THREATS TO SOIL QUALITY IN DENMARK

A REVIEW OF EXISTING KNOWLEDGE IN THE CONTEXT OF THE EU SOIL THEMATIC STRATEGY

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These reports primarily comprise research results and summaries from studies aimed at Danish conditions. The reports may also describe large research projects or act as handouts at meetings and conferences.

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The three main tasks in the Soil Framework Directive as related to the equally important and interacting issues of 1) soil management and weather impact, and 2) soil characteristics. The terms ‘disturbing agent’ and ‘system’ relate to the risk assessment concept, while the ‘pressures’ and ‘state and impact’ relate to the DPSIR concept.

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Preface

In 2006, the EU Commission launched a proposal for a Soil Framework Directive (SFD) with the purpose of protecting the soil resource across Europe. The proposal is based on the 'EU Soil Thematic Strategy', which incorporates the opinion of hundreds of experts, stakeholders, NGOs and politicians throughout Europe. It provides a comprehensive review of challenges, knowledge gaps and suggestions for actions. The SFD has not yet been approved by the Council. Despite the efforts of several presidencies, the Council has been unable so far to reach an agreement on this legislative proposal due to the opposition of a number of Member States constituting a blocking minority. The issues causing debate are mostly related to the identification and inventory of contaminated sites, which will not be dealt with in this report. The latest negotiations under the Czech Presidency (first half of 2009) have not changed this situation. The Swedish Presidency (second half of 2009) and the upcoming Presidencies (Spain and Belgium in 2010) will need to resume the discussions in order to make progress on this file as stated on the EU homepage for the Soil Thematic Strategy (http://ec.europa.eu/environment/soil/process_en.htm). It is, nevertheless, expected that the SFD will be approved in some form in the future, at least with respect to the threats to soil quality not related to soil contamination.

If the SFD is agreed upon, its implementation will require a number of actions from the Danish government. This includes the identification of areas at risk (priority areas), the establishment of risk reduction targets, and decisions on measures to reach these targets. Although the reports issued by the Soil Thematic Strategy contain a comprehensive collection of knowledge regarding these tasks, a national basis will be needed to evaluate which actions to be taken in Denmark.

This report was initiated by The Faculty of Agricultural Sciences, Aarhus University, in 2007, and it contributes to this basis by first suggesting an unambiguous use of the risk assessment concept. Next, it focuses on three of the threats that are of particular importance for Danish conditions: (i) compaction, (ii) soil organic matter decline, and (iii) erosion (by water and tillage). For each of these threats the report reviews and identifies national risk / priority areas, and suggests options for regulating management. The report also summarizes and strengthens the framework for analyzing the quality of soils as affected by management and weather and points out the main research gaps to be filled in order to successfully implement a future SFD if/when agreed upon.

University of Aarhus, Faculty of Agricultural Sciences

September 2009, Susanne Elmholt and Jørgen E. Olesen

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Sammendrag og anbefalinger

EU-kommissionen har fremsat et såkaldt Jordrammedirektiv (JRD) med det formål at beskytte Europas jorde mod forringelse. Forslaget er endnu ikke vedtaget i Ministerrådet (september 2009), men det må forventes gennemført i en eller anden form under de kommende formandskaber. Forslaget vedrører seks trusler mod jordens kvalitet og funktioner: a) erosion ved vind, vand (og jordbearbejdning), b) nedgang i jordens indhold af organisk stof, c) jordpakning, d) saltdannelse, e) jordskred, og f) forsuring. Implementering af JRD vil – ud fra de foreliggende udkast til Direktivet – sandsynligvis indebære følgende opgaver i hvert af medlemslandene: i) identificering af risiko- / prioritetsområder, ii) etablering af mål for nedbringelse af risici / identificering af acceptable risici, og iii) beslutninger om virkemidler og programmer til at nå de opstillede mål for nedbringelse af risici.

Denne rapport behandler de tre af de ovennævnte trusler, der vurderes at være af særlig betydning under danske forhold: 1) jordpakning, 2) nedgang i organisk stofindhold, samt 3) vanderosion og jordbearbejdningserosion. For hver trussel gives en vurdering af, i hvilket omfang truslen er et problem under danske forhold, og på hvilken måde, den pågældende trussel kan nedsætte jordens kvalitet og funktioner. Endvidere gennemgås for hver trussel en mulig fremgangsmåde for identifikation af risikoområder. Denne omfatter en eksplicit udpegning af påvirkningen (klimaet/dyrkningsmetoden) på den ene side og jordens sårbarhed for disse påvirkninger på den anden side. Endelig diskuteres muligheden for at opstille mål for nedbringelse af risici, virkemidler til at nå disse mål, samt afledte viden- og forskningsbehov.

I et selvstændigt kapitel gennemgås strategien for udpegning af risikoområder. De eksisterende dokumenter, som er udarbejdet af EU's instanser i forbindelse med og som opfølgning på EU's "Soil Thematic Strategy", vurderes at være ufuldstændige og tvetydige. Vi anser det som helt afgørende, at dette arbejde baseres på en tydelig differentiering af elementerne i konceptet risikokortlægning (risk assessment). Der er således behov for meget eksplicit at definere på den ene side de mekanismer, der påvirker jorden (dyrkningsmetoder og klimaet), og på den anden side jordens sårbarhed overfor disse påvirkninger. Vi diskuterer de klassiske termer tilknyttet risikokortlægning i dette perspektiv og forbinder disse med det såkaldte DPSIR-koncept, der ofte anvendes i arbejdet med miljøbeskyttelse.

Jordpakning. Jordpakning vurderes at være en alvorlig trussel mod jordkvaliteten i Danmark. Det hænger sammen med, at pakning af jord i dybder under ca. 0,4 meter har vist sig at være stort set permanent (irreversibel). Pakning i underjorden bestemmes først og fremmest af vægten af maskinerne. Det betyder, at de meget tunge maskiner, der anvendes i det højteknologiske danske land- og skovbrug, har stor risiko for at give en vedvarende skade på vigtige jordfunktioner som produktivitet, udvaskning af nærings- og forureningsstoffer samt emission af drivhusgasser. En gennemgang af den seneste forskning på området påviser, at hjullaster over ca. 3-4 tons indebærer stor risiko for at give varig pakning, hvis færdselen sker ved et vandindhold svarende til forårets afdræningstilstand. Da sådanne hjullaster meget ofte forekommer i dansk land- og skovbrug, vurderes det, at al dansk dyrket jord må betegnes som risikoområde for jordpakning. Forsigtighedsprincippet, der eksplicit indgår i JRD, medfører, at et mål for nedbringelse af risiko for jordpakning kan beskrives som trafik, der ikke giver anledning til plastisk (vedvarende) pakning af jordlag under 0,4 meter. Jord har en højere mekanisk styrke ved lavere vandindhold, hvorfor højere hjullaster end nævnt herover vil kunne accepteres under tørre forhold. I rapporten diskuterer vi muligheden for etablering af et beslutningsstøttesystem, der baserer sig på kendskab til jordtype, jorddybde og vandindhold (mekanisk styrke) på en given lokalitet i landet. En prototype for et sådant system er under udarbejdelse ved Det Jordbrugsvidenskabelige Fakultet, Aarhus Universitet, og vil kunne danne basis for et virkemiddel til regulering af trafik på jorden med henblik på at undgå permanente skader på underjorden. Der er imidlertid behov for yderligere viden om de kræfter, der virker i kontaktfladen hjul-jord, om transmission af kræfterne ned gennem jordprofilen, samt om relevante udtryk for jordens mekaniske styrke ved forskellige vandindhold. Disse forskningsbehov bør snarest muligt adresseres for at gøre de nævnte beslutningsstøtteværktøjer til pålidelige virkemidler på tværs af jordtyper og vandindhold.

Fald i indhold af organisk stof. I nogle egne af Danmark har jorden gennem de seneste årtier været dyrket ensidigt med enårige salgsafgrøder. Før 2. verdenskrig udgjorde permanente græsmarker en meget større del af landbrugsarealet og datidens sædskifter var mere alsidige. I samme periode er dræningen af de danske jorde forbedret og jordbearbejdningen er intensiveret. Samlet set betyder dette, at dansk landbrugsjord gennem de seneste årtier har oplevet et fald i indholdet af organisk stof, og der rapporteres jævnligt om problemer med jordstrukturen i forbindelse med jordbearbejdning til såbed på de mere lerholdige jorde. En undtagelse i den generelle tendens med nedgang i jordens indhold af organisk stof er sandede jorde med intensive/store malkekvægbesætninger, hvor en stor del af arealet er udlagt med flerårige græsmarker. I rapporten gennemgår vi den seneste forskning omkring dynamikken i jordstrukturdannelse. Disse studier viser, at jordens lerpartikler ved lave indhold af organisk stof nemt dispergeres (frigøres) til jordens vandfyldte porer med efterfølgende risiko for dannelse af indre 'skorper' i jorden. En sådan 'cementering' har dramatisk effekt på jordens evne til at smuldre og danne et godt såbed samt også andre vigtige jordfunktioner. En grundig gennemgang af den danske forskning på området peger på tilbageførsel af planterester – herunder øget brug af halmnedmuldning – og udvidet dyrkning af efterafgrøder som de vigtigste virkemidler til at opretholde jordens indhold af organisk stof. Internationalt er udstrakt brug af pløjefri dyrkning også fremhævet som et virkemiddel til beskyttelse af jordens organiske stof. Det forskningsfaglige grundlag for vurdering af dette virkemiddel under danske forhold er endnu spinkelt. Det er almindeligt accepteret, at der ikke findes een nedre grænseværdi for et tilstrækkeligt indhold af organisk stof på tværs af alle jordtyper og klimaområder. Arbejdsgruppen omkring nedgang af organisk stof i forbindelse med EU's "Soil Thematic Strategy" påpegede dette og foreslog identifikation af de relevante grænseværdier for alle kombinationer af jordtyper og klimaområder. Vi finder dette meget lidt operationelt og påpeger i rapporten to alternative strategier. For det første har nylig forskning vist, at en simpel indikator - beregnet som kvotienten mellem jordens indhold af ler og organisk kulstof - korrelerer til vigtige egenskaber ved jordstrukturen. I rapporten har vi kortlagt denne indikator for hele Danmark. Et sådant kort kan bruges til at udpege risikoområder for lavt organisk stof i jorden. Hvorvidt denne indikator kan danne basis for regulering af driftsmetoderne frem mod bæredygtige niveauer af organisk stof i jorden er dog ikke fyldestgørende afklaret. Der er således brug for yderligere forskning bl.a. omkring anvendeligheden i relation både til flere forskellige jordtyper men også mht biologiske og kemiske egenskaber og funktioner i jorden. Alternativt foreslår vi udpegning af ’dyrkningsmetodetærskler’ (engelsk ’management thresholds’), der er karakteriseret ved at give tilfredsstillende mængder af organisk stof i jorden på tværs af forskellige jordtyper. En dyrkningsmetodetærskel kan generelt defineres som ’den stærkeste påvirkning en given driftsforanstaltning må udvirke uden at medføre betydelige forandringer i en ikke bæredygtig retning’. I relation til organisk stof i jorden kan dette være bestemte sædskifter, en given andel grønne marker i sædskiftet, tilbageførsel af en given mængde afgrøderester eller anden tilførsel af organisk materiale, for eksempel i form af husdyrgødning, spildevandsslam eller kompost. Vi har allerede i Danmark et omfattende forsøgsmæssigt grundlag, som ville give et godt udgangspunkt for fastlæggelse af sådanne dyrkningsmetodetærskler.

