchell grassland sheep property when run through 40 years of realistic climatic variability, using RANGEPACK Herd-Econ to simulate the three property management strategies.**




From these figures we can draw some more reliable conclusions from Iteration 2. Firstly, the transition to C-III(high) can be rejected. Second, for C-III(low) to be acceptable on economic grounds, a considerable and on-going investment would be needed; this is probably unrealistic unless the enterprise structure could be altered such that no further investment is needed beyond the carbon capture period (eg. much larger properties through amalgamation, or complete buy-out by the investor). The gain would be non-trivial in terms of local value (eg. offsetting a coal mine), but also not very large. Consequently, it seems unlikely that this would be perceived sufficient to overcome the distaste for exotic species in the broader Australian community. Finally, we have not discussed uptake, but at least some pastoralists in the region would not care to carry out this management regardless of the perceived benefits, due to its incompatibility with their perceived management goals. Consequently the realistic regional benefits are quite considerably smaller than the maximum estimated in Table 5.

*Summary for northern Mitchell grasslands*

The two iterations have provided considerable insights for this ecosystem. Transition C-2 is likely to be beneficial to both carbon storage and production; precise amounts could be estimated, but the management change should occur on awareness grounds alone and does not need much further analysis. Transitions C-3 and C-6 have been shown to be beneficial for carbon storage, but a further iteration of analysis indicates that this is only if the consequent ecosystem is grazed lightly, and that the economics of doing this is questionable. This change is also probably culturally unacceptable in Australia on grounds of 'naturalness' and conservation of ecosystems. When we come to compare these alternatives with others, we are likely therefore to conclude that this system is one of low priority for intervention other than normal extension recommendations for production. A little awareness education about the carbon storage response might enhance the likelihood of responses to the extension work.

If further iterations were required, it would be possible to carry out CENTURY runs to obtain a more comprehensive assessment of the biophysical results; further economic options could be explored, including property amalgamation to reduce economic losses; shires in which *Ac.nilotica* has already become established could be mapped from existing GIS sources, thus better identifying which areas and properties might participate in the management if desired; people from these areas could participate fully in further assessment and in the final decision as to whether the management changes should proceed.

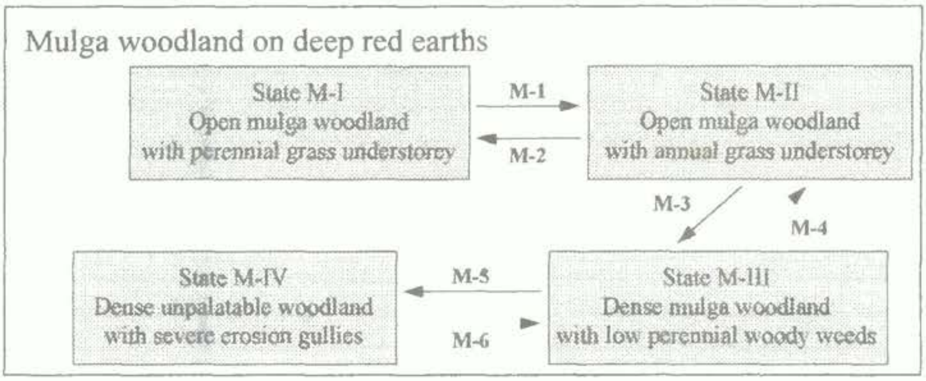
Case 2: mulga woodlands, south western Queensland, Australia

The mulga woodlands of Queensland are based on deep red earths or red sands, and occupy an area of 0.1 m $km^2$. They are characterised by an open to closed woodland of *Acacia spp*, in particular many subspecies of *Ac.aneura* or mulga after which they are named. Comparable soil and vegetation types cover nearly 30% (1.5 m $km^2$) of arid and semi-arid Australia; they are relatively poor for pastoral production, having severe phosphorus limitations, but are still the basis for some 500 pastoral properties nationally, with small sheep properties in the higher rainfall areas (to ca. 400 mm) and larger cattle enterprises in remoter and drier areas (down to ca. 250 mm). The 'hard' and 'soft' mulga region in Queensland carries about 0.29 m average beef equivalents at a mean stocking rate of 30-40 ha per head (Tothill and Gillies 1992). Many properties are suffering severe economic problems, with massive debt loads and declining biological productivity (Hoey 1994; Mills *et al.* 1989; Passmore and Brown 1992), and the region is currently the subject of a major government-supported rescue package aimed at removing some producers and enlarging the holdings of others (Hoey 1994).

*Iteration 1*

The main states and transitions experienced by Mitchell grass and mulga communities are shown in Fig.7 - these states are simplified from Jones and Burrows (1994) for the present purposes where some intervening unstable states are of no significance and are omitted. Tables 7 and 8 contain the key characters of the resulting states and transitions. In the mulga, very little country is still in state M-I; the conversion M-3 is advantageous for carbon storage, but with continuing grazing use is likely to lead to state M-IV which is not. A large part of the benefit of shrub invasion has probably already been captured *unintentionally in the region over the past century, so that* widespread deliberate conversion to state M-III is no longer a future option. For productive purposes most graziers would prefer to be promoting the return to M-II or M-I through clearing and the use of fire.

**Fig.7: Key states and transition for mulga communities in south-west Queensland, simplified from Jones and Burrows (1994).**




From these first iteration estimates, we can plot the transitions as in Fig.8. The pattern is rather different to the Mitchell grasslands example. Notably in M3 and M4 the desirability for carbon sequestration and production are strongly opposed. However, the cost of achieving M4 is high, so it is less economically feasible than producers would prefer - these losses of productivity and the costs of trying to clear the 'woody weeds' are major contributors to the economic woes of the region. In Fig.8(b), M3 is shown as marginally acceptable culturally, but this may be incorrect - at the local level this is true since land users are increasingly appreciating the lost production, but society at large is possibly moving towards effective abandonment of these lands. This is occurring very slowly, through property amalgamation (with increased areas within properties only used lightly), but could speed up in the future if new enterprises enter the region, or more leases are acquired for public purposes.

