4.2 Soils

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4.2.1 Introduction

4.2.1.1 Background: soils around the world

Soils are crucial for profitable and sustainable biomass feedstock production. They provide nutrients and water, give support for plants, and provide habitat for enormous numbers of biota. There are several systems for soil classification. FAO has provided a generic classification system that was used for a global soil map (Bot *et al.*, 2000). The USDA Natural Resources Conservation Service has produced a world soils map based on the FAO system (Eswaran *et al.*, 2003, Figure 4.1). An initiative of the Digital Soil Mapping Working Group of the International Union of Soil Sciences is moving forward with making a world digital soil map to assist decisions on food and energy production. Soils are extremely diverse due to the diversity of parent materials, climates, topography, and biota that act over time to produce what we know as soil (Jenny, 1941).

Soils in Latin America are dominated by weathered clay soils in the north-east (Brazil) and more productive soils in the south. Soils in North America are characterized by mostly productive clay soils in the south, glacial-derived soils (containing layers of accumulated soil organic matter) and peat soils in the, poor sandy and clay soils in the east, and the highly productive prairie soils of the mid-continent. Asia has a similar distribution of soils as North America: predominantly peats and spodic soils in the north, less fertile clays in the south-east with large areas of productive clays and sandy soils in the remainder of the continent. Soils in Africa are mostly weathered clay soils in the north and south, with a zone of wetland soils in the middle of the continent from the Sahel to Egypt and large parts of the south-east. Heavily weathered dominate large parts of West and Central Africa. An overview of FAO and USDA soil types and their properties was assembled by Deckers *et al.* (1998).

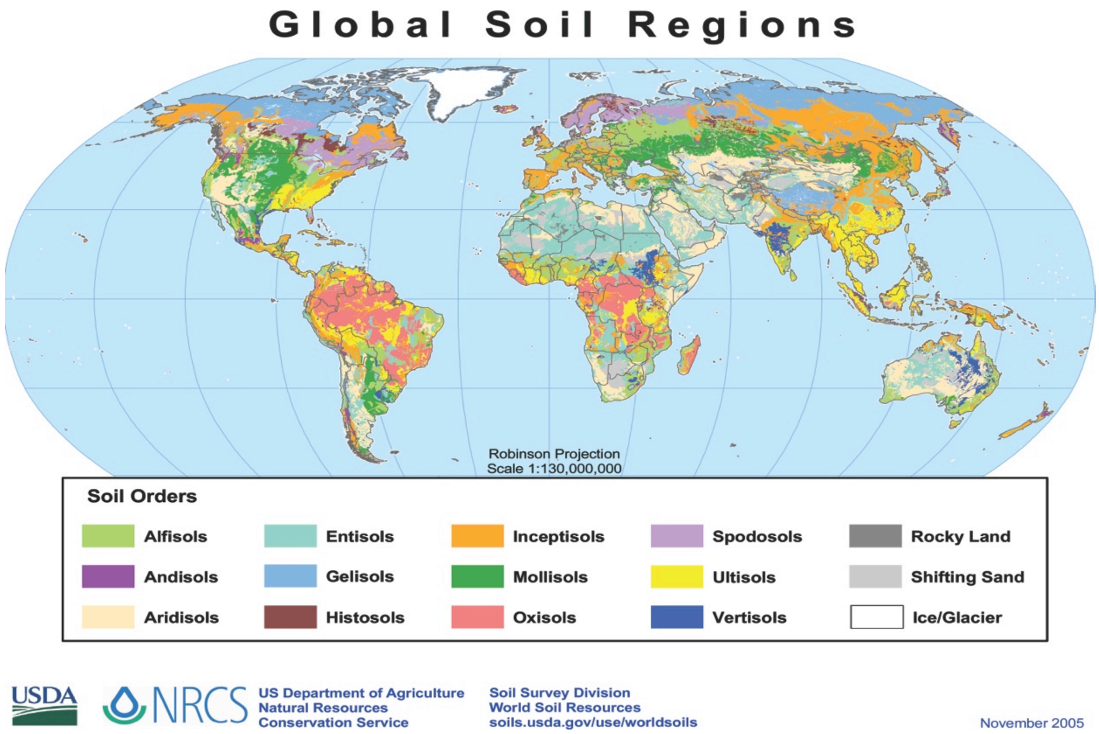

Figure 4.1: The USDA Natural Resources Conservation Service world soils map based on the FAO system and the USA soil taxonomy.

Major problems in soils include poor nutrient status, soil drainage, soil depth, slope, workability, physical limitations, damage due to soil erosion, and chemical restrictions (Figure 4.2). An overview of soil physical and chemical restrictions is given in Table 4.2. Major restrictions are related to high erodibility, aluminium toxicity, shallow soils and problems with soil morphology. Less important problems are low nutrient buffering capacity, phosphorus fixation, cracking and salinity/sodicity. Most problems are rather well distributed around the globe. Exceptions are low capacity for nutrient exchange (mostly restricted to Sub-Saharan Africa), aluminium toxicity (dominant in Latin America and – to a lesser extent – Sub-Saharan Africa) and phosphorus fixation (mostly found in Latin America). Generally, Sub-Saharan Africa and Latin America seem to be endowed with the least favourable soils. Better endowed regions include North Asia (east of the Ural), North America and Europe (Bot *et al*., 2000).