Jorderosion. Erosion af jord kan forekomme med indflydelse af vind (vinderosion), vand (vanderosion) og jordbearbejdning (jordbearbejdningserosion). Vinderosion har historisk været af stor betydning i Danmark, men problemet er nu stort set løst via læplantning og en stor forekomst af vinterafgrøder. Dramatiske vanderosionshændelser, som det ses i udlandet, er sjældent forekommende i Danmark, der normalt betegnes som et lavrisikoområde for erosion. Ikke desto mindre har forskning og moniteringsprogrammer påvist, at vanderosion faktisk forekommer, især i perioder med langvarig nedbør i efterårs- og vinterperioden. I plot-forsøg på to jordtyper fandtes jordtab i størrelsesordenen fra 0,2 til 26 t ha^{-1} $år^{-1}$, og i et omfattende moniteringsarbejde på dyrkede danske marker fandtes en median-værdi på 0,7 t ha^{-1} $år^{-1}$, en 75%-fraktil på 1,9 t ha^{-1} $år^{-1}$ men en maksimalt observeret erosion så høj som 37 t ha^{-1} $år^{-1}$. Vanderosion forringer jordens dyrkningsværdi og medfører desuden eutrofiering af vandmiljøet. Nedbørens erosivitet er forholdsvis moderat i Danmark i forhold til andre dele af verden. I forbindelse med klimaændringerne forventes dog en øget erosivitet, især i efterårsperioden. Erodibiliteten – dvs. jordens sårbarhed over for erosion – er også moderat i forhold til nogle jordtyper i andre lande. Danske undersøgelser har vist, at dyrkningssystemet er afgørende for erosionens omfang ved et givet nedbørsmønster. Således er det vigtigt, at jordoverfladen er plantedækket og/eller dækket med afgrøderester, at der er et rimeligt højt organisk stofindhold i jorden for at sikre en god jordstruktur (høj infiltration af vand), og at pakning ikke har dannet lag med lav vandledningsevne. Selv om omfanget af vanderosion er forholdsvis beskedent og næppe på kort sigt har nogen betydende udbytteeffekt, kan effekten af mange års erosion skabe en trussel mod jordens kvalitet. Det foreliggende grundlag gør det ikke muligt at opstille et kvantitativt mål for et acceptabelt jordtab ved vanderosion. Jorderosion bør imidlertid modvirkes, fordi den selv i Danmark kan give risiko for markante miljøbelastende jordtab. Der foreligger en prototype for et beslutningsstøttesystem, der kan danne basis for et virkemiddel til regulering af dyrkning, målrettet en nedsat erosionsrisiko. Der er behov for forskning, der kan forbedre den kvantitative viden om effekten af langvarig erosion på jordens frugtbarhed. Ligeledes bør klimaændringernes påvirkning af processen følges nøje, bl.a. i relation til valg af fremtidige afgrøder.

Jordbearbejdningserosion er en proces, hvor variation i jordens flytning ved bearbejdning på skrånende arealer giver nettotab eller ophobning af jord lokalt inden for marken.

Jordbearbejdningserosion fungerer således som et effektivt transportbånd, der flytter jord fra bakketoppe til lavninger. Nylige studier i Danmark har vist, at jordtabet typisk er 20 t ha^{-1} $år^{-1}$, og ophobningen er i den samme størrelsesorden. Arealer med markant jordbearbejdningserosion er målt til at have mistet ca. 0,15 m jord over en 45 års periode. Processen reducerer produktiviteten på de eroderede arealer. Jordbearbejdningserosion forekommer overalt på kuperet terræn og medfører derfor en betydningsfuld jordforringelse for store områder i Danmark. Desuden giver dannelse af nyt organisk stof på de eroderede områder og tildækning af kulstofholdig jord i områder, der modtager materiale, anledning til en netto fastlæggelse af kulstof i jorden. Jordbearbejdningserosion er størst ved bearbejdning ned ad bakken. Som for vanderosion er det vanskeligt at fastlægge en grænse for et acceptabelt omfang af processen. Da der er tale om markante flytninger af jord, må jordbearbejdningserosion betegnes som en alvorlig jordforringelse i det lange perspektiv. Den mest effektive metode til reduktion af jordbearbejdningserosion er overgang til pløjefri bearbejdning. Desuden vil reduceret hastighed og dybde ved bearbejdning samt antallet af bearbejdninger påvirke processens omfang. Der er behov for at fastlægge den nøjagtige indflydelse på jordens produktivitet, på processens effekt på kulstoffastlæggelse samt for afgrødens vand- og næringsstofforsyning, herunder især N-omsætningen. Jordbearbejdningserosion kan medføre en større variation i afgrødens behov for næringsstoftilførsel, hvorved arealet i marken, hvor der sker over- og undergødskning, øges. Dette kan medføre mindsket ressourceudnyttelse i planteproduktionen. Der er behov for at udvikle et praktisk, interaktivt værktøj til at forudsige og kortlægge jordbearbejdningserosion på markskala ved forskellige bearbejdningsstrategier.

Samlet må det konkluderes, at der for de nævnte trusler mod jordkvaliteten er tale om processer, der reelt giver anledning til forringelse af jordressourcen i Danmark. Uanset skæbnen af et EU rammedirektiv er der behov for, at der iværksættes foranstaltninger til at modvirke disse forringelser. Såfremt et direktiv vedtages, kan nærværende rapport bidrage til at anvise løsninger af de opgaver, Direktivet vil stille de danske myndigheder overfor.

Summary and recommendations

The EU Commission has launched a proposal for a Soil Framework Directive (SFD) with the purpose of protecting the soil resources in Europe. The proposal has not yet been approved by the Council (September 2009), but it is anticipated that it will pass in some form following further negotiations. The proposal addresses six major threats to a sustained quality of soils in Europe: a) erosion by wind, water (and tillage), b) organic matter decline, c) compaction, d) salinisation, e) landslides, and f) acidification. The implementation of the SFD will most probably include three main tasks at the member state level: i) identification of risk / priority areas, ii) establishment of risk reduction targets / identification of risk acceptability, and iii) decisions on measures and action programmes to reach the identified risk reduction targets.

This report addresses three of the threats listed above that are considered the most relevant under the prevailing soil and climatic conditions in Denmark: compaction, soil organic matter decline, and erosion by water and tillage. For each of these threats we first document their relevance and geographic distribution for Danish soils. The damages to soil functions exerted by the specific threat are discussed. Next, a procedure is suggested for identifying areas at risk. This exercise involves an explicit identification of: i) the disturbing agent (climate / management) exerting the stresses to soil, and ii) the vulnerability of the soil to those stresses. Finally, for each threat, we discuss risk reduction targets, measures required to reach these targets, and the knowledge gaps and research needs to effectively cope with each threat.

One chapter is devoted to the strategy in risk area assessment. We find that the current guidance documents are too vague and lack a clear distinction between the disturbing agent on the one hand and soil vulnerability on the other. In our opinion, the recognition of this dichotomy in risk assessment is crucial. In this perspective, we discuss the classical terms involved in risk assessment and link them to the so-called DPSIR concept, which is often used to analyse and plan environmental protection measures.

Soil compaction is considered a severe threat to Danish soils due to frequent traffic with heavy machinery in modern agriculture and forestry. Compaction of soil layers deeper than approximately 0.4 m has been shown to be effectively persistent and to affect important soil functions such as productivity, leaching of nutrients, and emission of greenhouse gases. We review the most recent research and conclude that wheel loads higher than 3-4 tonnes are likely to create persistent compaction of the subsoil, if the soil is trafficked at typical spring soil water contents. Thus, all managed soils in Denmark are considered at risk of compaction solely because of the character of the ‘disturbing agent’. Based on the low or non-existing resilience to compaction of the deep subsoil, the risk reduction target suggested is that traffic that causes plastic soil deformation deeper than 0.4 m should be avoided. We discuss the potentials for setting up decision support systems including knowledge of soil types and water contents at any given position, depth and date in Denmark. A prototype of such a system is currently being established, which may serve to avoid persistent subsoil compaction in the future. There are still several weaknesses in our quantitative knowledge of the ‘chain of cause and effect’ in the compaction process. More research is needed on the mechanical stresses exerted in the wheel-soil interface as well as on the stress propagation in the soil profile. It is even more important to establish reliable estimates of the mechanical strength of different

soils at a range of water contents. Such research should be initiated as soon as possible in order to enhance the reliability of the above-mentioned decision support systems.

In some parts of Denmark, most of the agricultural land has grown annual cash crops continuously for decades. In former times, permanent grassland was a more common feature in the landscape, and the arable land included a more diversified crop rotation than today. Improved drainage of the fields and more intensive tillage systems may have enhanced decomposition of the inherent soil organic matter (SOM). Altogether, this means that the SOM content has declined for most of the agricultural land, and tilth problems ascribed to low SOM contents are frequently observed. An exception in the decline in SOM is sandy soils dominated by intensive cattle production systems that have a high frequency of grass in their crop rotation. In the chapter on SOM decline, we review the most recent knowledge on the dynamics in the creation and stabilization of soil structure. Evidence in the literature suggests that low contents of SOM may change the role of clay minerals in soil aggregation: clay particles disperse more readily in water and – when drying up – potentially form internal, cemented crusts rather than flocculate in an interaction with organic matter. This has dramatic effects on tilth properties, e.g. friability during tillage, the water content 'window' for tillage, and transport of water and air. A review of Danish research on soil use and management affecting the SOM content points to a range of options to counteract the SOM decline. These include recycling of plant residues, an expansion of the incorporation of straw, and a more frequent use of catch crops. No-till production systems are frequently used in other parts of the world and are claimed to induce a significant increase in the more stable SOM pools. The experimental results for Danish conditions on this issue are not yet sufficiently clear. It has long been recognized that there is no universal lower threshold for SOM that can support sustainable tilth conditions across all soil types. This was also acknowledged in the concluding report from the EU Soil Thematic Strategy (van Camp et al., 2004). They suggested an alternative procedure involving the identification of target SOM values for a number of well-defined regional soil units. These should be delineated on the basis of the important factors determining SOM levels in soil, namely a combination of climate type (which will vary according to geographical region and altitude), soil type (texture), and drainage. However, this would almost unavoidably lead to a confusingly large number of target SOM levels for Europe. In the chapter on SOM decline, we therefore examine two alternative approaches. One is based on a recent observation that the ratio between a given soil's content of clay and organic matter seems to constitute an indicator of important tilth characteristics across soil types. This clay/SOM ratio indicator has the potential to serve as a tool for the identification of risk areas for critically low SOM contents. However, more research is needed to confirm and validate its use in a soil protection context. This includes the use of other expressions of the mineral fraction active in aggregation (e.g. the specific soil surface area). In addition, the concept has as yet only been tested on soil physical functions and properties. We need research on soil biological and chemical properties as related to the clay/SOM ratio. We therefore suggest an alternative approach for setting up action programmes to minimize the risk of reaching critically low SOM contents: We define a *management threshold* as *the most severe disturbance any management may accomplish* *without inducing significant changes towards unsustainable conditions* and suggest the identification of sustainable ‘best management practices’ based on expert knowledge. More research is urgently needed to develop quantitative expressions of sustainable conditions regarding the SOM content of soils.