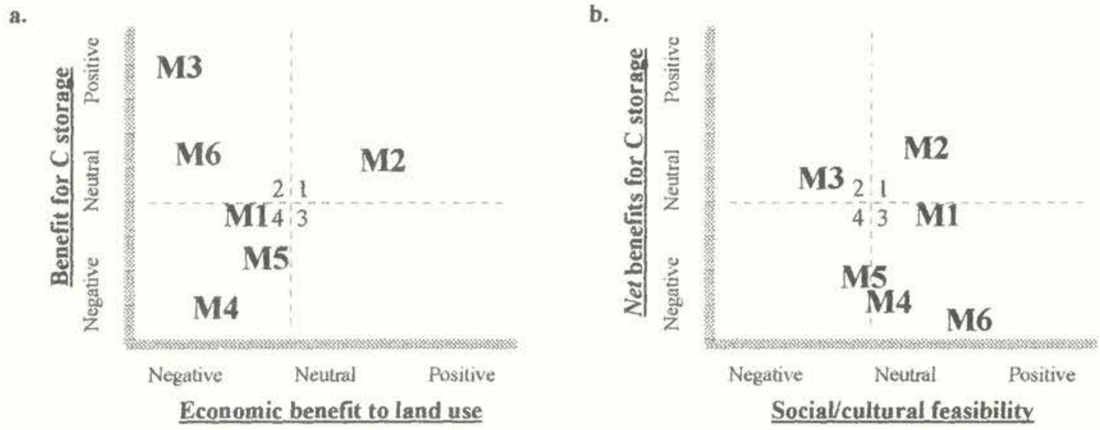
**Table 7: Key characteristics of each simplified ecosystem state for mulga communities in south-west Queensland.**

| State | Extent in region | Carbon storage | Productivity |
|---|---|---|---|
| M-I | Minor | Moderate | Moderate but low stocking rate |
| M-II | Widespread | Slightly lower | Tolerates higher stocking rate but unstable over time |
| M-III | Moderate | Higher? | Low, unstable |
| M-IV | Small | Declining | Low, unstable |

**Table 8: Key characteristics of each transition for mulga communities in south-west Queensland.**

| Transition | Desirability for C storage | Desirability for production | Conditions for occurrence |
|---|---|---|---|
| M-1 | - | -? | Overgrazing, mostly already occurred |
| M-2 | + | +? | Lighter grazing, possibly seed supplies and protection after summer rains; possibly fire |
| M-3 | + | -- | Overgrazing especially after drought; fire suppression |
| M-4 | - | ++ | Reduced or no grazing and fire, but usually very difficult without herbicide or mechanical intervention due to lack of fuel |
| M-5 | -- | -- | Overgrazing especially during drought - with the reduced productivity, overgrazing now occurs at light stocking rates |
| M-6 | + | + | Requires mechanical intervention with earth-works and removal of grazing for extended period; very expensive |

**Fig 8: First iteration simplified assessment of (a) the carbon storage benefits and economic costs and (b) the net carbon storage benefits and cultural feasibility of different ecosystem changes inulga communities in south-wesQueensland (cf. Figs. 2 and 3).**




The conversions M5 and M6 are a different matter. If grazing is maintained in the state of reduced productivity represented by state M-III, then erosion often follows (Mills *et al.*, 1989). This is negative for both production and carbon storage (although possibly not as much as it would appear on a site basis, since much material is re-deposited elsewhere). Unfortunately, though, the reverse transition M6 is extremely hard to achieve - expensive bulldozers and banks are probably required. Thus, despite its potential cultural acceptability it would seem very unlikely on economic grounds. One further factor which we have not addressed here is that these transitions also represent significant changes in diet quality, at least in drier years; this would have an additional effect on rumen methane production, which increases with declining diet quality. Since methane has a much higher greenhouse warming potential than carbon dioxide, the effect of diet quality may be important.

Despite these comments, this first iteration suggests that it would be worth seeking more information on transitions M2 (which should occur anyway, but possibly represents a modest opportunity in areal terms now), M3 and M6. By comparison with the Mitchell grass region, M3 revolves around increases in native shrubs rather than exotic ones, so is likely to be more acceptable to society than *Ac.nilotica*; the major problem here would be to overcome the mindset induced by many years of encouraging producers not to allow this transition. Investment could either be in *capture*, by actively encouraging M3, or, given that the transition has occurred widely already, in *maintenance* by paying people not to try to clear their lands again.

## CONCLUSIONS

Although we have sought to work through some case studies in some detail, the principal theses of this paper are that:

- we require a good assessment framework and procedure for identifying local/regional opportunities for carbon sequestration,

- this procedure must be strongly iterative in nature, and,
- it would be feasible to apply such a framework to all major ecosystems in a reasonable timeframe.

The complexity of the range of issues which must be considered in a proper assessment of sequestration opportunities means that it is essential to have a comprehensive approach to assessing their feasibility locally. It is also necessary then to consider other key factors which will determine how attractive a project based on some given opportunity may be for public or private investors (see Trexler, this volume), but this can be regarded as a somewhat separate question. We have proposed a framework which seems to incorporate all the critical technical issues for identifying and comparing opportunities, which include biophysical, socioeconomic and cultural/political aspects of the options.

The application of such a framework must be iterative for several reasons. First, this enables a swift superficial scan of opportunities to efficiently narrow the field to those which have some likelihood of success, without wasting time on undue details. Second, it enable large numbers of regional alternatives to be compared for modest effort. Third, it ensures that all three categories of constraints - biophysical, socioeconomic and cultural - are considered and given equal weighting early in the process, rather than risking an undue focus on any one factor. Fourth, it provides an opportunity to introduce local community participation, since a first superficial iteration can identify options which could then be discussed in greater detail with a community. Close community participation in planning is vital for eventual local ownership of a problem and its solution; however, it is desirable neither to raise expectations too high too early, nor to swamp a community with too many alternatives. An independent first iteration can therefore avoid these pitfalls while still engaging the community early in the process.

We believe that it would be a feasible desk-top study to carry out a simple, first iteration assessment of all major ecosystems in a region quite quickly. In Australia, for example, state and transition models are known for most ecosystems (perhaps 15-20 that would be worth investigation for the arid and semi-arid areas of the continent), and a modest input of expert opinion would suffice to characterise the transitions and plot them on graphs such as Figs.2 and 3. This would provide a far better basis for objective decision making than an *ad hoc* approach to perceived opportunities. Transitions with potential could then be subjected to further analysis, and also assessed against the project-based criteria suggested by Trexler (this volume). To identify a sensible suite of ecosystems for such a study, it would be valuable to have a better functional classification of drylands, as has also been called for in global change studies (Stafford Smith *et al.* 1995).

A final point is that indicators of success will be needed for at least two purposes. First, it must be possible to verify that projects are actually meeting their local objectives and are actually storing carbon. Second, society will wish to confirm that this storage is actually meeting the broader goal of reducing the carbon increase in the atmosphere in a cost effective way. The first type of indicator may also be very important in encouraging local communities to adopt proposed practices - feedback to local communities and individuals greatly helps to change behaviour. For example, the ability to produce realistic statistics along the lines of "the shire soaked up 1Mt C last year, but could do better" will help people to perceive when actions work and create pride in success. Such feedback indicators could be linked to local or regional performances, such as by comparing the flow with the outputs of local industry. For example, although this has not yet been done, it would now be possible to model and report methane releases in 'real-time' (eg. month-by-month) at a shire level in Queensland, using a GIS based model which could respond to pasture type, shire stocking rates and the current pasture quality relative to climate driven growth and decay (*cf.* Brook and Carter 1994). Innovative approaches such as this will be needed to motivate change.