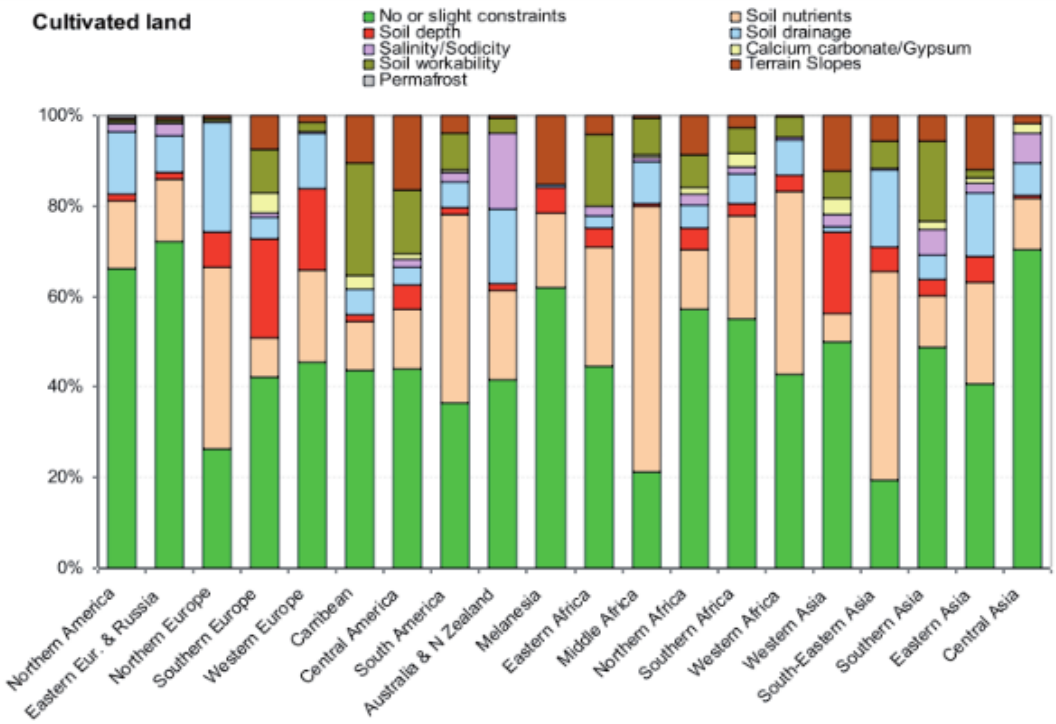

Figure 4.2: Major soil limitations of cultivated land. Source: Fischer *et al*. (2008).

It should be noted that the occurrence of soil limitations does not immediately translate to limitations in crop production. According to Mueller *et al*. (2010), for example, the most production-restricting factor is limited soil moisture which is only indirectly linked to the factors shown in Figure 4.2. The authors further conclude that most soil classification systems provide only limited information on soil productivity. The exact impact of soil-based limitations will depend on local conditions, input level and management style. As a rule, multiple restrictions for crop growth may be cumulative. Some restrictions may be overcome, but options for soil improvement are often limited while costs are often prohibitive. There are three key considerations.

First, soil characteristics are multi-factoral. The yield potential of a soil is the end-result of a combination of soil-related factors, in which the contribution of each factor depends on the total set of soil factors. High organic matter, for example, contributes to good nutrient availability (both inherent and by buffering nutrients that have been applied), good water availability (buffering water), and healthy soils. Any of these capacities can be provided at least partly by other factors: good nutrient availability by rich parent material, good water availability by good soil structure.

Second, soil characteristics are both the end-result and the beginning of dynamic processes. Availability of soil organic matter, for example, is the end-result of previous processes of organic matter contribution, removal of organic matter, and mineralization. Nutrients may be accumulated over many years and sometimes can be lost fairly quickly.

Third, some impacts of soil deterioration are not felt by the one who is cultivating the soil. Sometimes, the damage occurs at a different location. Externality effects related to soil use include mainly soil erosion and the transport and deposition of soil material by sun or wind to other locations.

4.2.1.2 The Soil Management Link to Sustainability

Sustainability is the stewardship goal of crop production and forestry in general, including bioenergy programs. However, a more specific definition of its goals and attributes is complex and open to interpretation (Moir and Mowrer, 1995; Dale *et al.*, 2012). Many ecologists have attempted to answer the "what", "what level," "for whom," "biological or economic," and "how long" questions of sustainability. Allen and Hoekstra (1994) discussed the emergence of the concept of sustainability and the difficulty in defining it. They point out that there is no absolute definition of sustainability, and that it must be viewed within the context of human conceptual frameworks and societal decisions on the type of ecosystem and the spatial and temporal scales over which attainment of sustainability is to be judged.

Sustainability is also defined in terms of the needs of society, the experiential frame of reference of ecosystem managers, and the ecological models that are used to predict future conditions for natural resources. However, the ability to predict future ecosystem conditions is confounded by the uncertainties of increasing encounters with extreme events, poorly understood ecological processes and linkages, high degrees of natural variability, surprises created by the 'law of unintended consequences', the development of critical thresholds, and chaotic system behavior.

Another approach to the definition of sustainability is to define the conditions that warn of or mark ecosystem deterioration into unsustainability (Moir and Mowrer, 1995). For example, although the goals of the Montreal Process and the Santiago Declaration are to ensure management of forest lands for sustainability, the Criteria and Indicators of that process are in essence warning flags to obtain the attention of land managers before ecosystems decline into unsustainability.

Soils are an important, basically non-renewable resource, that provide the physical, biological, chemical, and hydrologic foundation upon which agricultural and forest bioenergy feedstocks grow (Johnson, 1994; Kimmins, 1994; Burger, 2002). Soils are able to renew themselves after being degraded but the time period might be several centuries or even millenia, depending on climate and vegetation. Because of this long time factor, soils are considered to be non-renewable from the viewpoint of human use and management. They are heterogeneous and highly diverse components of ecosystems that form over long time periods under the influence of parent mineral material, climate, landscape position and biological activity.