In a historical context, wind erosion has been a significant problem. However, today the extensive use of hedges in regions with sandy soils and of winter crops has diminished the problem. In Denmark, spectacular soil water erosion events are rare, and erosion risk is generally perceived to be low. Nevertheless, water erosion occurs on most soil types typically in autumn and winter after prolonged periods of rainfall, in connection with snowmelt and with rainfall on frozen soil. Soil loss in experimental plot studies on two soil types varied between 0.2 and 26 t ha^{-1} $year^{-1}$. An extensive, 5-year erosion survey on farmers’ fields across Denmark observed median soil losses of 0.7 t ha^{-1} $year^{-1}$, while the 75% quantile was 1.9 t ha^{-1} $year^{-1}$ and the maximum observed loss was 37 t ha^{-1} $year^{-1}$. Soil erosion is detrimental to soil quality because it truncates the soil and especially because it removes the fine material and nutrients. In addition, it contributes to eutrophication of the aquatic environment. The climatic or rainfall erosivity is rather low in Denmark compared to other parts of the world. However, due to the predicted future climate changes, higher erosivities are expected, especially in the autumn. The erodibility of most Danish soils is also moderate compared to common soil types in other parts of the world. Under Danish conditions, the cropping system, soil texture and low-permeable soil layers are the key parameters that determine soil erosion risk, while topography has only a minor influence. Although water erosion in Denmark is moderate and although it has primarily been addressed due to its contribution to pollution of the aquatic environment, we argue that the preservation of the long-term integrity of the soil resource ought to take precedence over a more short-term abatement strategy based on documented soil productivity loss. No quantitative threshold for acceptable soil loss can be set up, but water erosion ought to be prevented on agricultural land where it is likely to cause rill erosion. Measures to mitigate water erosion include management options that increase water infiltration and reduce detachment. This may be accomplished by increasing soil surface cover (growing crops or plant residues), maintaining a high SOM content and hence sustaining good structural conditions, and avoiding soil compaction. An existing prototype of an expert system for Denmark may be implemented as a web-based tool for erosion control and conservation planning. The impact of water erosion on long-term soil fertility ought to be quantified as the basis for risk assessment. We also need to assess the impact of climate change on water erosion risks.

Tillage erosion is a process whereby spatial variations in the magnitude of soil movement during tillage along a hillslope cause net gain or loss of soil locally within fields. Tillage erosion acts as a conveyor belt that moves soil from convexities to concavities. Next to land levelling, tillage erosion is the most severe process of soil redistribution in Denmark. Recent studies in Denmark have found soil loss from eroding sites of typically 20 t ha^{-1} $year^{-1}$, with corresponding deposition rates. Areas with maximum tillage erosion had lost about 0.15 m of topsoil over a period of 45 years. Tillage erosion is thus a very significant process of soil degradation in the parts of Denmark with a rolling topography. By massive soil truncation it decreases the productivity of eroding sites. In addition, the formation of new SOM at eroding sites and the burial of eroded SOM below plough depth provide an important mechanism of C sequestration on sloping land. The mouldboard plough is the primary tillage implement in Denmark and in general the most erosive. Tillage direction also exerts an important influence on erosivity. Tillage erosion rates are highest on steep slopes tilled in a downslope direction. As with water erosion, thresholds for critical soil truncation or soil burial have not yet been defined. However, since both tillage-induced soil loss and soil accumulation within fields are much more severe and widespread than for water erosion, tillage erosion must be considered a substantial long-term threat to soil productivity in Denmark. Defining risk reduction targets also requires the practicality and cost of mitigation strategies to be considered. The most effective measure in reducing this type of soil degradation is to convert from conventional to reduced tillage systems. Erosion is eliminated with a no-till management. Other options are reduced tillage speed and depth as well as the frequency of tillage operations. There is an urgent need to better understand the exact impact of these measures on soil productivity, SOM storage and nutrient cycling. There is also a need to develop a practical, interactive tool for predicting and mapping tillage erosion for different tillage scenarios at the field scale.

In conclusion, the three threats addressed in the present report are likely to induce significant degradation of Danish soils. Irrespective of the fate of the EU SFD proposal, it is relevant to initiate a programme of measures to counteract further degradation of soil quality. If the EU SFD is approved by the Council, the authors hope that the present report may contribute to the scientific basis for the future decisions on actions to be taken for Denmark.

1. Background and motivation

Soil is the most complex biomaterial on the planet (Young & Crawford, 2004). While best known for its role in providing nutrients and water in crop production, soil is fundamental for waste disposals, ground water purity and recharge, and climate impact. Soil regulates water and element fluxes between the atmosphere and aquifers and thereby controls their quality. Vital economic, environmental, and human health issues are linked to the functionality of soil. Blum and Santelises (1994) and Blum (1998) considered the functions and services of soil as related to human activity and grouped them into six categories. Three ecological uses are 1) the production of biomass, 2) the use of soils for filtering, buffering and transforming actions, and 3) the provision of a gene reserve for plant and animal organisms. Three other functions relate to non- agricultural human activities: 4) a physical medium for technical and industrial structures, 5) a source of raw materials (gravel, minerals etc), and 6) a cultural heritage. The current focus on climate changes has added a new, important function to soil: carbon sequestration in order to remove CO_2 from the atmosphere. Soils are thus extremely important for sustaining welfare of mankind. However, soil resources are under increasing pressure due to population growth and intensification of agriculture and forestry.

The general concern regarding the protection of non-renewable resources has brought about conventions on, for example, biodiversity, climate change and desertification. No similar global convention has so far been approved for soil. Nevertheless, the concern regarding a sustained soil quality is reflected in the final reports of major international conventions (the European Soil Charter, 1972, the Rio Convention, 1992, and the Kyoto Protocol, 1997). Several groups joined to promote a soil protection effort leading to a draft proposal for ‘Convention on Sustainable Use of Soils’ (Tutzing Project, 1998). Bouma (2004) predicted the need for a ‘World Soil Agreement’ as a binding treaty on the optimal use of soil resources on a global level. However, Wynen (2002) examined the possibility of creating a UN Convention on soil quality and concluded that the most realistic and appropriate approach would be a ‘Code of Conduct’ for soil management. The concern for a sustained soil quality also formed the basis for a major review on soil protection issues performed during 2002-2004 by the European Union. Hundreds of scientists, stakeholders and NGOs joined forces to create the ‘EU Soil Thematic Strategy’ (van Camp et al., 2004). Based on this comprehensive material, the EU Commission launched a proposal for an EU Soil Framework Directive (SFD) (COM, 2006), which is expected to be approved by the EU Parliament and the Council of Ministers in the near future.

The SFD identifies a number of threats to a sustained soil quality and lists a range of commitments for the EU member states. The present report considers these possibly upcoming obligations in a Danish context. The purpose of the report is to analyse to what extent the threats are relevant for Danish soils, and to identify the potential knowledge gaps that need to be filled in order to implement the SFD in Denmark. The report also includes an analysis of the tools and concepts best suited for identification of risk areas and measures to control the threats. Therefore, we also intend to initiate and contribute to a debate – not only in Denmark but also internationally – on the most relevant criteria and approaches to be used in relation to the SFD, a debate which was encouraged by Eckelmann et al. (2006).

2. The EU Soil Framework Directive (SFD)

The Thematic Strategy for Soil Protection consists of a Communication from the Commission to the other European Institutions, a proposal for a framework Directive (a European law), and an Impact Assessment. The Communication sets the frame (Commission of the European Communities, 2006a). It explains why further action is needed to ensure a high level of soil protection, sets the overall objective of the Strategy and explains what kind of measures must be taken. It establishes a ten-year work programme for the European Commission. The proposal for a framework sets out common principles for protecting soils across the EU (Commission of the European Communities, 2006b). Within this common framework, the EU Member States will be in a position to decide how best to protect soil and how to use it in a sustainable way within their territory. Further documents contains an analysis of the economic, social and environmental impacts of the different options that were considered in the preparatory phase of the strategy and of the measures finally retained by the Commission. Our report will exclusively address the suggested Directive.

The original SFD text has been changed during the political negotiations. The most recent draft for the Directive text was prepared by the Czech Presidency in June 2009 (Commission of the European Communities, 2009). In the following, we refer primarily to the original text from the Commission from 2006. This means that we throughout the report use the originally suggested term 'risk area', which is identical to 'priority area' in the most recent drafts of the Directive.

Chapter I of the Directive lists a range of soil functions that should be protected through the implementation of the SFD: a) biomass production; b) storing, filtering and transforming nutrients, substances and water; c) biodiversity pool; d) physical and cultural environment for humans and human activities; e) source of raw materials; f) acting as carbon pool; and g) archive of geological and archaeological heritage. This is a super-condensed expression of the soil functions discussed in the reviews provided during the work with the EU Soil Thematic Strategy (van Camp et al., 2004). It is in agreement with previous summaries of essential soil services (e.g. Blum & Santelises, 1994; Blum, 1998).

The SFD identifies six main threats to a sustained quality of soils in Europe (issue f added during the modification of the proposal):

a) Erosion by water and wind*
b) Organic matter decline
c) Compaction
d) Salinisation (accumulation of salts)
e) Landslides (downslope, moderately rapid to rapid movement of masses of soil)
f) Acidification by significantly decreasing the soil pH value

*Although the SFD text specifically mentions only these two processes of soil erosion, succeeding reports regarding the SFD includes soil erosion caused by tillage as well (e.g. Eckelmann et al., 2006).

Chapter II describes the procedure to be followed in the implementation of the SFD by member states. This includes

1) Identification of risk / priority areas (see section 4.6 for definitions)

2) Establishment of risk reduction targets / risk acceptability
3) Decisions on measures / action programmes to reach the identified risk reduction targets

Chapter III deals with soil contamination, including the inventory and remediation of contaminated sites. Chapter IV lists additional member state commitments in terms of raising the awareness of the importance of soil human and ecosystem survival, and of reporting the progress in the implementation of the SFD to the EU Commission. Finally, Chapter V explains the role of the EU Commission during the implementation phase of the SFD. This includes inputs to a technical Annexe listing common elements for the identification of areas at risk. This work has been initiated by the establishment of two EU-funded projects: ENVASSO, addressing the potential of using common indicators for soil quality, and RAMSOIL, addressing existing and potential procedures in the specific risk area assessments. In addition, a report on the latter issue has been produced by a group of experts as scientific and technical support for the European Commission's Joint Research Centre (Eckelmann et al., 2006).