ACKNOWLEDGMENTS

We are grateful to UNEP for the opportunity to put thought into these matters, and to Vic Squires for organising this opportunity. AIDAB contributed to the presentation of the paper in Nairobi. Comments by several workshop participants improved the draft.

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## Report of Working Group 2 on opportunities for carbon sequestration in drylands

Introduction

Given that drylands seem to represent a sufficient potential opportunity for carbon sequestration that further analysis is worthwhile, two steps should be undertaken to properly identify and assess specific regional opportunities:

1. a comprehensive, iterative procedure for identifying what landuse/ecosystem changes would preserve or capture carbon stocks; this procedure should be applied in a superficial fashion to major landuse/ecosystems around the world, to highlight those options which have promise. It involves identifying what managed states each landuse/ecosystem may be in, what the transitions between these states mean for carbon storage, and then assessing biophysical, socio-economic and cultural/political constraints in relation to transitions which are positive in terms in carbon storage (Stafford Smith et al, Figure.1, this volume). This analysis should be applied to a suite of systems which represent different climate, soil and social conditions (Scholes, this volume).
2. a second comprehensive procedure should be applied to promising options, to determine whether they show sufficient returns, credibility, low riskiness, and ancillary benefits to be considered by a public or private donor source (Trexler, this volume), and, if so, for what principle reason.

Future work should be commissioned to carry out a preliminary comprehensive assessment of the world's dryland systems in terms of the first procedure. In this summary, we have identified a subset of possible management options which would emerge from such an exercise and submitted them to the second procedure, as follows.

Criteria for evaluation of dryland carbon sequestration proposals

Brief description What actions are intended, and by what mechanism will they reduce climate change?

Feasibility What are the technical dimensions of the project? Is it credible

- potential realistic area involved
- change in C per unit area, over what period
- are there empirical data or pilot studies to substantiate the idea?
- directness - is the climate benefit direct or indirect?
- simplicity
- additionality - will the climate benefit be additional to what would happen anyway?
- is C the main or subsidiary reason for action?

Risks and constraints

- form/security of C being stored
- social sustainability and participation, acceptability
- sensitivity to external factors
  - climate change
  - political stability
  - economic change
  - land use pressure

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Costs and benefits - what are they?

- distribution of costs and benefits between individual, local, national and global levels
- when are the benefits accrued?
- once obtained, are the benefits secure or at risk?
- hidden costs: eg monitoring and setup costs
- \$/tC for climate purposes, and \$/ha for desertification control purposes

Ancillary costs and benefits - not directly climate or carbon related.
Examples: health, state of environment, rangeland productivity, sedimentation, biodiversity

Management of subhumid woodlands for maximum yield

Brief description Forestry management in woodlands can increase their biomass and unsustainable harvest rate. Actions include optimised silviculture (tree selection, increased canopy cover, sustainable harvest rates), fire suppression, control of illegal harvest, overgrazing and slash and burn and efficient timber milling, multiple use of forest products .

Feasibility

The potential area involved is 500 million ha in Africa, about 8 million ha globally, less about 20% currently under crop agriculture. The potential C uptake rates are 0.5 tC/ha/yr for 30 years with similar rate indefinitely through sustainable harvest of long-lived wood products or fossil-fuel displacing biofuels. Forest growth rates and soil carbon accumulation rates are partially quantified and validated in trial plots.

The benefits for carbon sequestration and the avoidance of other greenhouse gases are direct; some benefits (such as the improved efficiency of milling) depends on the efficiency gains sparing timber elsewhere. The forest management procedures are simple and tested. The gains are largely additional -- the alternate land use is degraded woodland, slash and burn or high-input agriculture. Non-additional benefits include the enhanced productivity under elevated $CO^2$.Carbon sequestration can form a major part of the justification of the improved management; it is not the only argument, but may be the one which tips the balance between agriculture and forestry.

Risks

Three-quarters of the carbon is stored as wood, one quarter as soil organic matter. Both are moderately secure (50-year half-life). The wood could be lost by fire (repeated fire for total loss) or by conversion to agriculture or by loss of effective management control of harvest rates. Soil C could be lost through conversion to agriculture

In Africa, these woodlands are mostly under communal tenure and management, with greatly varying integrity and security. There are extensive areas where community-based resource management is in place and successful. Susceptible to increased land use pressure brought about by high population growth rates.

Most of the subhumid woodlands in Africa occur in countries with a moderate stability of governance. The economic sustainability depends on the relative prices of tropical hardwoods to cash crops. Currently this is favorable, especially on very infertile soils far from markets and poorly served by transport infrastructure.

Current GCM scenarios suggest that the areas between the inter-tropical convergence zones are likely to increase moderately in precipitation, and slightly in temperature.

Costs and benefits

Carbon-related costs include the setup costs, which are predominantly in the supply of equipment for timber harvest and milling and fire suppression, and operating costs, which are labour (fire control, technical supervision), vehicle and maintenance costs. The local communities are largely outside the cash economy and therefore cannot contribute startup costs; rapid cash flow should allow them to pay operational costs indefinitely. The startup costs are estimated as \$10/ha; carbon costs over a 20 year period are therefore \$1/tC.

The benefits include C storage and reduction of methane (0.75 kgCH4/ha/yr avoided, equivalent to about 4 kgC/ha/y) and ozone-forming gases; reduction in aerosols is a disbenefit. The major non-carbon benefit is the income from woodland products: timber, fuelwood, some grazing, fruits and honey.

The monitoring costs are reasonable, since detailed stand monitoring is needed for accurate sustainable management, and therefore does not represent an additional cost. Soil carbon increases will need to be monitored at a cost of \$0.40/ha/yr. Independent verification of forest cover and timer output can be obtained from remote sensing and sales invoices.

Ancilliary benefits

1. Improved quality of life through job provision and infrastructural development.
2. Biodiversity conservation by habitat preservation and control of poaching and illegal harvesting.
3. Preservation of an ecotourism resource
4. Disbenefits include small loss in grazing potential and cropping opportunities.

Savanna fire management

**Brief description** Savanna fires release large amounts of CO2, CO, CH4, NOx and particles into the atmosphere. The CO2 is not considered a net emission, but if the fire frequency is reduced, savannas sequester C. CH4 is a powerful greenhouse gas which is a net emission under all circumstances, CO and NOx react to form tropospheric ozone, also an important greenhouse gas. The proposed action is the reduction of fire frequency from once every 2-3 years, to once every 20-50 years.