As the base of bioenergy production system, soils are a major factor in determining crop yields. They provide the physical anchor which tie plants to the earth, supply water and mineral nutrients for plant growth, decompose and recycle organic material and residues, and mediate hydrological processes (Lavelle and Spain, 2001; Neary, 2002; Brooks *et al.*, 2003). Bioenergy feedstock systems are part of agricultural and forest management systems that provide multiple ecosystem products and services, including plant biomass, water flow, water quality control, biodiversity, cultural heritage, and carbon storage (Richardson *et al.*, 2002) (see also Chapter 4.5). Soils are important factors in each of these services (De Groot *et al.*, 2002). Therefore, it is critical that in the process of managing soils for bioenergy production, soils must be managed to sustain all forest values important to human communities.

Table 4.2: Occurrence of major restrictions related to physical and chemical soil characteristics. Source: Bot *et al*. (2000).

Global Location	Area ($km^2.10^6$)	Hydromorphy (%)	Low nutrient exchange (%)	Aluminium toxicity (%)	Phosphorus fixation (%)	Cracking (%)	Salinity and sodicity (%)	Shallowness (%)	Erosion hazard (%)
Sub-Saharan Africa	23.8	8	16	18	4	5	4	13	15
North Africa, Middle East	12.4	1	2	0	0	1	6	23	10
Asia, Pacific	29.0	11	4	14	5	5	11	17	16
North Asia	21.0	27	0	4	0	0	10	13	16
South, Central America	20.5	10	5	39	15	2	5	11	19
North America	21.4	16	0	10	0	1	1	12	18
Europe	6.8	17	1	8	0	1	3	12	20
Total	13.5	13	5	15	4	2	6	14	16

4.2.2 Relevance of soils in bioenergy-related LCA studies

Many problems related to productivity of crops, including biomass crops for energy are soils-based and include low fertility, physical limitations (e.g. parent material, texture, depth, drainage, moisture content, etc.), chemical restrictions (e.g. cation exchange capacity, alkalinity, acidity, carbon content, etc.). Related problems include slope, soil erosion, compaction, and leaching. Soil quality also co-determines the impact of low water or nutrient availability.

Soil compaction, erosion, and organic matter losses are the chief factors that affect decline of ecosystem productivity (Powers *et al*., 1990; Burger, 2002). They can alter ecosystem carbon allocation and storage, nutrient content and availability, water storage and flux, rhizosphere processes, and insect and disease dynamics. The chief disturbances that affect these factors are wildfire, insect and disease outbreaks, climate extremes, vegetation management (wood harvesting and stand tending activities, grazing, prescribed fire, chemical weed control, and manual removal of plant species), and recreation (foot traffic and vehicles) (Hart and Hart, 1993). Management activities that eliminate natural disturbances (e.g. fire suppression, insect control) or alter ecosystem properties can also affect ecosystem sustainability.

As previously indicated, soils are highly variable depending on parent mineral material, climate, landscape position and biological activity. The impact of relatively small changes in soil conditions on crop productivity and long-term sustainability can be huge and often cannot be easily mitigated economically. In addition, disturbances to soils that occur during bioenergy feedstock establishment, harvesting, and transportation can have significant impacts on important environmental parameters such as water quality and ultimately site productivity (Burger, 2002; Neary, 2002). Other elements which play a role in biofuel feedstock cultivation potential include weather, market conditions, energy policy, and societal preferences. Except for weather, these are more or less manageable factors. They may be resistant to manipulation but, compared to soils and weather, they are relatively malleable in the short-term. Soils and weather are physical attributes of bioenergy systems while the other factors are sociological and economic.

4.2.3 Current LCA approaches to soils

4.2.3.1 How are soils normally treated in LCA studies?

How soils can be included in LCAs depends to a large extent how the effects of changes in soil characteristics during bioenergy feedstock production are known, quantified and attributed. While the relevance of soils and their coverage in LCA studies is clear, quantification and allocation of impacts are very difficult to determine. A number of different approaches have been used to assess the effects of bioenergy feedstock production on soils, soil characteristics, and changes in soil quality but direct measures of soil properties and subsequent inferences on sustainability impacts have remained problematic.

Dale *et al*. (2012) discussed many scale issues that need to be considered in the sustainability of agricultural feedstock production systems. They noted the need to quantify the effects of specific management practices on carbon sequestration, nutrients, water, and energy fluxes. An important conclusion was the identification of the need to adopt management practices that reduce pollution, and erosion, reduce wasteful fertilizer and water use, and sustain ecosystem services at field, farm, and regional scales. This means, in essence, adopting Best Management Practices rather than measuring physical, chemical, and biological parameters and inferring sustainability changes.

There is a data dilemma in trying to measure soil characteristics. Many data are needed, which are not easily measured, and correlations with sustainability are problematic, because soils are highly variable and host a number of dynamic processes. Is there a solution to this problem or should a different approach be taken? These questions are difficult to answer. However, estimates of carbon dynamics have been given by IPCC (2006). Generalized concepts and state changes have been developed to evaluate impacts of bioenergy production. These include: Global Warming Potential (GWP); carbon and greenhouse gas (GHG) emissions occurring during production and distribution cycles (Izursa, 2013); Eutrophication Potential (EP); contribution of nutrient emissions to potential eutrophication of surface and groundwater resources (Izursa, 2013); Acidification Potential (AP); and the potential contribution to acidification of land and water (Izursa, 2013), water use (Milá i Canals *et al*., 2008) and ecosystem quality (Núñez *et al*., 2012a) See also Table 4.3.