3. Threats to and concern on soil quality in Denmark

In many respects, Denmark has been a pioneer when it comes to protecting the aquatic environment from pollution with plant nutrients from agriculture and urban point sources (e.g. Simmelsgaard & Djurhuus, 1998). In contrast, the soil itself has received negligible political attention. Nevertheless, many scientific studies have addressed a range of soil quality aspects. The concern regarding the losses of phosphorus from agricultural land led to research that gave some insight into soil erosion as a side-effect (e.g. Olsen et al., 1994; Schjønning et al., 1995; Sibbesen et al., 1996). The impact of management on changes in soil organic matter contents has been addressed in a number of studies related to soil fertility, nutrient turnover and losses to the environment, soil carbon storage and crop productivity (e.g. Bruun et al., 2003; Christensen, 1988, 1990; Christensen & Johnston, 1997; Christensen et al., 2009; Hansen et al., 2004; Heidmann et al., 2001; Kristiansen et al., 2005; Thomsen & Christensen, 2004). A number of projects performed within the context of the Danish Research Centre for Organic Farming allowed studies on the effect of soil organic matter decline and also indirectly revealed some information on soil compaction (e.g. Schjønning et al., 2000a, 2002ab, 2007; Munkholm et al., 2001b, 2002; Elmholt et al., 2000, 2008). Two recent projects on soil compaction gave new and important information on the magnitude of and the principles of transmission of stresses in soil (Lamandé et al., 2006ab, 2007; Schjønning et al., 2006ab, 2008; Schjønning & Lamandé, 2009; Lamandé & Schjønning, 2008ab, 2009abc).

The studies mentioned above and observations during years of research in related topics identify three of the threats focused on in the SFD as being highly important for Danish conditions: soil compaction, soil organic matter decline, and soil erosion (water, tillage). A detailed description of each of these threats – specifically for Danish soils – will be given later in this report.

In the late 1990s, the Danish Parliament funded a review on the soil quality in modern, industrialized agriculture. The output of this initiative was: 1) a report reviewing international efforts towards a worldwide convention on soils (Wynen, 2002), and 2) a book on soil quality as affected by a range of management impacts (Schjønning et al., 2004). The conclusions from these two parts of the Danish initiative agree in many respects: Soil is a very heterogeneous medium, and it is very difficult to set up common standards for a ‘good’ soil quality. For air and water, specific concentrations of e.g. toxic material can be set up as thresholds for acceptable conditions. Soil type differences as well as local climate and weather conditions make it impossible to define similar universal thresholds for soil properties. The recommendation of codes of conduct rather than general conventions (Wynen, 2002), and the emphasis on management thresholds rather than soil indicator thresholds (Schjønning et al., 2004) is in good agreement with the intention of the EU work with a Soil Thematic Strategy: a <u>framework</u> Directive that enables local conditions to play a major role in defining sustainable management.

The present report addresses three of the six threats mentioned in the SFD: soil compaction, soil organic matter decline, and soil erosion. Erosion may be driven by wind, water and tillage. Wind erosion has historically been a significant problem in Danish agriculture (Odgaard & Rømer, 2009). However, today the problem is practically solved by the extensive use of hedges and winter cover crops. Hence, only water erosion and tillage erosion will be dealt with in this context. This means that the main threats exerted on the cultivated land (primarily agriculture) are the subject of the present report. Contamination of the cultivated land may be an important issue too (e.g. in application of sewage sludge) but is not addressed in this report. Neither do we include the SFD tasks relevant for contaminated sites in urban areas.

4. Tools and concepts relevant for the SFD

4.1. Stability: Resistance and resilience

Evaluation of systems requires estimates of their stability when stressed or disturbed. Stability may express 1) the resistance to change in function or form during a stress event or 2) the capacity to recover functional and structural integrity (resilience) after a disturbance. It is important to distinguish between resistance and resilience. In population ecology, resistance is defined as ’the capacity to resist displacement from an equilibrium condition’, while resilience is defined as ’the capacity of a population (or system) to return to an equilibrium following displacement in response to a perturbation’ (Swift, 1994). In this presentation, we tend to follow Seybold et al. (1999) by using the term resistance instead of stability, which occasionally has been used to express the capacity of resisting disturbance (e.g. Kay, 1990). In this study, we find that stability is more appropriate as a common denominator for resistance and resilience. The term vulnerability (e.g. Kay, 1990) is closely linked but inverse to stability. A low stability means a high vulnerability. Jones et al. (2003) also used the term vulnerability. They suggested the term susceptibility to denote the inherent vulnerability (meaning resistance) to soil compaction given by the soil type, whereas vulnerability was to be used for the soil’s resistance towards compaction at any given water content (i.e. any soil strength).

Eswaran (1994) emphasized that soil resilience relates to either ‘performance’ or ‘state or structure’ of the system. The same applies to resistance. According to Eswaran, ‘performance’ refers to functions and processes in the soil, while ‘state or structure’ refers to the pedological composition of the material. The latter is analogous to the structural form (Kay, 1990), although Eswaran had a larger time span in mind than Kay. Thus, resilience relates to the ability of recovering functions as well as physical form.

A soil may exhibit a high resistance, but a poor resilience with respect to some specific property. This would, for example, be the case if subjecting a dry clay soil to heavy mechanical loads. The soil strength and thus its resistance to compaction is large. If, however, the ’structural form’ collapses, which would happen at a very high load, it would probably be associated with a compaction along the ’virgin compression line’ (Larson et al., 1980), and the resilience – the ability to recover - to such compaction effects is poor (e.g. Håkansson & Reeder, 1994). Alternatively, a soil may exhibit a poor resistance, but a high resilience for some attribute. A number of microbial soil functions show examples of this when subjected to, e.g., pesticide applications. Pesticides may cause response deficits of more than 90% and yet the soil function may return to its original level so quickly that the ecotoxicological effect can be regarded as insignificant when compared to natural stress effects (Domsch et al., 1983).

Although the stability of soil systems should be assessed both in terms of resistance and resilience, particularly the latter property deserves attention when evaluating soil quality in managed ecosystems. As any form of agriculture disturbs the original equilibrium of the native ecosystem, it is evident that resilience is a key parameter when judging the sustainability of agricultural systems. The concept of resilience was originally coined by Holling (1973) with emphasis on the persistence of relationships within a system. Resilient systems may show the capacity to occupy more than one state of equilibrium (Swift, 1994). Each state of equilibrium may maintain a qualitative structural and functional integrity, but the quantitative properties may differ among equilibriums. This dimension of the resilience concept is crucial when dealing with managed ecosystems. Any form of agricultural activity disturbs the original equilibrium of the native ecosystem, and soil resilience can be invoked to connote the ability of management to maintain the performance of the soil (Eswaran, 1994). This interpretation may be controversial but logical when dealing with managed ecosystems. Management is an integral part of the agroecosystem, and resilience should be related to equilibriums in the managed system, not the performance or state that would prevail in the original, native ecosystem (Blum, 1998).

Resilience has been defined from various points of view for various purposes (Szabolcs, 1994). One important aspect is the time scale. The rate of soil formation from the parent rock is extremely low as compared to the potential rate of soil loss in unsustainable agricultural systems (Lal, 1994; Pennock, 1997). Lal (1994) reviewed the estimates of rates of soil formation for a number of soil types and concluded that most soils can be considered a non-renewable resource within the human life span. However, a soil subjected to severe gully erosion may be judged resilient also to this disturbance if regarded in the context of geological time spans of hundreds or thousands of years. Thus, the time factor has to be considered when discussing soil resilience.

It should be emphasized that the expression of resilience has no meaning without an explicit statement of the agents, forces or effects (disturbance) facing the soil (Szabolcs, 1994). Blum (1998) discussed the potential ’disturbances’ and classified the corresponding ’type’ of resilience into three groups: 1) resilience to physical disturbances, 2) chemical resilience, and 3) resilience to biological disturbances.

Especially the term resilience is crucial in a soil conservation context. The compaction of the annually tilled topsoil may severely restrain a number of soil functions resulting in reduced yields. However, the topsoil has a high resilience to such impacts (through a combined action of soil biology, freeze-thaw and drying-wetting cycles, and tillage). In contrast, compaction of soil deeper than ~40-50 cm appears to be effectively permanent, i.e. the soil has a low resilience with respect to this impact (e.g. Håkansson & Reeder, 1994). There is no need to let the SFD address impacts on soil with a high resilience. Instead the full focus should be on low-resilience impacts.

4.2. Soil indicators

Soil quality assessment typically includes the quantification of indicators of soil quality. Such indicators may be derived from reductionistic studies, i.e. specific soil parameters obtained from different disciplines of soil science (e.g. Larson & Pierce, 1991). However, also descriptive indicators, which are inherently qualitative, can be used in assessing soil quality (Seybold et al., 1998; Munkholm, 2000). Soil quality indicators condense an enormous complexity in the soil. They are measurable surrogates for processes or endpoints such as plant productivity, soil pollution and soil degradation (Pankhurst et al., 1997). Herdt & Steiner (1995) and Carter et al. (1997) drew attention to situations where individual indicators show opposite or different trends.

Larson & Pierce (1994) and later Doran & Parkin (1996) realised the weaknesses in expressing soil quality information in single numbers, at least in comparative studies of soil management. As stated by Doran and Parkin, such indicators may provide little information about the processes creating the measured condition or performance factors associated with respective management systems. Thus, the interpretation of soil quality indicators requires experience and 'skill' of the researcher and/or soil manager. Doran (2002) realized that several soil quality indicators would be too complex to be used by land managers or policy makers. Hence, he suggested focusing on simple indicators, which have meaning to farmers. The use of indicators like topsoil depth and soil protective cover in a given management system was hypothesized to be the most fruitful means of linking science with practice in assessing the sustainability of management practices (Doran, 2002). Schjønning et al. (2000a) showed that quantitative soil mechanical properties derived by analytical procedures in the laboratory correlated well with qualitative behaviour of soil in the field. It seems important to evaluate such links when considering the use of soil quality indicators obtained by reductionistic studies in controlled environments.

Larson & Pierce (1991) suggested a minimum data set to describe the quality of a soil. This data set should consist of a number of indicators describing the quality/health of the soil. Using an analogue to human medicine, reference values for each indicator would set the limit for a healthy soil (Larson & Pierce, 1991). The use of indicators has been widely discussed in the literature on soil quality (e.g. Doran & Jones, 1996). Lilburne et al. (2002) and Sparling & Schipper (2002) presented achievements obtained in a New Zealand soil quality project. In contrast to most other soil quality assessments, they focused on a regional rather than a farm or field scale. Management was similarly addressed in terms of distinct land uses (e.g. arable cropping, dairy farms, conifer plantation). Much effort was allocated to identifying the most adequate indicators, and seven key parameters were chosen: soil pH, total C and N, mineralizable N, Olsen P, bulk density, and macroporosity (Sparling & Schipper, 2002). Lilburne et al. (2002) identified the difficult task of isolating the relevant target/threshold values of indicators. Sparling & Schipper (2002) acknowledged the problem in addressing satisfactorily all combinations of soil types and land uses. Generally, however, they found the approach useful to raise an awareness of soil quality issues with regional authorities, scientists, and the general public.