Technical feasability

Savannas occupy 900 million ha in Africa, and about 1300 million ha globally. All are subject to fire, at frequencies ranging from annually in moist savannas to once in 20 years in very dry savannas. Documented C accumulation rates in fire exclusion plots are 0.8 tC/y for a period of at least 30 years. A principal benefit is the suppression of methane (0.75-6 kg/ha/y, equivalent to 4-40 kg C/ha/y) and ozone-forming gases, which continues indefinitely as long as fires are suppressed.

There are several well-documented fire-exclusion studies in savannas, and the trace gas production from savanna fires is well quantified. Direct uptake of C, direct prevention of GHG emission. Fire suppression techniques are simple and verifiable. The benefits are additional since fire suppression to a high degree would not take place except for global change considerations.

Risks and constraints

The accumulated C is three-quarters as woody biomass and one quarter as soil organic matter. These are pools of moderate stability. A single intense accidental fire will release up to about half of the aboveground wood-stored carbon; repeated fires would desequester the soil carbon over a period of 50 years. Most African savanna regions are under communal tenure, which varies greatly in the degree of control over fire. Where the communal structures are intact, there are examples where they have established effective management. The use of fire is deeply culturally embedded, but could be overcome with suitable alternate benefits. Centrally-imposed fire prevention has universally failed.

Climate change-anticipated climate change may change fire risks slightly, but not sufficiently to invalidate the project. The fire control would be at local level, and therefore not very sensitive to central government changes. Changes of tenure policy are a risk. The suppression of fires would be dependent on continued payments for C stored or GHGs avoided. Savanna lands are currently lightly populated, but subject to increasing pressure.

Costs and benefits

Costs and benefits would be locally carried. The C storage benefits accrue approximately linearly over 30 years; the GHG avoidance benefits accrue year by year. Setup and overhead costs are low in both cases; about \$1/tC and \$1/ha

Ancillary costs and benefits

Excessive burning is a form of degradation; fire at natural frequencies is not. Therefore fire suppression is not in general a method of reversing degradation. Loss of fire-dependant biodiversity is a cost; partly balanced by gain of fire-sensitive biodiversity. Loss of grazing potential is a cost.

Improved pasture management in arid Asia

Brief description. Application of phosphorus and control of grazing (particularly its timing relative to flowering and seed set) allows the development of a sustainable legume/grass pasture, which then pomotes the recovery of soil carbon.

Technical feasibility

10 million ha is potentially involved the Near East. About 0.1tC/ha/y could be accumulated for 30 years. Several demonstrations of technical feasibility exist. C storage is direct, but relatively complex grazing management is involved. Carbon storage is additional, but incidental to the main objectives.

Risks and constraints

The form of C is Soil Carbon, about 1/3 of which is highly stable, and the other 2/3 has a half-life of 10-30 years. Requires secure tenurial control and fencing, which are problematic in most areas. Not highly sensitive to climate change, sensitive to tenure policy change, sensitive to changes in the relative value of crops and livestock, highly sensitive to increasing land pressure.

Costs and benefits

Costs and benefits largely locally born; benefits in terms of improved pastures are the main local motivation. Forage improvement within 3 years, carbon storage gradual and sigmoidal (initially low, then increasing, finally saturating). High initial costs for fertilizer, seed and fencing. Monitoring/verification costs high.

*Indicative costs:* \$10/tC and \$30/ha.

Ancillary costs and benefits

Biodiversity preservation, erosion control, improved livestock production.

Methane oxidation in desert soils

Brief description All well-aerated undisturbed soils not subject to N fertilization oxidise methane at a low rate, including very dry desert soils.

Technical feasability

1 billion ha globally of hyperarid lands involved; also applies to much of the 5.2 billion ha of drylands. No C uptake, but methane uptake of 1-30 kg/ha/y (equivalent to 5-150 kgC/ha/y), indefinitely. Several demonstrate the existence of the oxidative sink. The benefit is direct and simple, but it would happen anyway.

Risks and constraints

No social, political or economic risks, sensitive to N deposition

Costs and benefits

No operational costs, no net (additional) benefits. Significant validation costs

Ancillary costs and benefits.

None

Agroforestry (trees on smallholder farms)

Brief description Smallholder farmers can be encouraged to plant multipurpose trees, which store carbon apart from their benefits in terms of microclimate, nutrient flows, fruit and timber.

Technical feasibility

10% of the arid land in Africa (50 million ha), 15% of the semi-arid land in Africa (75 million ha) and 20% of the subhumid in Africa, Asia and South America (150 million ha) have realistic agroforestry potential. Rate and duration of carbon uptake is 0.2 tC/ha/yr in arid areas for 30 years; 0.5 tC/ha/y for 20 years in semi-arid, 1.5 tC/ha/y for 15 years in subhumid areas. Many supporting studies .Direct benefit, relatively complex crop/tree management. Partially additional (additional investment could increase and accelerate adoption).

Risks and constraints

*Form of C*: Wood and soil, moderately secure. High acceptability at individual farmer level. Not highly sensitive to anticipated climate change, sensitive to tenure change (only effective under individual tenure), not highly sensitive to economic change, promoted by population pressure.

Costs and benefits

Costs and benefits born by the individual. Carbon benefits sigmoidal over time; other benefits begin after 3-6 years. Moderate setup costs, high validation costs.

*Indicative costs:* \$2-5/tC and \$0.3-2.0/ha

Ancillary costs and benefits

Improved quality of life and diet, improved farm productivity, risk spreading, erosion control.

Domestic biomass fuel use efficiency in developing countries

Brief description Improving the efficiency of domestic biofuel use will spare trees, reduce $CO_2$, methane and CO emissions and replace fossil fuels. Actions include:

1. Efficient use of firewood in cooking
2. Production and distribution of high-tech stoves
3. Exploit potential methane fuel resources (digesters)
4. More efficient charcoal production

Feasibility

Applicable to drylands dependent on biofuels but generally not in serious fuelwood deficit; at least 200 million tons of fuelwood are consumed annually in Africa alone.

For non-fossil fuel effects, the time-span involves the period over which existing carbon is preserved, or carbon recovers on over-utilized areas. Active uptake over 30-50 years.

Potential C uptake rates are highly site specific. Open hearth efficiencies are around 10-15%, modern stove designs can achieve 60%. Charcoal-making efficiencies are 15-20% using traditional methods, efficiencies can be raised to 50% and methane production decreased using improved kilns. Applying these approaches, a maximum of 75 million tons C could be saved annually in Africa, plus a large quantity of methane. Maximum carbon "uptake" per stove a direct function of stove efficiency (existing vs. traditional efficiency X per capita wood consumption). Digester impacts easy to calculate.

*Pilot*: Stove production and distribution systems are beyond pilot phase in many areas. Carbon implications of alternative program designs and situations not well understood. Solution -- set up Digester pilots in arid lands?