Table 4.3: Example of generalized impacts related to soil, water and air quality (adjusted from Izursa, 2013).

Category Indicator	Impact Category	Description	Unit
Energy Use	Primary Energy Demand	A measure of total amount of primary energy extracted from the earth, expressed in terms of demand from non-renewable (petroleum, natural gas, uranium, etc.) and renewable (hydropower, wind, solar, biomass, etc.) resources.	MJ
Climate Change	Global Warming Potential	A measure of greenhouse gas emissions (CO_2, methane, etc.) which increase the absorption of radiation emitted by the Earth, magnifying the natural greenhouse effect.	kg CO_2 equivalent
Eutrophication	Eutrophication Potential	A measure of emissions causing eutrophication. A stoichiometric procedure which identifies the equivalence between N and P for both terrestrial and aquatic systems.	kg N equivalent
Acidification	Acidification Potential	A measure of emissions causing acidification. Assigned by relating the existing S, N and halogen atoms to the molecular weight.	kg H^+ equivalent
Water Use	Water Use	A measure of water used from sources including surface and ground water	m^3

4.2.3.2 Examples of LCA studies discussing soil quality changes

Soil organic matter (SOM) was addressed by Milá i Canals *et al.* (2007, 2008) who showed a novel approach to addressing land use impacts beyond the inventory indicator “m^2 y^{-1} of occupied land.” Soil organic carbon (SOC) was used as an indicator of soil quality, and potential changes to SOC linked to different land uses were compiled for the complete life cycle of the products assessed. The results showed that, contrary to assumptions in several other life cycle impact assessment methods, life cycle stages other than cropping may dominate the impacts related to land use, even if cropping still dominates in terms of area per year. Núñez *et al.* (2012b) used soil and weather data to assess impacts of soil erosion in a spatially explicit LCA analysis by calculating soil carbon loss at grid cell levels for energy crops in Spain (Figure 4.3). Gan *et al.* (2014) included damage due to soil erosion in an economic optimization approach for corn stover removal in a lignocellulosic ethanol production region in Iowa, USA. Cowell (1998) and Cowell and Clift (2000) examined soil compaction effects. Croezen *et al.* (2013) analyzed and presented generalized data on soil carbon changes, nutrient inputs, application of agro-chemicals and water use in major bio-based production chains.

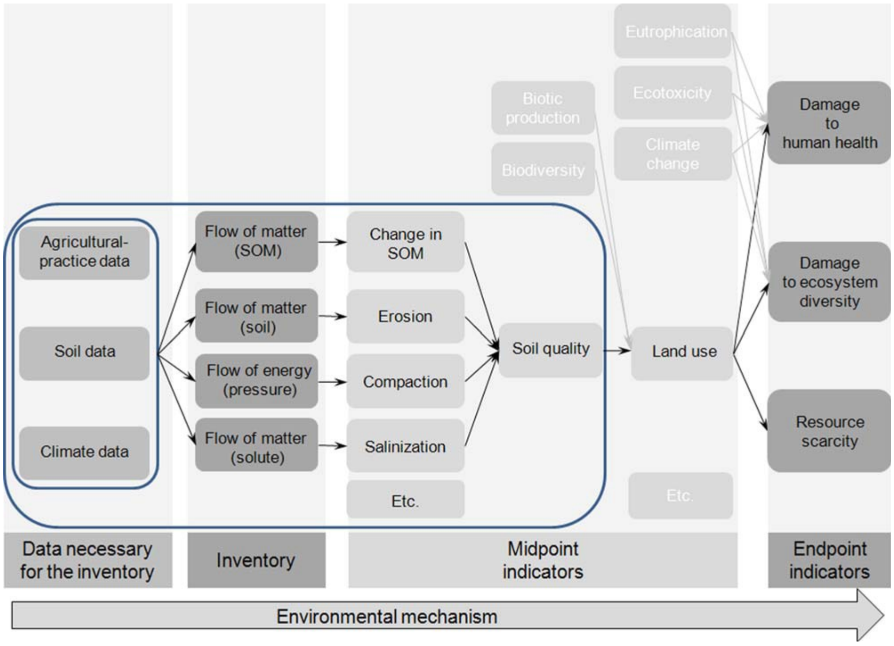

Figure 4.3: Data collection and analysis in LCA (after Garrigues *et al*., 2012).

4.2.3.3 Regional approaches to soil sustainability

Different approaches exist in different parts of the world for addressing soil sustainability in land use and forest management systems. This section discusses specific examples from Canada, the European Union, New Zealand and the USA.