Based on the above, we judge that indicators *per se* as well as their thresholds may be relevant for some purposes. On the other hand, the vast number of soil types and agroecosystems pose a significant difficulty. Thus Seybold et al. (1998) and Sojka & Upchurch (1999) stressed the difficulty in dealing with the 18-20,000 soil series occurring in the USA. Compared to soils, the human species is well defined, and a body temperature of 37^{o}C is an established threshold for a healthy person regarding infectious diseases. Considering the diverse agricultural uses of soils (e.g. growing different crops with dissimilar soil requirements) and the different optima associated with each specific use, Sojka & Upchurch (1999) emphasized understanding rather than rating the soil resource. However, within a well-defined scenario, as for example research in agricultural management at one specific site or region, the quantification of soil attributes and the use of these as indicators of soil quality may be quite useful (e.g. Campbell et al., 1997).

4.3. Indicator threshold and management threshold

Threshold was defined by Smyth and Dumanski (1993) as 'levels beyond which a system undergoes significant change; points at which stimuli provoke response'. As an example, Smyth and Dumanski defined the threshold for erosion as the level (extent of erosion) beyond which erosion is no longer tolerable if sustainability is to be maintained. Gomez et al. (1996) adopted this definition and used the term threshold to denote the boundary between sustainable and unsustainable indicator values. Thus, thresholds are values of a variable beyond which rapid, often exponential, negative changes occur (Pieri et al., 1995).

A main issue in the EU Soil Thematic Strategy and the SFD is how to identify sustainable management. A crucial question is whether soil indicator thresholds relate similarly to management options across all soil types and climate conditions. Management cannot be addressed without evaluating soil attributes (i.e. indicators). This justifies the work on identifying the most relevant soil parameters to be addressed in the implementation of the SFD (ENVASSO project, www.envasso.com). On the other hand – as will be discussed in more detail later in this report – the focus in the comprehensive Eckelmann et al. (2006) report as well as in the RAMSOIL project (www.ramsoil.eu) seems to be biased in favour of soil quality indicators and at the expense of knowledge on management effects on soil conditions across soil types. Here we want to introduce the term management threshold, i.e. "*the most severe disturbance any management may accomplish without inducing significant changes towards unsustainable conditions*" (Schjønning et al., 2004). Soil acidity may serve as an example for comparing indicator threshold and management threshold: Here, soil pH is a soil quality indicator for which a threshold can be established, while the rate of liming (e.g. kg $CaCO_3$ ha^{-1} $year^{-1}$) required to retain the pH at a prescribed level represents the management threshold. Note that the management threshold may be constant across a range of soils with different indicator thresholds.

4.4. Sustainability and the precautionary principle

Smyth & Dumanski (1993) stated that '*Sustainable land management combines technologies, policies and activities aimed at integrating socio-economic principles with environmental concerns so as to simultaneously: (i) maintain or enhance production and services; (ii) reduce the level of production risk; (iii) protect the potential of natural resources and prevent degradation of soil and water quality; (iv) be economically viable, and (v) socially acceptable*'. Especially issue (iii) in the statement above has a direct relation to the SFD, but the other issues should be considered too when identifying policy measures for reaching the targets set up by the SFD.

According to the definition above, land management is not sustainable if it degrades soil (issue iii). As mentioned previously in this report, compaction of soil below a depth of approximately 40 cm is effectively permanent. Nevertheless, Lebert et al. (2007) suggested that this would be sustainable if the soil retained a saturated water conductivity of at least 10 cm per day (combined with fulfilment of a few additional indicator thresholds). The crucial question is now whether water conductivity and related simple physical expressions of soil quality provide a full and sufficient description of soil functioning. The historical experience from man's application of new technologies indicates that precaution is advisable (e.g. the use of DDT for combating insects in the 1950s). It might almost inevitably turn out as hubristic to claim that we are aware of all ecological functions delivered by soil, and that thresholds for water conductivity can be used for evaluating all functions.

The precautionary principle is a culturally framed concept that takes its cue from changing social conceptions about the appropriate roles of science (O'Riordan & Cameron, 1994). The concept is related to and interacts with the sustainability concept. The basic principles of the precautionary principle are (i) thoughtful action in advance of scientific proof, (ii) leaving ecological space, (iii) care in management, (iv) shifting the burden of proof, and finally (v) balancing the basis of proportionality. It is beyond the scope of this report to go into detail with this principle. Principle (i) is rather difficult to combine with natural sciences. 'Thoughtful action in advance of scientific proof' in the context of agricultural ecosystems means that management decisions should be based on a 'burden of evidence' when 'hard' data may not exist (O'Riordan et al., 2001). It is explicitly mentioned in the Executive Summary of the report on the EU Soil Thematic Strategy that the precautionary principle should be used in the implementation of the SFD (van Camp et al., 2004), and it may be wise to take such precautions at least when we are dealing with low-resilient (or even permanent) impacts.

4.5. Risk and risk assessment

The Executive Summary report of the EU Soil Thematic Strategy recommended the development of a generic conceptual framework for soil risk assessment and management (van Camp et al., 2004). Risk, risk analysis, and risk assessment are key terms and concepts in health risk assessment, chemical risk assessment and ecological risk assessment (e.g. Cohrssen & Covello, 1989; Suter, 2007; Barnthouse et al., 2007). It seems crucial to have common and clear definitions of the terms and concepts used in this field. The OECD (2003) made a comprehensive inventory of the meaning of terms used in risk assessment. According to this glossary, the core term 'risk' is defined as '*The probability of an adverse effect in an organism, system or (sub)population in reaction to exposure to an agent*'. We note that when addressing 'risk', we have to identify and address a disturbing *agent* as well as a *system* experiencing an adverse effect from that agent. Accordingly, OECD defines 'risk assessment' as '*A process intended to calculate or estimate the risk to a given target organism, system or (sub)population, including the identification of attendant uncertainties, following exposure to a particular agent, taking into account the inherent characteristics of the agent of concern as well as the characteristics of the specific target system*'. Again, it is emphasized that there are two parts of the concept: (1) the *agent* exposing (2) the *target system* by some adverse effect. According to the OECD (and other texts on risk assessment), the exercise of risk assessment includes four steps: i) hazard identification, ii) hazard characterization, iii) exposure assessment, and iv) risk characterization. 'Hazard' is defined as '*The inherent property of an* *agent or situation having the potential to cause adverse effects when an organism, system or (sub)population is exposed to that agent*' (OECD, 2003). We note that *hazard* thus is simply a differentiation of the characteristics of the disturbing *agent* used in the definitions above. The four steps included in the 'risk assessment' are further defined in the OECD glossary and may be consulted in OECD (2003).

4.6. Risk area identification and risk management

In the SFD, 'priority areas' are defined as '*areas where there is decisive evidence, or legitimate grounds for suspicion, that one or more soil degradation processes exceeding the level of risk acceptability is occurring or is likely to occur in the near future*'. 'Priority area' is the term used in the most recent version of the SFD proposal (Commission of the European Communities, 2009), but it is nearly identical to the definition of the 'risk area' used in the original SFD proposal. The only exception is the inclusion of the term 'risk acceptability' in the definition of 'priority areas'. From a first view, this is an important change from the initial definition of 'risk area'. However, as already discussed, the SFD includes a task for establishment of risk reduction targets. This is where the risk acceptability will be addressed.

Eckelmann et al. (2006) and Heesmans (2007) adapted the terminology described above in their follow-up reports on risk area assessment in relation to the SFD. However, the Eckelmann report did not actually use the concepts in their discussion of risk area assessment for each specific threat addressed in the SFD; and the RAMSOIL project did not use the concepts described by Heesmans (who participates in RAMSOIL) in their questionnaires on risk area assessment. Heesmans (2007) further seemed to confuse hazard identification with hazard characterization. This indicates that a strict adherence to the classical risk assessment terms may not be desirable when dealing with soil protection in the SFD context. This is, in general, in accordance with Suter (2007), who emphasized that there may be situations, where the classical risk assessment procedure is not practical. We thus find it appropriate to restrict the use of the risk assessment concept for the evaluation of

i) the disturbing agent (management, weather impact)
ii) the stability of soil to the specific agent
iii) the probability/frequency of exposure of the agent to the soil.

In essence, this is what is included in the concept of risk assessment and what was applied for each of the SFD threats by Eckelmann et al. (2006).

After having launched the SFD proposal, the EU Commission has taken initiatives to provide common criteria for implementation of the SFD by member states (e.g. the ENVASSO projects on selection of common soil indicators, the RAMSOIL project on harmonization of risk assessment methodologies, and the ESBN report on risk area identification (Eckelmann et al., 2006)). A denominator for these activities is a strong focus on soil characterization and soil mapping. This follows the bias of the preliminary list of common elements for the identification of risk areas (Annex I of the suggested SFD (Commission of the European Communities, 2006b), in which soil management information is given as 'Land use' along a range of specific soil indicators (e.g. soil density). However, as argued above, risk assessment includes a full and balanced focus of the disturbing agent as well as the exposed system. In the SFD context the disturbing agent is soil management and weather impact.

'Risk management' was defined by OECD (2003) as '*Decision-making process involving considerations of political, social, economic, and technical factors with relevant risk assessment information relating to a hazard so as to develop, analyse, and compare regulatory and non-regulatory options and to select and implement appropriate regulatory response to that hazard*'. So, risk management focuses on the disturbing agent in a risk assessment framework; the focus is on the disturbing agent and its regulation rather than on soil stability. We find it imperative that the implementation of the SFD includes risk management considerations from the very beginning. Conversely, Eckelmann et al. (2006) suggested a 'tiered' approach, where policy measures (regulation of disturbing agents) are addressed following an initial phase (Tier 1) of risk area identification. Our recommendation is based on the variability of soils and hence the basic difficulty in arriving at soil indicator thresholds. Management thresholds as defined in section 4.3 of this report may, in contrast, serve as practical solutions to fulfil the demand for sustainable conditions across a range of soil types. Our suggestion does not exclude a focus on soil parameters. However, in order to avert threats to soil quality, there is a demand for knowledge on soil stability as well as on the impact of management and weather. Soils at risk are soils with a low stability and/or soils that are severely stressed. Thus, the identification of risk areas has to be based on knowledge on the soil as well as on the impact of management. An approach emphasizing the monitoring of soil indicators without including the knowledge on management impact on these soil properties will be biased and not provide the most efficient route towards protection of European soils against degradation. We find our point convincing when considering the soil compaction threat for Denmark. There is no need to put a lot of efforts into identifying risk areas based on soil properties alone. The machinery used in Danish agriculture today (i.e. the 'disturbing agent',- the other part of the dialectic partners *agent* <-> *system*) exerts such high pressures on the soil that the wheel load carrying capacity (van den Akker, 2004) is often exceeded (Schjønning et al., 2006b). As this machinery is potentially applied to all fields in Denmark, all agricultural land is a risk area in terms of soil compaction (detailed discussion in Chapter 5).