*Simplicity and Directness*: As long as stove design accounts for cultural and design issues, not a complex mechanism. However, very indirect (assumes reduced pressure on natural resources).

Risks

The benefit is at risk throughout the project period. Risk to above-ground biomass constant (both natural and anthropogenic factors). Project sustainability vital. Susceptible to population growth and land-tenure variables that affect potential use of "saved" carbon. Are there areas with good land tenure that are not in significant fuelwood deficit? Potential cultural barriers to adoption.

Susceptible to factors (e.g. land tenure) which may change resource management potentials. Susceptible to economic downturns or decreased fossil fuel prices (encouraging increased resource exploitation).

Costs and Benefits:

Carbon-related costs to a funder include establishment of production and distribution systems, possible financing mechanisms for both systems, possible costs associated with improved woodlands management and maintenance.

Carbon benefits include carbon storage and sequestration, reductions in other greenhouse gases from more efficient combustion. Timing of benefit is advantageous (starts immediately).

Monitoring and verification requires showing a link between stove use and woodland biomass stores. Could be expensive/complicated. Remote sensing may be useful. Issues of current and projected fuelwood deficits very influential.

Ancillary Benefits: Non-carbon benefits are extensive.

1. health benefits
2. economic development and business establishment
3. saved labor
4. enhanced range productivity

Energy Crops on Drylands
Brief description: biofuel crops are planted on marginal land and used in place of fossil fuels.
Feasibility
Area potentially involved: Salt or water-limited areas, not under other productive land use. Existing or pending fossil fuel use in the region. To the extent of displacing existing fossil fuel use, the benefit is fast and ongoing. The potential C uptake rates are relatively high on good sites (4-8 tons C/ha for ideal non water-limited halophyte sites). Carbon benefit is a direct function of fossil fuel displacement, rather than increased on site carbon density.

There is very little integrated bioenergy project experience in arid lands. Biology of halophyte and drought-resistant cropping is established under variety of situations.

Ranges from simple (oil crop production for very local oil usage) to complex (small grid establishment). Direct carbon benefit.
Risks

As long as displacing fossil fuel, carbon security is very high. Lands should be excess to other uses, and less susceptible to population growth and related pressures. Secure individual land-tenure important to long-term economics.

Susceptible to factors (e.g. land tenure) which may change economics. Moderately susceptible to economic downturns and reference case fossil fuel usage. Project sustainability vital to carbon economics, therefore local infrastructure should be carefully screened or established. Contrary incentives (e.g. fossil fuel subsidies) should be accounted for in project design. Climate is not a key variable.
Costs and Benefits

Carbon-related costs to a funder include potential local R&D for species selection, establishment of cropping systems, integrating into existing markets, technical support for mini-grid establishment. Financing costs, but potentially recoverable.

Carbon benefits primarily fossil fuel substitution. Timing of benefit is advantageous (starts immediately). Monitoring and verification should be simple if local fossil fuel markets exist that energy crops can be integrated with. Where fossil fuel markets are nascent or projected, baseline and reference case establishment become more difficult (but not intensive physical data collection).
Ancillary Benefits:

1. economic development and business establishment
2. Establishes incentives for resource protection and management.
3. Energy crop residues can be used for fodder, with associated grazing impact benefits.

Bush encroachment in semi-arid savannas
Brief description

Enhance or permit the encroachment of shrubs in semi-arid savanna and woodland landscapes or discourage their clearing once this has occurred .
Feasibility

Area worldwide susceptible to encroachment is 200 million ha. The change in C per unit area: 0.1-0.5 tC /ha/y over 15-50 years (at the low end if the conversion is in lower rainfall areas); also note that the conversion has already happened in a significant (20%?) proportion of these woodlands. There are abundant data on changes in different systems which could be synthesized; some systems (*eg.* Jornada) show very small gains, however. The activity must be integrated with the local community's use of the resource, and hence usually depends on some indirect management effects. The basic management is simple, but the consequences and need for local support are less so.

In most areas encroachment is undesirable for grazing production so, although it is occurring widely, there are genuine additional opportunities to enhance this or to discourage future clearing. C storage would be the principal goal, since lower densities of shrubs would be preferable for other productive purposes (possible exception: fuelwood production).

Risks and constraints

Form of C: this is mainly in above ground woody biomass; in some systems perhaps this could be enhanced by deliberate burning strategies to create soil charcoal, although this has implications for other GHGs. The encroachment is compatible with (although disadvantageous for) continued grazing of the understory, or browsing; with appropriate management it could provide fuelwood, once at equilibrium. Other economic benefits to the local land users are not great.

On the one hand, once established so that fires are suppressed, the system will tend to stay in an encroached state; on the other hand, there are probably insufficient community benefits to prevent clearing if the economic imperatives became strong enough to justify this. Climate and atmospheric $CO_2$ changes are likely to mildly enhance the ease of conversion. Substantial areas occur in regions with 'good' political stability (eg. US, Australia, EU), although much of the capture (vs maintenance) benefits may already be gained in these countries economic and land use pressures could cause clearing since local communities gain relatively little direct benefits

Costs and benefits

*General note*: these differ greatly between commercial and subsistence areas: in the former, bush encroachment is seen as a problem, so that its C benefits may be perceived as a solution; in the latter, pressures on fuelwood, etc, usually mean that more shrubs would be seen as beneficial, but the short-term cost of permitting their establishment and growth may be seen as a problem. Costs are establishment protection (*eg.* fire suppression, possibly reduced grazing if extreme, etc), then protection from clearing and fires once established.

In subsistence areas, some local fuelwood benefits could accrue in the medium term, but mostly locals would see costs of reduced access during establishment; in commercial areas there would be loss of pastoral production (or opportunity costs in the case of maintaining existing encroachment). In both cases the main benefits would be C storage externally.

*Timing*: costs are immediate in establishment of protection, with modest ongoing monitoring and maintenance; benefits are over decades, except from prevention of clearing which begins immediately

*Security*: since much storage is above ground, it is never highly secure; security would be much higher in commercial than subsistence rangelands

*Hidden costs*: there would be an on-going need to protect from fire, although the reduced herbage growth under the shrubs means that this is relatively simple if large areas are involved, and especially if grazing continues; since most storage is in aboveground biomass, monitoring is

relatively simple and could be carried out cheaply from remote sensing (*eg.* aerial videography) or even on the ground. For the community to accept and participate in management, education opportunities may be needed.
*Indicative costs:* \$10-20/tC, \$5/ha
Ancillary benefits

If food security and power and water needs are met, a wooded environment is often regarded favorably. If all grasslands are converted to woodlands, or if unconverted grasslands are then used heavily, there may be severe biodiversity implications for grasslands biota (solution: maintain significant patchiness in the approach). If grazing continues under the dense bush, overgrazing can cause severe erosion (as is occurring in Queensland, Australia); this may be deleterious for C storage but also trigger sedimentation and water supply problems

Fuelwood would be an ancillary benefit. Additionally, in the African context the development of an integrated system of natural resource use could provide more general social benefits and be compatible with this management.