Canada

About 36% of Canada's forests have been certified as being sustainably managed by globally recognized certification standards (Natural Resources Canada 2009). Codes of Forest Practice are in place in Nova Scotia, Ontario, and British Columbia. Canada's forest laws and regulations are considered to be among the strictest in the world. British Columbia has led Canada in developing procedures to ensure forest sustainability. The *Forest Practices Code of British Columbia* of 1996 established the legal framework for monitoring soil disturbances caused by forest operations (British Columbia Ministry of Forests 2001). It has since been augmented by the *Forest and Range Practices Act* of 2002. The Province has an iterative adaptive-management process that provides constant feedback to forest operations and research to improve Best Management Practices and operations planning and execution (Figure 4. 4)

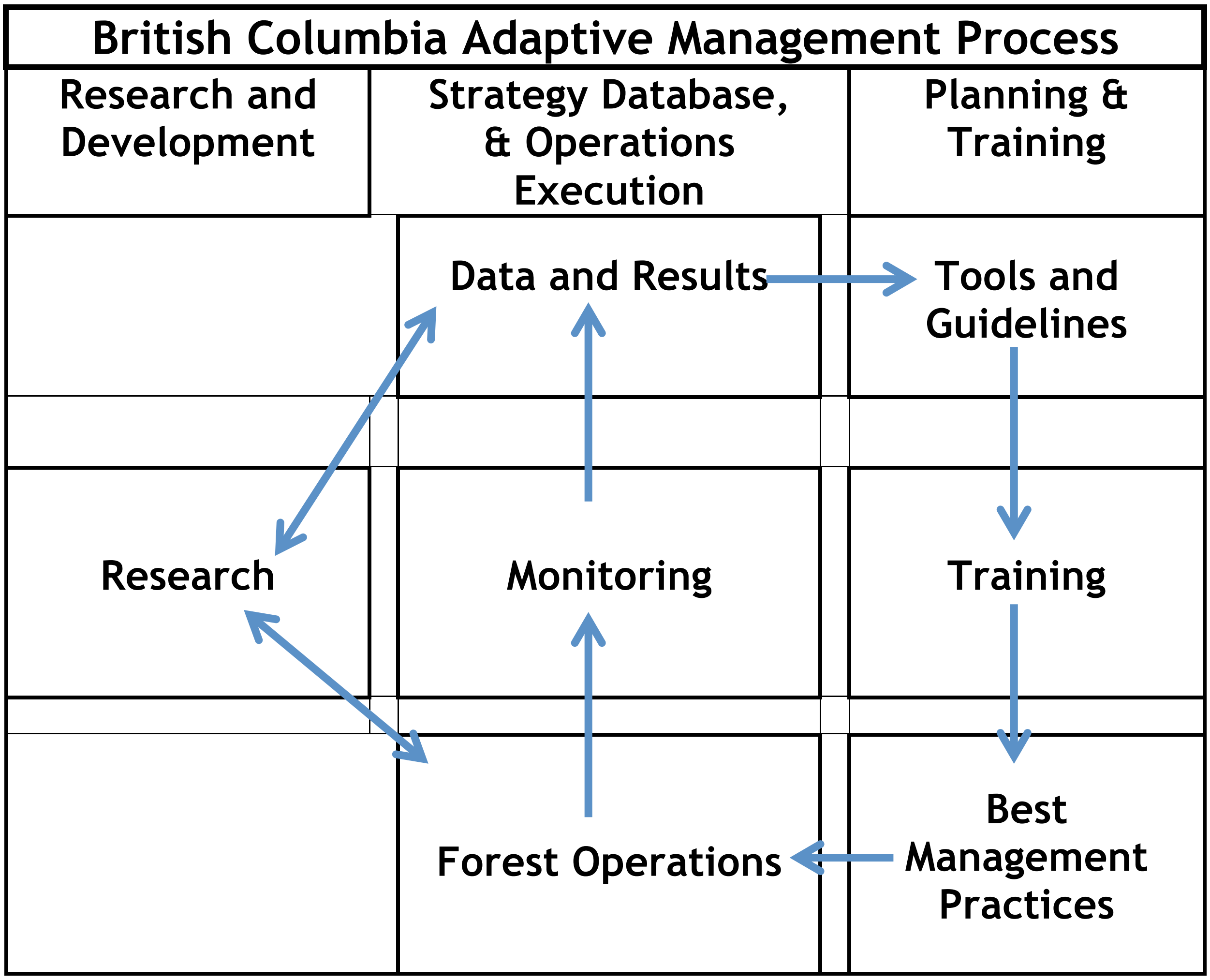

Figure 0.1: British Columbia soil monitoring adaptive management process (adapted from Curran *et al.*, 2005, 2007)

European Union

Forests cover 160 million ha within the European Union (EU), or about 42 percent of the area. Six of the 27 member countries account for two-thirds of the forest area, with Sweden and Finland together accounting for 30% of the total forest area (EUROSTAT, 2009). Official protocols exist in most member countries for soil monitoring (Morvan *et al.*, 2007), but there is a lot of variation in the methodologies used and the intensity of sampling. The EU Monitoring Network has been active since the early 1990s using a 50 x 50 km grid with variable re-measurement periods. Parts of the EU Network have dense sampling grids (United Kingdom, Ireland, Austria, Denmark) while in other areas the network is still sparse (Spain, Italy, Greece) (Figure 4.5). About 90 percent of the EU soils and land cover classes have at least one monitoring site. However, the density of soil monitoring sites within the European Soil Database units is highly variable. Some units (7 percent) do not have any monitoring sites. The highest density of soil monitoring sites is found on pasture lands, as well as on arable land and forests which, however, occupy slightly less area. A grid of 16 x 16 km has been established for forest soils (ICP, 2004). The key soil parameters being monitored in the EU include erosion risk, compaction risk, presence of peat, heavy metals, desertification, and presence of livestock (Morvan *et al.*, 2007).