4.7. The DPSIR concept

The so-called DPSIR framework has been adopted from studies of environmental problems to be used also as a tool in analyses and regulation of the cultivated land (Blum, 1998). The DPSIR approach distinguishes between Driving forces, Pressures, State, Impacts and Responses and facilitates identification of the mechanisms acting in the entire agricultural system (Fig. 4.1). Using again the analogue to human medicine, the study of soil (e.g. by using indicators) may help diagnose the patient, but the DPSIR analysis lends itself to definition of the most relevant cure. An example may help explain the concept. In soil compaction, the driving force (D) is the economic conditions in crop production: in order to minimize costs, progressively larger and more efficient machinery is used in the field. The pressure (P) is thus identified by heavy machinery. The state (S) in this example is a dense

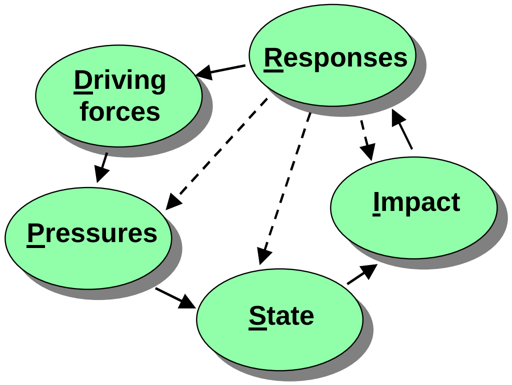

Figure 4.1. The DPSIR concept in its classical representation. Please consult the text for an example illustrating the concept in practice.

soil with constrained and reduced macropores. The impact (I) is reflected in a range of biological functions: reduced crop production, increased leaching of plant nutrients and production of greenhouse gases, etc. The response (R) may be directed towards the symptoms (e.g. increased application of nutrients to compensate for a reduced root system) or more reasonably towards the development of smaller machines, low-pressure tyres, etc.

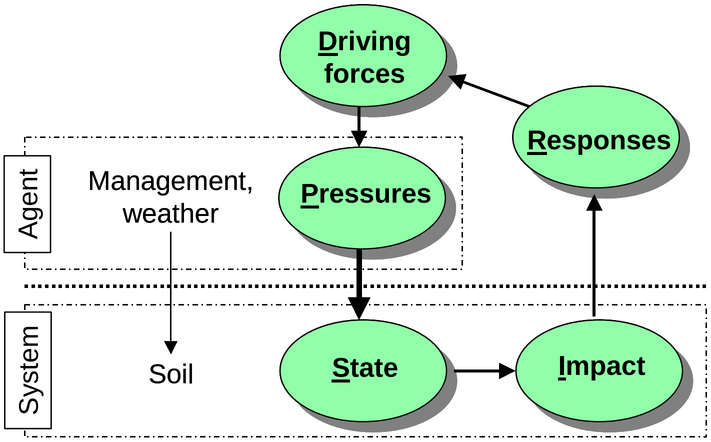

Figure 4.2. The DPSIR concept with focus on the relation to the aspects in risk assessment. 'Agent' denotes the disturbing agent in terms of soil management and weather impact, while 'System' denotes soil. For simplification only, not all potential response links are shown.

The EU Soil Thematic Strategy used the DPSIR concept when analysing the soil threats (van Camp et al. 2004). This yielded a much more focused approach than would have been the case if discussing the complex management <-> soil system without this organization. However, this conceptual framework would be even more valuable if explicitly combined with the risk assessment concept. In Figure 4.2, we have kept the five DPSIR elements but rearranged them in order to emphasize that the Pressure is analogous to the risk assessment term disturbing *agent*, which in an agricultural context is some kind of soil management and/or weather impact. Similarly, the Figure emphasizes that the State and the Impact are allocated to the affected *system*, i.e. the soil.

4.8. Suggested methodology for implementation of the SFD in Denmark

Eckelmann et al. (2006) arrived at three alternative approaches that may be used in the identification of risk areas:

1) A qualitative approach based on expert knowledge
2) A quantitative approach relying on measured data from inventories/monitoring
3) A model approach to predict the extent of soil degradation

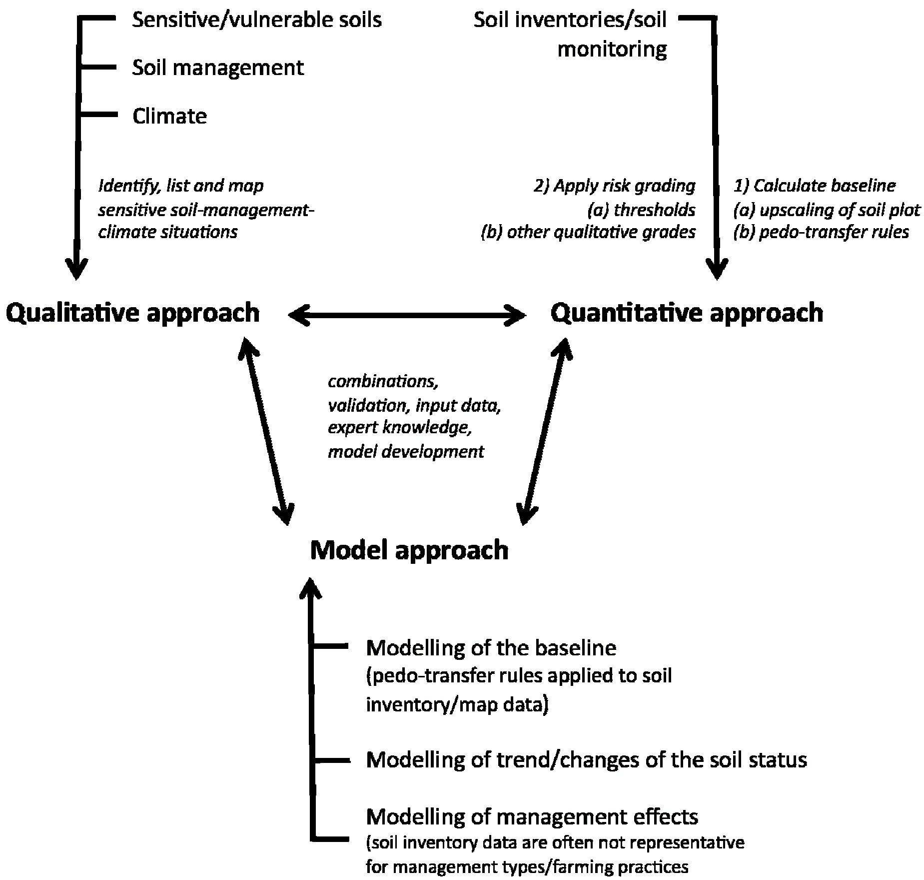

Figure 4.3. The three alternative approaches in risk area identification set up by Eckelmann et al. (2006).

Figure 4.3 describes the elements included in each of these approaches and indicates how the three approaches may be combined (Eckelmann et al., 2006). Our focus on management thresholds rather than on soil indicator thresholds seems to be identical to the qualitative approach of Eckelmann et al. (2006) However, modelling may well be a tool to identify management thresholds, and the effect of management regimes selected from qualitative assessments of existing knowledge may well be quantified in modelling.

As previously summarized, the SFD imposes three tasks to be solved in each of the EU member states (risk areas in the most recent drafts are labelled priority areas):

1) Identification of risk / priority areas
2) Establishment of risk reduction targets / risk acceptability
3) Decisions on measures / action programmes to reach the identified risk reduction targets

We recommend that the working process on these three aspects is undertaken simultaneously, and that the work embodies a continuous and explicit focus on the dualism / dichotomy in the problem: a soil may be at risk to some threat due to a high 'level' of the threat and/or due to a low stability (a high vulnerability) of the soil to that threat.

In the following sections, we shall address the three threats mentioned in the SFD that require special attention under Danish conditions: compaction, organic matter decline, and soil erosion (water and tillage). For each threat, we shall attempt to follow the DPSIR approach starting with an analysis of their state and the impact under Danish conditions. Next, we shall address the three tasks above as set up by the SFD by discussing the pressures (hazards) in terms of management and weather impact and review the knowledge for Danish soil types and climatic conditions. Probably the character of the interaction between the disturbing agent (soil management and weather) and the soil will turn out very different for each of the three threats. The specific situation for each threat will determine which approach will be suggested for 1) identification of risk areas, 2) establishment of risk reduction targets, and 3) decisions on measures to reach the identified risk reduction targets. Figure 4.4 summarizes our approach and highlights the interaction between soil management and weather impact on the one side and the soil on the other. This interaction should be explicitly included when addressing each of the three tasks. Figure 4.4 also emphasizes the links to the risk assessment concept as well as to the DPSIR concept.

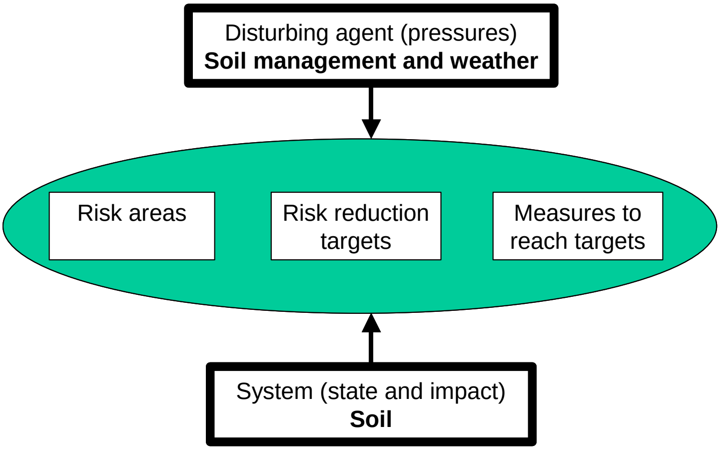

Figure 4.4. The three main tasks in the SFD as related to the equally important and interacting issues of 1) soil management and weather impact, and 2) soil characteristics. The terms ‘disturbing agent’ and ‘system’ relate to the risk assessment concept, while the ‘pressures’ and ‘state and impact’ relate to the DPSIR concept.

5. Soil compaction

5.1. Are Danish soils compacted?

The degree of compaction of Danish arable soils has not been subject to systematic monitoring. However, evidence exists that Danish soils generally have dense parts in the soil profile that can be related to the compaction effect of machinery.

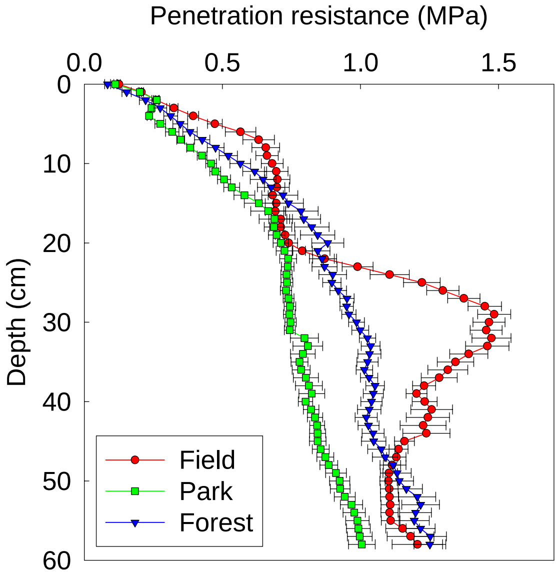

Figure 5.1. Cone penetration resistance measured at Barritskov Manor (loamy soil) in a non-trafficked park, a forest and in an agricultural field. Bars denote standard deviation (n=40) (Schjønning et al., 2000b).