Halophytic shrub plantations
Feasibility

*Area worldwide:* mainly aimed at re-vegetating salinized soils - suitable areas occur in marginal cropping drylands worldwide - estimated 150 million ha. Change in C per unit area: 0.5-5tC/ ha/y. Data/studies: there are many studies on establishment feasibility in the US, Africa, India, Australia, etc. C benefits are direct, simplicity varies with the option selected. Although some planting is already being done for forage, it would be easy to determine how additional this was; in general it would have other benefits. Improved reliability and amount of forage supply would probably be the main goal, although in rehabilitation areas C storage could be primary
Risks and constraints

C is mainly in the form of aboveground biomass, with a small amount of soil C. Since much storage is above ground, it is never highly secure; however, salty ground has little value for other land use. In the very long-term, large areas of such rehabilitation should presumably restore water tables and make the region potentially productive for other purposes again - this is probably not a concern since the time frame for this is likely to be many decades to centuries This option could be done with minimal participation. Sensitivity to external factors: climate and atmospheric $CO_2$ changes are likely to mildly enhance the ease of conversion; substantial areas occur in regions with 'good' political stability (eg. US, Australia, EU); economic and land use pressures should not interfere with this since it will occur on salt-affected ground and will be perceived already as productive
Costs and benefits

Costs are immediate on establishment, with modest ongoing monitoring and maintenance; benefits are also quite quick, since growth may only take 5 years to equilibrium above ground. Hidden' costs include fire and re-seeding. Since most storage is in accessible aboveground biomass, monitoring is particularly simple.
Ancillary benefits
Rehabilitation of particularly intractable degraded areas, with potential benefits to watertable quality, etc. Although rehabilitation is not to a pristine state, the ecosystem improvement is likely to benefit biodiversity. However, most projects are proposed to use monocultures of non-native species - this

could be seen as a significant disbenefit in some countries (solution: seek 'natural' mixtures of native species and accept a possibly slightly lower return)

**Fertilization and residue management in dryland crops**

**Brief description:** Carbon inputs to the soil can be increased by increasing crop yield through fertilisation, and by returning crop residues to the soil

Feasibility

All of the rainfed croplands (450 m ha globally) are potential candidates. Carbon can be stored at a rate of 0.3-1.0 t C/ha/yr for 5-20 years, dependent on climate variability and also capacity for a soil to sequester C. Texture will govern the storage capacity in the long-term. Empirical data: Quantified, but data not necessarily analyzed for long and medium term agronomic trials worldwide.

Relatively complex, considering specialized equipment may be needed for strategic fertilizer applications and residue management on a large scale and fertilizers and herbicides to control disease are expensive. Fertilizer uptake and herbicide effectiveness are also heavily dependent on water status of soil. Partially happening already, but the rate and extent of adoption could increase. Carbon storage is subsidiary justification, however the relation between improvement of fertility and soil structure is generally accepted by landholders and thus selling it in terms of C storage is possible.

Risks and constraints

*Form/security of C:* Dependent on rotation but generally moderately secure

*Sustainability/long-term participation:* Participation would be long-term in terms of sustainability issues, however cost is prohibitive to support its use every year.

*Sensitivity to external factors:* Climate change: reduction in croplands

Political stability: Yes in US and Europe as fertilizer subsidies are politically correct, yes in Australia if fertilizer costs were subsidized. Yes in developing countries if external sources will fund operations, equipment and chemicals.

Economic change: Only viable if all chemical and equipment costs are reduced globally. Land use pressure: Little considering need for increased crop production to meet world demands; croplands will therefore expand in future.

Costs and benefits

Better return and increase in cash economy of local producers. Regional benefits through increased cash flow, employment improvement to meet demands during crop season. Costs born by individual farmer. Immediate benefits in terms of crop production improvement, later benefits will be as fertility increases, a reduction in fertilizer usage may improve cash economy of landholders whilst still achieving benefits secured from previous high-input management. This will be cyclical and fertilizer inputs will need to increase again later. C storage benefits asymptote.

Hidden costs of herbicide resistance, breeding of new crops to overcome disease problems and verification of stored carbon.

*Indicative costs:* \$1-5/tC

Ancillary costs and benefits

- Quality of life improvement through high crop production and cash inflow.
- Loss of diversity, but new crops may offset this.
- Landscape stability through erosion control

Summary of options

| Option | Area (mill ha) | Rate (tC/ha/y) | Period (yr) | Cost (US$/tC) | Total (MtC/y) |
|---|---|---|---|---|---|
| Dryland crop mangt. | 450 | 0.3-1 | 5-20 | 1-5 | 135 |
| Halophytes | 130 | 0.5-5 | Indefinite if harvested 5 yrs if not | 170 (irrig & harvested) 20 (??) (dryland not harvested) | 65 |
| Bush encroach-ment | 150 | 0.1-0.5 | 15-50 | 10-20 | 37 |
| Energy crops | 20 (5% of dryland crop area) | 4-8 | indefinite | 150 | 80 |
| Domestic biofuel efficiency | not applicable | not applicable | indefinite | 2-5 | 75 |
| Agro-forestry (arid) | 50 | 0.2 | 30 | 2-10 | 10 |
| Agro-forestry (semi-arid) | 75 | 0.5 | 20 | 2-10 | 38 |
| Agro-forestry (subhumid) | 150 | 1.5 | 15 | 2-10 | 225 |
| Improved pasture (semiarid Asia) | 10 (2500 degraded globally) | 0.1 | 30 | 10 | 1 |
| Savanna fire control | 900 (globally) | 0.5 | 30 | 1-5 | 450 |
| Woodland mangt. | 400 (globally) | 0.5 | 30 | 1-5 | 200 |

See explanatory notes above

## Report of Working Group 1. Indicators of Carbon Sequestration in Degraded Lands.

Land degradation in dryland areas is a syndrome in which net productivity declines and soils lose the physical, chemical and biological properties that confer resilience to additional stresses. As a result of land degradation, system carbon storage may become reduced through loss of vegetation biomass and soil organic matter. Carbon sequestration is the process of carbon stock aggradation and may be viewed as a key to reverse land degradation. Carbon sequestration is not the carbon stores themselves, but increases in those pools for substantial periods of time as a means of offsetting atmospheric changes. However, carbon sequestration and its potential may be evaluated by comparing the carbon pools of lands at various stages of degradation and recovery. Carbon pools and fluxes are distinguished from one another for several reasons. Carbon pool data rely on widely applicable field procedures and chemical analyses. Flux measurements involve repeated or comparative measures, or more state of the arts sensors. We propose these indicators to assess the carbon status of drylands at several scales of resolution for use in projects designed to combat land degradation and desertification.