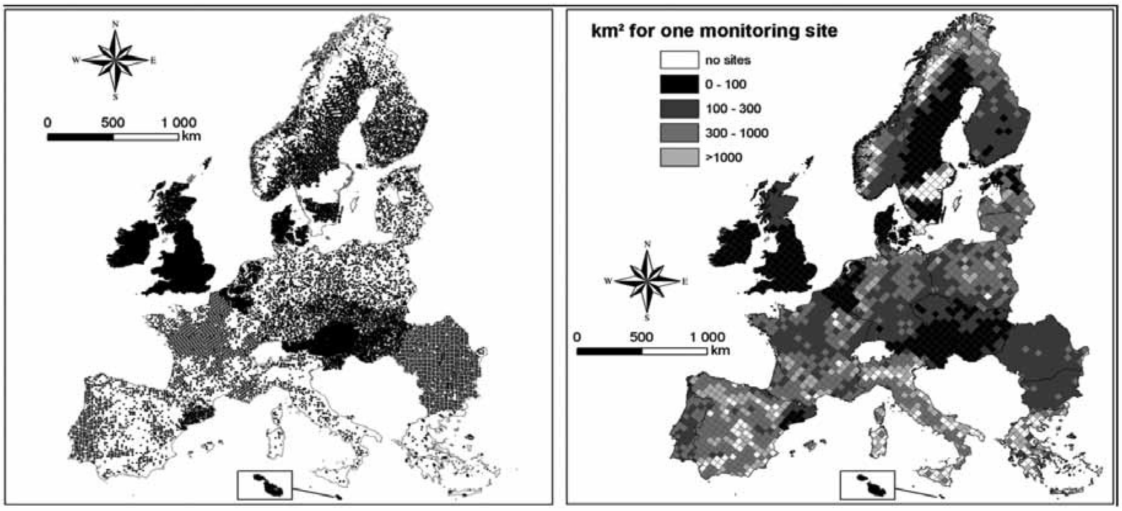

Figure 0.2: European Union soil monitoring network, GIS data subdivision into smaller, more manageable data sets (right) and actual density (left) in km^2 for one monitoring site in grid of soil monitoring sites of the 50 x 50 km Cooperative Program for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe (EMEP) (Moran *et al.*, 2007).

Other indicators being measured are texture, pH, organic matter, bulk density, cations, and earthworm activity (Morvan *et al.*, 2007). The EU Soil Monitoring Network is basically an inventory system that has no interaction with land management, and does not have any regulatory power. Its needs include adding 4,100 sites in the lower density part of the network and standardizing sampling and analytical methods. Of the countries with mandated soil monitoring (Table 4.4), Sweden requires measurements of soil physical conditions, coarse woody debris, and soil chemistry. Ireland requires measurements of soil condition, soil fertility, erosion, and other parameters as needed. Although the United Kingdom does not require soil monitoring at the present time, changes to codes of forest practice will mandate this activity in the future (Bone *et al.* 2010).

Within the EU, Ireland is a good example because of its well-developed code of forest practice. Over 70 percent of Ireland's 636,164 ha of forests are owned by the Irish Forestry Board (Coillte Teoranta). Soil monitoring in Ireland is included in the country's *Code of Best Forest Practice* and is based on EU and national laws (Ireland Forest Service, 2000). As in a number of other countries, the Irish code is focused on planning, monitoring, and adaptive management rather than punitive regulatory actions.

Table 4.4: Monitoring standards and requirements for conventional forestry and forest bioenergy in some European Union countries.

Country	Forest Harvest Practices Code	Bioenergy Guidelines	Monitoring		
			Required	Type	Soil
Denmark	Yes	No	No	None	No
Finland	Yes	Yes	Yes	Operations	No
Germany	Yes	Yes	Yes	Inventory	No
Ireland	Yes	No	Yes	Operations	Yes
Netherlands	Yes	No	Yes	Operations	No
Sweden	Yes	No	Yes	Multiple	Yes
United Kingdom	Yes	Yes	No	None	No

Monitoring is performed to evaluate the performance of the Irish Code of Best Forest Practice as well as the skills of individual forest harvesting operators. It consists of a self-evaluation impact appraisal carried out by the individual operators and an external assessment by the Irish Forestry Board (Ireland Forest Service, 2000). The Irish impact appraisal evaluates environmental, economic, and social impacts of forestry operations. The Irish focus is on assessing potential impacts in terms of their level, likely consequence, importance, and the length of time for which the impacts will occur. Potential impacts evaluated in the Irish impact appraisal were assessed descriptively or on a 'points' system on the basis of four subjective severity levels (very high, high, moderate, and low), and follow-up mitigation actions are then planned. In the appraisal, soil fertility was evaluated as being at high risk because of the soil type and the whole-tree harvesting planned for the cut block. So the mitigation technique prescribed for this stand was the addition of a nitrogen-phosphorus-potassium fertilizer. The other potential soil impacts were evaluated as being low so no mitigations were planned.

New Zealand

A national-scale soil quality monitoring program was conducted in New Zealand between 1995 and 2001 at 222 sites in five regions (12 soil orders and 9 land-use categories) (Sparling and Schipper, 1998, 2002). Land uses in the survey included arable cropping, mixed cropping, pasture, grassland, plantation forests, and native forest. Sampling of the topsoil (0–10 cm) was done and the properties measured were total carbon (C) and nitrogen (N), potentially mineralizable N, pH, Olsen phosphorus (P), cation exchange capacity, bulk density, total porosity, macroporosity, and total available and readily available water. Seven of these soil parameters (total C, total N, mineralizable N, pH, Olsen P, bulk density, and macroporosity) explained 87 percent of the total variability. Some of the issues that arose during the soil quality sampling were minimum data set, how to stratify, level of precision, cost, centralized data and sample management, re-sampling for trends, and sampling strategy. Important conclusions and recommendations from the program related to methodology were that (1) a precision of 10 percent was impractical due to cost, (2) a precision of 25 percent was more realistic, (3) central storage of data and samples was essential to success of this type of survey, (4) re-sampling needed to be over a 3- to 10-year time period with some re-sampling every year, and (5) financial constraints prohibited random sampling.