Figure 5.1 shows the resistance to mechanical penetration of a metal cone through the upper ~60 cm of a loamy soil situated in the eastern part of Jutland (Schjønning et al., 2000b). These measurements took place at a soil water content about field capacity, i.e. it reflects the inherent soil strength at that reference water content. Three areas with different land use situated close to each other were included in the study. A park site provided reference data for soil never exposed to traffic by heavy machinery: the penetration resistance (PR) increases with depth as a consequence of the stress of the overlying soil. We note that the PR is generally higher for the forest soil than for the park. This may be due to the impact from stresses exerted by machinery occasionally used in forestry operations. The PR for the arable land is rather moderate in the top 20 cm layer, but increases dramatically below that depth. This shift is due to the frequent ploughing of the arable soil. The high strength of the layers beneath ploughing depth is interpreted as being due to mechanical stresses from machinery. Conventional ploughing implies that two of the tractor wheels are running in the open furrow, exerting a high stress impact directly to the soil at the ~20 cm depth. However, also traffic associated with for example slurry application and harvesting contributes to increased soil density and strength. The PR for the arable site is higher than for the park site for all depths investigated. The results in Figure 5.1 only give the status for a single location, but Håkansson et al. (1996) performed a similar investigation in Sweden for a total of 17 locations. The trend in that study – which is relevant to Danish conditions because of similar soil types and weather conditions – was exactly as discussed here.

Figure 5.2 shows the PR for the upper 60 cm soil of three fields on Sjælland with different production systems for decades (Schjønning et al., 2002b). All three fields display an increase in PR below ploughing depth. The increase is, however, much higher for the two fields with animal husbandry than for the cash crop field. This is ascribed to repeated traffic with heavy equipment to manage the forage production (application of animal manure and harvesting of forage crops in wet conditions late in the autumn). It is recalled that these results display case studies, where it is not possible to relate cause and effect directly.

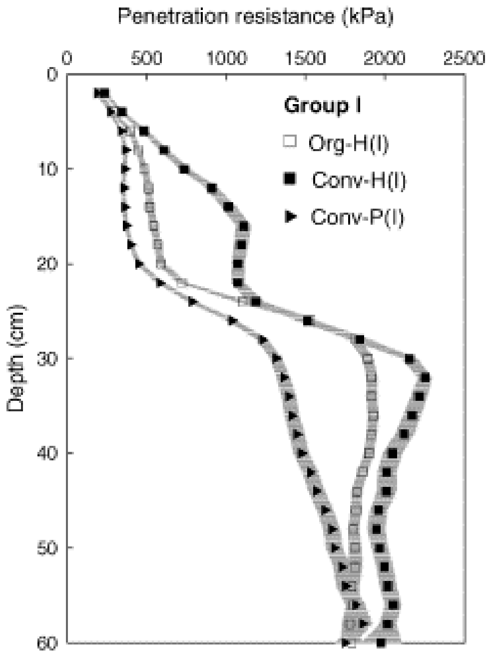

Figure 5.2. Penetration resistance (kPa) for the top 60 cm of three soils on Sjælland with different production systems. Org-H(I): Organic farming system with animal husbandry; Conv-H(I): Conventional farming system with animal husbandry; Conv-P(I): Conventional cash crop production. The shaded areas around the lines indicate +/- 1 SE of the mean. Reproduced from Schjønning et al. (2002b).

A number of other studies of Danish soils frequently exposed to heavy machinery reveal PR patterns that are similar to that of the arable soil shown in Figure 5.1 (e.g. Schjønning, 1991; Djurhuus & Olesen, 2000; Schjønning et al., 2002b). The dense layer of soil below tillage depth has also been observed in other studies of Danish soils (e.g. Schjønning, 1989, 1999;

Schjønning & Rasmussen, 1989, 2000; Munkholm, 2000; Munkholm et al., 2001a; Munkholm et al., 2005ab; Schjønning et al., 2005).

5.2. How is soil compaction harmful to soil functions?

As discussed in section 4.1, the concern regarding harmful impacts on soil should especially focus on low-resilience changes, which are persistent effects or effects where soil requires a long time to recover its functions. Håkansson & Reeder (1994) reviewed compaction effects reported in the literature and found that plastic deformation of soil layers below approximately 40 cm are effectively persistent. As also mentioned elsewhere, in our opinion the SFD should only address low-resilience functions. Compaction of the very top layer may be harmful to soil functions and may influence, for example, surface runoff, hence triggering soil erosion by water. Such effects are of course relevant for the SFD when considering erosion and should thus be addressed in that context. However, natural biological and physical processes as well as management (tillage) may easily alleviate such compaction effects. Hence, in this context we want to focus only on compaction effects in the subsoil. We define subsoil as soil below tillage depth. For most Danish soils, this corresponds to the depth of mouldboard ploughing (~20-25 cm).

5.2.1. Effects on soil productivity

The growth of plants is an integrating expression of a range of soil functions and processes. Crop yields may therefore indicate whether soil has been affected by some impact. A series of long-term field trials with subsoil compaction caused by heavy vehicles was carried out in an international collaboration between seven countries in northern Europe and North America (Håkansson, 1994). The number of experiments varied during the trial period. In the first four years of experimentation 22-24 field trials were included, while the number of active sites decreased to 14 in year 8 and only two sites remained in year 12. Similar experimental traffic treatments were applied in all trials at a field-capacity soil water content and on one occasion only. The treatments were 0, 1 and 4 passes track by track by vehicles with loads of 10 Mg on single axes or 16 Mg on tandem axle units. Tyre inflation pressure was 250-300 kPa. After the treatments, all plots in each individual trial were treated uniformly using vehicles with axle loads <5 Mg. Annual ploughing to a depth of 20-25 cm was performed in order to alleviate the compaction effects in the plough layer as quickly as possible. The crop responses to the traffic varied considerably between sites and years. For individual sites and years it was seldom statistically significant (Håkansson & Reeder, 1994). At an individual site, the effects sometimes disappeared for one to two years and then reappeared, which may probably be ascribed to different climatic conditions (Alakukku, 2000). Figure 5.3 (left) shows the mean crop response in plots with four passes track by track. During the first two years, crop yields were substantially reduced. A *t*-test was made to check whether the mean effect for the whole group of experiments from year 4 onwards was statistically significant. The mean crop yield reduction was 2.5%, and it was highly significant (Håkansson & Reeder, 1994). For the same period, the effect of one pass (not shown) was about 20% of that after four passes. In Figure 5.3 (right), the trend in yield response is ascribed to a plough layer effect (a), an effect from compaction of the 25-40 cm layer (b) and, finally, an effect attributed to compaction of the soil at >40 cm depth (Håkansson & Reeder, 1994). This interpretation of data is supported by a comprehensive Swedish dataset (Håkansson, 2005). It implies that plough layer effects last for five years, the 25-40 cm layer compaction effects are alleviated within a 10-year period, and that the compaction effects on layers deeper than 40 cm is persistent. The interpretation is based on the yield data of the international series but also supported by a range of other experiments (Håkansson, 2005). The idea of a persistent yield reduction effect is also based on compaction-induced increases in density of deep soil layers > 20 years after the compaction event as reviewed by Håkansson & Reeder (1994).

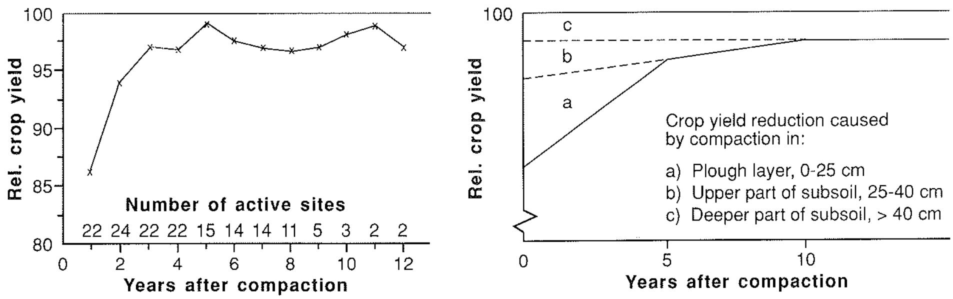

Figure 5.3. The results from a comprehensive international series of field trials with one initial soil compaction event (100 kN axle load [50 kN wheel load], four passes wheel by wheel). The Figures show the development in time of the relative crop yield in exact figures (left) and interpreted in relation to the compaction effect of different soil layers (right) (Håkansson & Reeder, 1994).

The international series of trials included two Danish experiments. They were located on a coarse sandy soil (Danish classification system: JB1) at Lundgård and a loamy soil (JB6) at Roskilde. Both were continued for eight-nine years after the compaction treatments. The crop was spring barley for nearly all years. At Lundgård, the trend in data resembled the pattern shown in Figure 5.3 (Schjønning & Rasmussen, 1994). At Roskilde, no significant compaction effects were found in any year. Averaged for the last four years of experimentation (interpreted as subsoil compaction effects only, see Figure 5.3), the four-time-replicated compaction treatment reduced the yield by 0.25 Mg (2.5 hkg) and 0.06 Mg (0.6 hkg) per hectare at Lundgård and Roskilde, respectively (Schjønning & Rasmussen, 1994).

The results outlined above relates to wheel loads of ~50 kN, which was considered an extreme load when the international experiment was planned in the early 1980s. In Danish agriculture anno 2009, higher wheel loads are generally used when applying slurry, and in the harvesting operations of most crops. Wheel loads as high as 120 kN are found for some sugar beet harvesters (please also consult section 5.3.1). Only a few studies have quantified the effects of such high wheel loads. Voorhees (2000) summarized a range of compaction experiments with high wheel loads in maize production. Wheel loads of 90 kN gave dramatic effects on the yield of maize in the first year after compaction. The residual effects interpreted as being due to persistent subsoil compaction were found to be 6% over an 11-year period for a clay loam in Minnesota and 12% for a clay soil in Quebec. In contrast, only minor effects on crop yield were observed in six long-term experiments with a self-propelled six-row sugar beet harvester loading 350 kN on four wheels (Arvidsson, 2001). More studies are therefore needed to give a more detailed picture of the mechanisms in soil compaction effects on crop growth. A decline in the yield of winter wheat in Denmark has been hypothesized to be partly due to subsoil compaction deriving from repeated traffic with heavy machinery during the last decades (Schjønning et al., 2009).

The potential impact of subsoil compaction on crop yield may be much more severe than deduced from average results of even long-term field trials. Compaction-induced poor drainage may reduce the number of workable days in the field, which in turn may affect the conditions for establishing the crop. Changing weather conditions may significantly influence the effect of compaction. Compacted subsoils may create anoxic soil conditions in wet growing seasons. A compacted soil may therefore suffer during a drought and in periods with surplus water. Alakukku (2000) found that wet growing seasons gave rise to higher yield reductions from subsoil compaction than did dry seasons. Compacted subsoils may restrict root growth, hence reducing the volume of soil exploited by the crop for water and nutrients. In a Danish study, a fine sandy soil (JB4) was trafficked by a slurry wagon with ~125 kN axle load (wheel load ~62 kN) at field capacity in March 1999. The field was then grown with spring barley in 1999 and ploughed in the autumn prior to establishment of winter wheat. The effective rooting depth of compacted soil measured in the growing season of year 2000 was decreased by up to 50 cm as compared to non-compacted reference plots (Andersen et al., 2004). This corresponded to a decrease in root zone available soil water of up to c. 90 mm and may also influence the uptake of plant nutrients. This effect was reflected in the results from the Finnish compaction experiment (Alakukku, 2000), where the compaction effect for all years was more pronounced for harvested nitrogen than for grain dry matter. As the crop was equally fertilized irrespective of compaction treatment, the decreased recovery of crop nitrogen probably indicates nitrogen loss to the environment (denitrification and/or leaching). The Finnish study also displayed a lower content of raw protein in the harvested crop (Alakukku, 2000).