Scales. Different suites of measurements obtain relevance at different scales of spatial resolution. For purposes of simplicity, these scales, from finest to broadest, may be identified as the:

1) field and site
2) human community and landscape
3) national and sub-regional scales.

Indicators identified by working group members that fall within the various scales include direct field measurements, semi-quantitative survey data and geo-referenced data sets (Figure 1).

Figure 1. Some indicators of carbon sequestration in drylands at various scales of observation

| Scale of Observation | Carbon Sequestration Indicator: carbon pools | Carbon Sequestration Indicator: carbon fluxes |
|---|---|---|
| field and site | biomass C<br>litter C<br>soil C<br>secondary carbonates | CO2 emissions<br>NPP<br>indicator species |
| landscape | litter cover<br>vegetation cover<br>land use<br>denuded caliches | livestock density<br>fire coverage<br>fire frequency<br>residue return frequency<br>soil erosion |
| national and subregion | vegetation type<br>albedo<br>vegetation cover<br>land use | sequestration potential<br>Pot C - Obs C |

Indicators of carbon status differ with scale of resolution but several measurements may be aggregated with scale.

1) Carbon Pools And Fluxes At The Field And Site Scale.

Carbon Pools.

**Organic Carbon:** Direct measurement of biomass, litter and soil carbon are relied upon at this scale. Lands are stratified by degradation stage, use or vegetation types and carbon pools are sampled along repeated, randomised transects. Woody vegetation is quantified within 50 m x 4 sample areas (at least 5) by recording the diameter at breast height (DBH in cm) of all shrubs and trees with DBH of greater than 2.5 cm and then calculating the biomass of each tree using allometric equations developed for appropriate regimes (see Anderson and Ingram, 1993; Brown et al., 1989). Understorey and herbaceous vegetation is measured in two 1m x 1m quadrats positioned at random within the larger sample area. All vegetation, including woody vegetation with diameters up to 2.5 cm is recovered by cutting at the soil level. Surface litter is then recovered to the mineral soil level. In areas of finer, more homogeneous vegetation such as grasslands, more (4-6), smaller quadrats are preferable. Surface litter is hand separated into fine, coarse and if necessary, charcoal. Soils are collected from a 20 cm x 20 cm x 50 cm monolith and roots recovered from the soil by washing and hand sorting. Herbaceous vegetation and litter fractions are separately chopped, mixed, subsamples, dried and analyzed for carbon. Soil subsamples are collected along the central axis of the plot (50 m) to 50 cm at regular intervals (10 subsamples at 5 m intervals), bulked, mixed, subsampled, dried and analyzed for carbon. Following the above procedures, the individual carbon pools; woody vegetation, herbaceous vegetation, coarse litter, fine litter and soil may be summed to estimate the total system carbon to 50 cm (or any other) depth. Greater detail on these procedures may be obtained from Anderson and Ingram (1993).

**Inorganic Carbon:** Secondary soil carbonates amount to ca 1000-1200 Pg C, with an annual growth/carbon sink of ca 10-20 Tg C, depending on release of Ca/Mg by weathering of silicate parent material. More recent variations may arise regionally from changes in precipitation cycles and increased acidity in the atmosphere (car exhausts, $SO_2$ or $(CH_3)_2S$, oxidised by OH- radicals, rising in concentration of ex- by thinning of the ozone layer and increasing radiation impact). C-flux changes in and around caliches can be assessed by thin layers wise $^{13}C$ - scan of caliches, whose $^{13}C$ level reflects C-flux contributions by primary carbonates, ($d^{13}C=\pm0$) 1 atmosphere ($d^{13}C$=ca -8) vegetation ($d^{13}C$= $C_4$ -12, CAM -17, $C_3$ -25).

Carbon Fluxes.

A variety of carbon fluxes may be measured at the field and site level. One of particular importance is changes due to fire. This observation requires that carbon pools be characterised before and after a representative burn. The measurement of greenhouse gas fluxes may be measured using various sensors for $CO_2$, methane or nitrous oxides or by trapping and subsampling volumes of atmosphere above vegetation or soils. Another important flux is actual and observed biomass accumulation or NPP. Vegetation measurements are repeated during the year in representative land uses and compared to the biomass accumulation in exclusion plots. Exclusion plots are fenced areas containing climax vegetation or allowing natural succession to proceed. Another, more subtle indicator of carbon fluxes at the field level is the presence or absence of key indicator plant species. Woody vegetation has potential to accumulate biomass carbon most readily and those the presence of long-lived species an indicator of C sequestration. Other plant species are associated with degraded landscapes and indicative of more reduced C status. These plant species are, of course site specific and their identification likely to emerge from vegetation description of a variety of land uses and soil types within an area. Additional carbon pools and fluxes are presented in Table 1.

## 2) Carbon Pools And Fluxes At The Community And Landscape Scale.

### Carbon Pools.

Several semi-quantitative, ranking and frequency observations suffice to serve as indicate C storage at the landscape or community level. Because it is at this level that project impacts will be assessed and it is through this level that individual measurements become aggregated, it is important to correlate these observational data to the more precise observations at the field level. Broadly applicable indicators include percentage of cultivated soils under litter cover during wet and dry seasons, percentage vegetation cover and exposed soil, presence of denuded caliches and coverages of stable, degraded and at risk land uses. Denuded caliches occur at the soil surface in areas which are affected by deflation and water erosion. They deprive the landscape for crop production because of their physical properties (hard const now in top soil) and or make further carbon sequestration by accumulation of biomass is possible. Under certain atmospheric conditions they can turn into C-sources again.

### Carbon Fluxes.

Relevant community-level carbon flux indicators are extremely important and require careful selection based on project goals and community practices. These indicators are to a large extent dependent upon the production systems, preferences and expectations of the local population and to achieve greatest relevance should spring from them through participatory activities. They may be thought of as participatory signals of either continued land degradation, or conversely, the access to tools necessary to combat land degradation and the willingness to use them. Examples of these indicators include degree of control over natural resources, availably of external inputs and community land care activities. The availability and use of improved breeds of livestock and plant varieties are also important socioeconomic indicators.

Other landscape indicators of carbon fluxes have more general application, and may be obtained from secondary information, reconnaissance and remote sensing data. These indicators include livestock density, the proportion of crop residues returned to the soil, the frequency and coverage of fires and the coverage and rates of soil erosion.