Key findings from the New Zealand Soil Quality Survey were:

- Soil order had a strong effect on the results.
- Land use accounted for only 21 percent of total C variability.
- There was no evidence of acidification under exotic tree species. Changes in soil quality between land uses can be detected.
- Biochemical parameters and total C indices are more sensitive to land management differences than physical parameters.
- Soil quality of mature pine plantations, before and after logging, was similar to that of native forests or low-productivity pasture.
- Many research needs were identified to make a national-scale soil quality survey a viable management tool.
- Changes in soil quality characteristics can be detected, but there is a general lack of a scientific framework to define acceptable and unacceptable ranges of soil quality parameters.

USA

The Natural Resources Conservation Service (NRCS) and the Agricultural Research Service (ARS) of the USA conduct research and development activities related to soil quality and soil monitoring (Doran and Parkin, 1994; Doran and Jones, 1996; Karlen *et al.*, 1997; Doran *et al.*, 1998; USDA NRCS, 2001). The ARS has also developed standardized methods for monitoring grassland, shrubland, and savanna ecosystems (Herrick, 2005a, b). Soil monitoring conducted by the Forest Inventory and Analysis (FIA) Program of the US Forest Service is discussed in detail by Amacher and Perry (2010) and by O'Neill *et al.* (2005a, b). Forest industries such as the Weyerhaeuser Company are committed to soil productivity. Weyerhaeuser uses a two-step strategy (Heninger *et al.*, 1997, 2002). First, its operations use equipment and operations practices that are appropriate to the soil, topography, and weather to minimize erosion and harmful soil disturbance. Secondly, the company employs forestry practices and technology to retain organic matter and soil nutrients on site.

A reliable monitoring protocol has been identified as a critical component of any adaptive management process for forest and rangeland soil conservation programs (Curran *et al.*, 2005). Uniform and unambiguous definitions of soil disturbance categories must also relate forest productivity and hydrologic function (Curran *et al.*, 2007). A soil monitoring protocol must incorporate a statistically rigorous sampling procedure and firm definitions of visually observable soil disturbance categories

A soil monitoring protocol, first developed by the US Forest Service in its Region 1 and Rocky Mountain Research Station, incorporates soil quality monitoring efforts pioneered in the Pacific Northwest (Howes *et al.*, 1983) and is a multi-faceted approach to soil disturbance and forest sustainability issue. The protocol uses visual soil disturbance classes (Howes *et al.*, 1983; Page-Dumroese *et al.*, 2006) and a standard inventory, monitoring, and assessment tool. It employs common terminology and has an accessible database. The visual disturbance considerations are soil resilience, degree of disturbance, duration, distribution, and location in relation to other resources.

4.2.3.4 Observations on soil monitoring approaches

In general, the soil monitoring systems discussed in this section give a clear definition of system boundaries. While visual disturbance classes can be well-documented, there appears to be a considerable amount of common methodology. This makes it relatively easy to train field personnel. On the other hand there are difficulties in assessing the importance of effects due to variation in time and space. Moreover, correlations to productivity change are not always given. As a rule, monitoring systems are reactive, and procedures for mitigating negative effects are not always clear or in place.

4.2.4 Discussion

4.2.4.1 Difficulties associated with improving assessment of soils in LCA studies

Notwithstanding a considerable amount of overlap, a large variation in soil databases between countries still exists. This is related to the fact that soil classification systems still differ. The EU and USA classification systems cover the largest land masses, but there are still differences between them. Systems like that developed by the FAO attempt to merge these differences and highlight the similarities (Deckers *et al.*, 1998; Bot *et al.*, 2000).

High spatial variability is another major problem in the inclusion of soils in LCA studies. The scale of variation is such that units normally covered by an LCA study are bound to have significant differences in major soil characteristics, e.g. soil organic carbon content, inherent nutrient fertility, nutrient buffering capacity, and water absorption ability. This may well be one of the most difficult and specific aspects of soils-based LCAs. Focusing on soil erosion as an example, the impact of crop cultivation will depend on topography, (i.e. the location of the field or plot on the slope). Higher-located plots may affect growing conditions on lower plots when they are causing soil erosion. In a similar way, water extraction in a given plot close to a river may affect water availability in other plots farther from the river, or farther from the origin of the river. How should this be handled in LCA methodology? Are there different thresholds for those plots that are affected by other plots? Or, does the analysis move to a larger scale and evaluate the entire slope or watershed?

Other interactions may give very conflicting outcomes. These include higher yields that lead to soil water depletion, favorable N responses associated with high water demands, and nitrate leaching versus nitrous oxide volatilization.

In addition to problems arising from high soil spatial variability, there is also a temporal variation problem. Each year the impact of cultivation and harvest on the soil can be different as impacts are co-dependent on weather conditions. This phenomenon also impacts other elements of LCA analyses such as economic aspects. Consider, for example, allocation procedures depending on economic values of (co-)products. These will have temporal variability as well, and this is considered one of the main reasons for not applying allocation to economic values in LCA studies.

4.2.4.2 What is the way forward?

This publication addresses the environmental performance of bioenergy feedstock supply chains. The way forward relative to assessing soils impacts of bioenergy systems and the sustainability of biomass production rests with three approaches that could be used individually but are more likely to be employed in some combination. These approaches are: (1) utilizing characteristics that can be quantified in LCA studies by software, remote sensing, or other accounting methods such as GHG balances and energy balances; (2) measuring and monitoring ecosystem characteristics that can be evaluated in a more or less qualitative way (e.g. maintaining SOC) which might provide insights on potential productivity and sustainability; and (3) employing other proactive management characteristics such as Best Management Practices that are aimed at preventing environmental degradation.

Life Cycle Assessment has been used to estimate the environmental impacts of biomass energy uses. Typically this includes GHG emissions, CO_2 emissions, energy balances and some indirect effects. Cherubini and Strømman (2011) reviewed 94 LCAs, most of which were published in scientific journals. More than half of the studies were from North America and Europe, with others from South Asia, Africa and South America. About 50% of the studies limited the LCA to GHG and energy balances without considering contributions of bioenergy programs to other impact categories such as soils and water. Cherubini and Strømman (2011) concluded that there are a number of issues and methodological assumptions in currently used LCA approaches that make it impossible to quantify environmental impacts from bioenergy programs.

Some of the key indirect effects strongly depend on local operations, vegetation, soil, and climate conditions that make accurate assessment of environmental effects very difficult. Although policy makers claim that methods exist for assessing environmental impacts on soil and water, the scientific foundation for estimating indirect effects of bioenergy programs is constrained by the lack of adequate validation research, accurate assessment methods, and the relative infancy of the LCA process. Cherubini and Strømman (2011) clearly pointed out that determination of environmental outcomes of bioenergy production is complex and can lead to a wide range of results. They stated that indirect environmental effects are not ready for immediate incorporation into LCA methodology, but remain the next research challenge.

In regard to the second approach suggested above, soil quality monitoring was developed as a means of evaluating the effects of forestry and agricultural management practices on soil functions that might affect site productivity (Doran and Jones, 1996; Neary *et al.*, 2010). A number of soil physical, biological and chemical parameters which have linkages to soil productivity have been proposed as a minimum monitoring set for screening the condition, quality, and health of soils relative to sustaining productivity (Doran *et al.*, 1998; Burger *et al.*, 2010; Johnson, 2010). Evaluation of soil condition would lead to a time-trend analysis that could in turn be used to assess the sustainability of land management practices and bioenergy programs. However, even though sustainability is the stewardship goal of land management, more specific definitions of its goals and attributes are often complex and open to wide interpretation (Allen and Hoekstra, 1994; Moir and Mowrer, 1995). As noted earlier, many scientists have attempted to answer the “what”, “what level”, “for whom”, “biological or economic”, and “how long” questions of sustainability. Allen and Hoekstra (1994) clearly pointed out that there is no absolute definition of sustainability, and that it must be viewed within the context of the human conceptual framework, societal decisions on the state of ecosystem to be sustained, and the temporal and spatial scales over which sustainability is to be judged. In short, this approach is loaded with considerable uncertainty and lack of consensus.

Absent some breakthrough in validating a key set of soil parameters that will predict soil productivity and sustainability trajectories, the most sensible approach is the third, specifically the development, implementation, monitoring, and assessment of Best Management Practices (BMPs). A large number of BMPs for forestry and agriculture have been developed throughout the world because of national regulatory demands and the international development of codes of land management practice”. The BMPs in the codes and regulations cover traditional forestry and agricultural activities. New BMPs have been developed for bioenergy applications such as energy production facilities, ash recycling, and short-rotation cropping. Best Management Practices were originally developed in the 1970s for water quality protection but now extend to other environmental concerns such as sustainability. An important part of BMP utilization is the cycle of application, monitoring, evaluation, refinement, and re-application. Research and development studies play a key part in the refinement and communication of improved BMPs. Existing studies of BMP effectiveness have demonstrated that most BMPs, if applied correctly, are very effective in mitigating or preventing adverse soil and water quality impacts. Some jurisdictions have mandatory BMPs but others operate completely under voluntary systems.

The key components of successful BMP-based codes of practice for bioenergy systems, whether voluntary or mandatory, revolve around the cyclical strategy of planning, implementation, monitoring, evaluation, adaptation and renewed implementation. The minimum number of BMPs needed should come out of the planning process and is dependent on resources to be protected, site physical characteristics, regulatory requirements, and overall organization and operation goals. These will vary by site, region, country and organization. Life cycle analysis should always be included in order to identify all water and ecological impacts. The next step is crucial: monitoring and evaluation should be conducted routinely in order to decide if selected BMPs are effective and can be reapplied, or if they need to be modified, researched further or discarded. Research and development studies play a key part in the refinement and communication of improved BMPs. They are also crucial in validating the effectiveness of BMPs. This is especially important where local environmental conditions or operational standards are unique. Best Management Practices ensure that bioenergy programs can be a sustainable part of land management and renewable energy production.

4.2.5 Conclusions

Soils are crucial for profitable and sustainable biomass feedstock production. They provide nutrients and water, give support for plants, and provide habitat for enormous numbers of biota. Sustainability is the stewardship goal of crop production and forestry in general and specifically of forest bioenergy programs. Soils are an important, basically non-renewable resource, that provide the physical, biological, chemical, and hydrologic foundation upon which agricultural and forest bioenergy feedstocks grow. Thus sustainable soil management is a prime goal of forest bioenergy programs. Soils are able to renew themselves after being degraded but the time period might be several centuries or millenia, depending on climate and vegetation. Because of this long time factor, soils are considered to be non-renewable from the viewpoint of human use and management. Soils are heterogeneous and highly diverse components of ecosystems that form over long time periods under the influence of parent mineral material, climate, landscape position and biological activity. As the basis for bioenergy production, soils are a major factor in determining the sustainability of biomass energy systems.

4.2.6 References

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