**Livestock** density on semiarid lands is major determinant of land productivity or degradation. Rising livestock density with rising population density widely accelerates land degradation, annihilates gains from advanced range-farming models. Mitigation due to shifts of climate/vegetation zones during extended climate extremes (Sahel, El Viro effects) increasingly leads to overgrazing when returning, back to mare arid environment, find their old living area occupied by new comers. Overgrazing is a tragic consequence of the basic trend, that each year ca 10 million ha - mainly cropland + grassland - are last by erosion versus a 92 million population increase.

**Soil erosion** and costs of repair costs of erosion by deflation and water erosion are estimated to amount worldwide to man through 60 lt per year. In arid and

semiarid lands erosional damage is characterised by the constant ragging of wind erosion by consistent slope erosion and in a dramatic measure by sudden periodic wasteful precipitations with variable land slides. Vegetation cover, possibly of deep rooted species with V-antoleophic legumes, terraced slopes, if possible with appropriate reforestation, are the main mitigation strategies. They also optimise the potential for C-sequestration in stable SOM. Recently the soil area, lost by erosion each year, is estimated globally with ca 10 million ha from that more that a commensurate share in the arid and semiarid lands.
Residue return frequency: Various techniques exist but the most easily monitored is the immediate return of crop residues to soils for use by the following crop. This assessment becomes more difficult when manures are produced from straw and stubble browsing, or composts produced. It is also difficult to assess the uniformity of secondary organic inputs.

## Carbon Pools And Fluxes At National And Subregional Scales.

### Carbon Pools.

Changes in six pools influence soil physical properties including soil moisture holding properties. In (degraded) dryland areas changes in surface-soil moisture holding capacity will be reflected in the albedo, which can be monitored by remote sensing (multi-temporal analysis). Key indicators of carbon fluxes that may be developed from comparing different albedo measurements are changes in land and vegetation cover over space and time. The primary determinants of plant grown and production in any location are insolation and length of the growing period (R/PET; TEMP). Actual production potential is determined by plant type, management practices (socio-economical) and the terrain/soil characteristics. Digitized and georeferenced data on climatic, water presence and terrain/soil data in a multi-layered, environmental GIS can provide the basic input for computing plant growth and C sequestration in any location, using appropriate models. Remote sensing can be used to estimate land cover type and land use changes, providing information on the dynamics of the 'carbon inputs'. Surficial losses of organic matter can be derived from water and wind erosion modules, which again require climatic, terrain/soil, and land use information - held on the GIS - as basic inputs. C losses associated with pipes follows from remotely sensed data. Combined analysis of the various data layers, and model outputs will permit the estimation of possible effects of land use/cover changes on organic matter production and the ultimate sequestration of carbon in soils.

### Carbon Fluxes.

Our approximation of carbon fluxes at the broadest scale we refer to as "carbon sequestration potential". For our purposes, "Carbon sequestration potential" is defined as the difference between soil and vegetation C pools within degraded environments and their undegraded counterparts. This difference represents the potential C accruement that may be achieved in degraded lands through directed land-management. As a result, it provides a predictive index of landscape C storage, which may be applied across local - to- regional scales over time. Measurement and comparison necessarily begins with field specific experimental plots, stratified to characterize the local heterogeneity of soils, vegetation, and land-use patterns. Further stratification across the landscape provides opportunity to aggregate these patterns into a regional context. This nested design can be use to integrate and monitor C pools relevant to site-specific rehabilitation projects and larger biogeochemical processes.

*Measurement and Application.* Field exclosures (1000 $m^2$) are established within degraded and undegraded plant communities stratified by land-use. A minimum of three plot replicates (per site) are necessary to insure statistical confidence. Within each exclosure subplots (100 $m^2$) are established and used to segregate sample collection through time. Annual measurements of net primary productivity (NPP) for overstory and understory species are grouped across exclosures by degradation classification. Mean NPP of undegraded sites less the mean NPP of degraded sites represents the "Vegetation Sequestration Potential". This difference is reported in units of C/$m^2$/yr. Total organic C in soils (to 1 m) is measured and compared similarly. This difference represents the "Soil Sequestration Potential" and is reported in units of g C/$m^2$. Carbon stored in surface litter and standing dead biomass are also quantified and reported (g C/$m^2$) on an annual basis. Carbon storage in each of the previous pools are summed for each exclosure, grouped by degradation class, and averaged. Mean differences between undegraded and degraded sites represent the "Total Sequestration Potential". Following application of management strategies to sequester C (in degraded sites), these indices can be used to assess rates of C gain/loss at localized sites.

Total Sequestration Potentials are aggregated to the landscape level, stratifying by land-use patterns, vegetation type, soil order, and topography. Annual changes in total average of variables 1 and 2 are determined by areal photography and ground-surveys. Total Sequestration within these broader categories can be linked to cover changes to determine landscape-level C fluxes, reported in kg/ha. Carbon flux at this scale is relevant to greenhouse emissions of individual countries through time.

Landscape C pools are aggregated at the regional scale and monitored using geographical information systems. At this scale, landsat imagery may also be applied to verify landscape cover changes and coincidentally, identify increases in degraded areas -which represent "C Loss Risks". Over periods of decades the Carbon Sequestration scheme may be applied to track large-scale C fluxes relevant to global biogeochemical cycling.

## References

Anderson, J.M. and Ingram, J.S.I. 1993. Tropical Soil Biology and Fertility: A Handbook of Methods (Second Edition). CAB International, Wallingford, UK. 221 pp.
Brown, S., Gillespie, A.J.R. and Lugo, A.E. 1989. Biomass estimation methods for tropical forests with applications to forest inventory data. Forest Science 35:81-902.

Table 1. Additional indicators of carbon pools at site and landscape levels and their dimensions.

| Processes | | Indicators |
|---|---|---|
| Biomass farming (biomass C) | t/ha<br>t/ha year<br>t/ha<br>%, co-efficient | 1) reserves of phytomass -<br>2) annual production -<br>3) reserves of dead plants -<br>4) relation between live and dead |
| Fluxes biomass (fluxes biomass C) | t/ha time<br>t/ha time<br>t/ha time | 1) changes of reserves<br>2) changes of production<br>3) changes of reserves of dead material (litter etc)<br>4) changes of relations of live/dead |
| Farming of soil C | $g/m^2$. 1m<br>$g/m^2$. 1m<br>$g/m^2$ day (season) | 1) reserves of soil C (organic)<br>2) reserves of inorganic soil C (soluble, insoluble)<br>3) respiration of soil $CO^2$ farming) |
| Fluxes of Soil C | $g/m^2$.10 year<br>$g/m^2$.5-10 year<br>$g/m^2$.day %<br>t/ha.year % | 1) change of organic soil C reserves -<br>2) Change of inorganic soil C (soluble, insoluble) -<br>3)change of soil respiration (microbiological activities)<br>4) after effects of organic fertilizations using (plus field, soil (a.o.) |

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