Models are valuable for evaluating the compaction effects at different climatic conditions, including the effects of the expected changes in climate (e.g. Feddes et al., 1988; Lipiec et al., 2003). Future modelling may take advantage of the Least Limiting Water Range (LLWR) concept suggested by Da Silva et al. (1994) as based on an idea by Letey (1985) (Fig. 5.4). The LLWR concept reflects that plant response to varying soil water contents is least limited inside a range and most limited outside the same range. The water content related limitations are expressed through available water, soil aeration, and mechanical resistance. Four water contents limiting soil functions are identified: θ_{fc}, θ_{wp}, θ_{afp} and θ_{pr}, representing field capacity (fc), wilting percentage (wp), 10% air-filled pore space (afp), and 2 MPa penetration resistance (pr), respectively. Figure 5.4a shows how compaction reduced the LLWR of differently tilled clay loam topsoils (Betz et al., 1998). The LLWR is read as the water range between the two narrowest limitations. This may be done at the average bulk density (ρ_b,

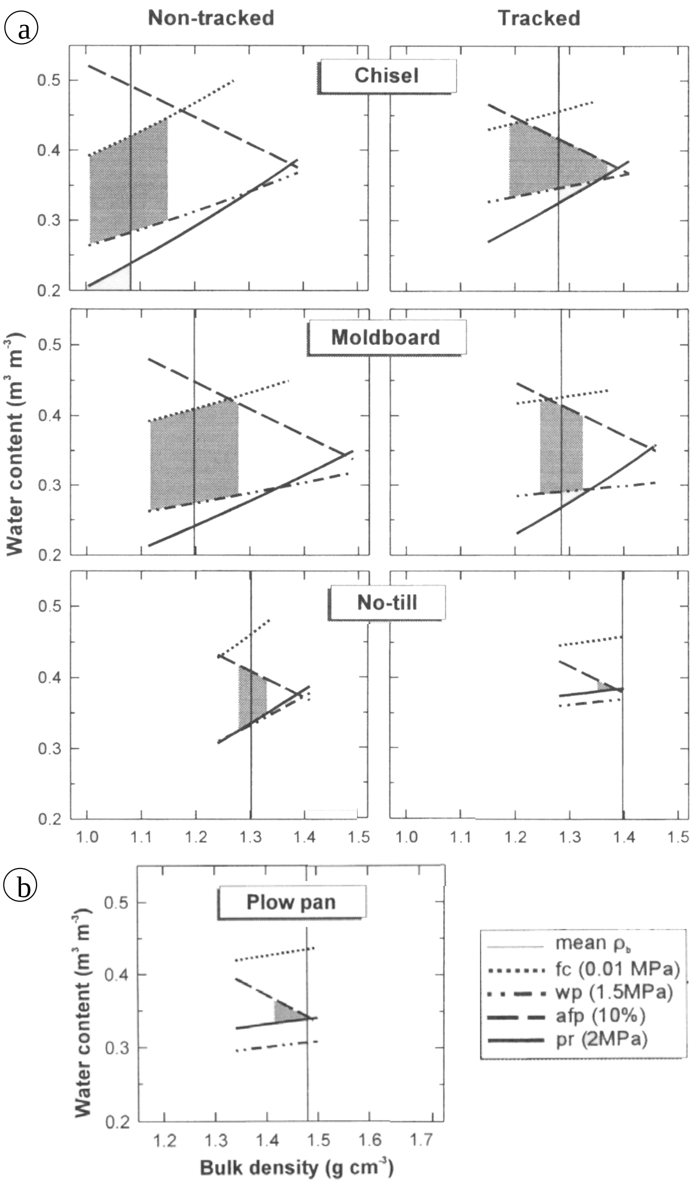

Fig. 5.4. The Least Limiting Water Range (LLWR) concept for (a) the topsoil and (b) the plough pan of a clay loam. Crop growth is constricted at water contents below and above the limits indicated by the shaded areas. Four limiting water contents for soil functions are identified: θ_{fc}, θ_{wp}, θ_{afp} and θ_{pr}, representing field capacity (fc), wilting percentage (wp), 10% air-filled pore space (afp), and 2 MPa penetration resistance (pr), respectively. Reproduced from Betz et al. (1998).

indicated by the vertical line in Figure 5.4a and b) or in the range of densities observed for each particular situation (the shaded areas). For the non-tracked part of the chiseled and moldboard ploughed soils, θ_{fc} formed the upper limit of the topsoil rooting environment, while for the tracked part of the same tillage systems, soil functions were limited at θ_{afp} (Fig. 5.4a). Note that the latter limit is active for the no-till soil irrespective of traffic. The lower water limit for optimal soil function in the no-till soil was a high mechanical strength, θ_{pr}, while it was the wilting percentage water content, θ_{wp}, that limited growth for the two other tillage systems (Fig. 5.4a).

Figure 5.4b shows the LLWR for the upper part of the subsoil for the same clay loam soil as in Figure 5.4a. This horizon was interpreted by Betz et al. (1998) as being highly compacted and was labelled a plough pan. Please note that the LLWR was close to zero at the average bulk density. The upper and lower water limits for optimal plant growth were determined by the 10% air-filled pore space and the root-restricting penetration resistance, respectively. The compaction effect is also evident from the huge difference between the limit given by θ_{fc} and that by θ_{afp} (Fig. 5.4b).

The data presented in Figure 5.4 is a very clear illustration of compaction effects on soil functions. Betz et al. (1998) considered primarily root growth when evaluating the LLWR. However, the 10% air-filled pore space suggested as a limit for adequate soil aeration for root growth (Grable and Siemer, 1968) has been shown also to indicate a threshold for aerobic turnover of organic matter in soil (Schjønning et al., 2003). Hence, the compaction effects on the LLWR given in Figure 5.4 may be interpreted also in relation to the fate of N in soil organic matter. An air-filled pore space below the 10% limit may trigger the production of greenhouse gases (N_xO) rather than nitrate (NO_3) for crop uptake. It is clear from Figure 5.4 that the range of water contents optimal for the key soil functions discussed (LLWR) may be very small for un-tilled topsoil and for the subsoil. Unsatisfactory conditions in the topsoil may be alleviated through a (temporary) change in tillage system, while the poor conditions obtained in the subsoil are much more difficult to manage.

5.2.2. Effects on other soil functions

The direct effect of compaction is a reduction of the pore volume and hence an increase in bulk density. Figure 5.5 shows the dry bulk density for the two Danish soil profiles included in the international series of subsoil compaction trials. The compaction impact is rather dramatic for both soil types. The increase in density was detectable to a depth of about 50 cm, although statistically significant only for soil layers above ~40 cm depth (Fig. 5.5).

Petersen et al. (2008) studied saturated hydraulic conductivity, K_S, and related soil properties in different horizons of a tilled sandy loam soil. They found an overall negative linear correlation between soil dry bulk density and log(K_S). However, for vertically sampled soil cores in the untilled subsoil, there were not even an indication of a relationship between log(K_S) and the bulk density. Although the variation in bulk density in such a study relates to many other pedogenetic factors than traffic-induced compaction, the result is very important: bulk density is not a primary regulator of saturated hydraulic conductivity in subsoils with continuous, vertical macropores. However, a compaction-induced increase in soil bulk density not only affects the macropores. High densities in the plough pan of a loamy sand soil were shown to include reduced volumes of >300 μm pores as well as smaller pores (Schjønning et al., 2005).

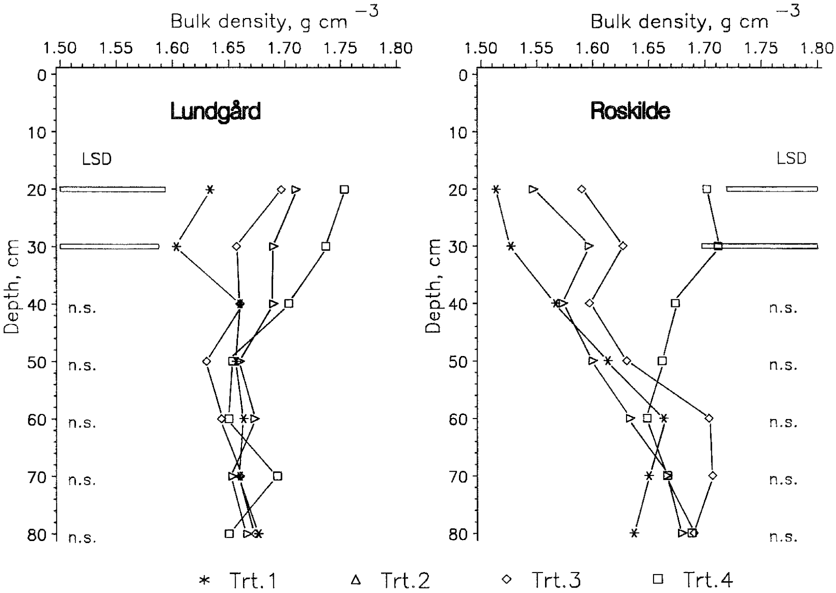

Figure 5.5. Soil bulk density measured with a gamma transmission technique one year after compaction treatments with a dump truck (~5 tonnes wheel load) for two Danish soils (Lundgård: coarse sandy soil; Roskilde: loamy sand). Trt. 1: control (no compaction); Trt. 2: one pass on topsoil; Trt. 3: four passes on topsoil; Trt. 4: four passes on exposed plough bottom. Reproduced from Schjønning & Rasmussen (1994).

A German study included four years of traffic on a loess soil with 10-20% clay (increasing with depth). Farm-realistic field operations took place with moderately large machinery (a maximum of 4 Mg wheel load and maximum 3 bar inflation pressure). Frequent measurements showed a significant decrease in the saturated hydraulic conductivity at 40 cm depth during the test period (Fig. 5.6). A regression model of all measurements estimated an initial hydraulic conductivity of ~5 mm h^{-1}, decreasing to approximately 1 mm h^{-1} after the test period (Semmel, 1993).

Iversen et al. (2007, 2008) analysed the saturated (K_S) and near-saturated hydraulic (k_{unsat}) conductivity of 500-800 undisturbed, large (20 cm height, 20 cm diameter) soil cores deriving from more than 200 Danish soil horizons. They developed pedo-transfer functions predicting K_S and k_{unsat} from basic soil parameters (e.g. soil texture) with root mean square errors (log(cm/d)) of 0.85-0.78 and 0.54-0.64 for the two parameters, respectively. In this study, K_S correlated negatively with bulk density, while the same trend for k_{unsat} was not significant. A low near-saturated hydraulic conductivity will increase the risk of preferential flow in macropores. A traffic-induced increase in soil bulk density may thus be expected to reduce near-saturated hydraulic conductivity and hence the frequency of events with by-pass flow in large macropores like earthworm channels if present after compaction at the investigated scale.

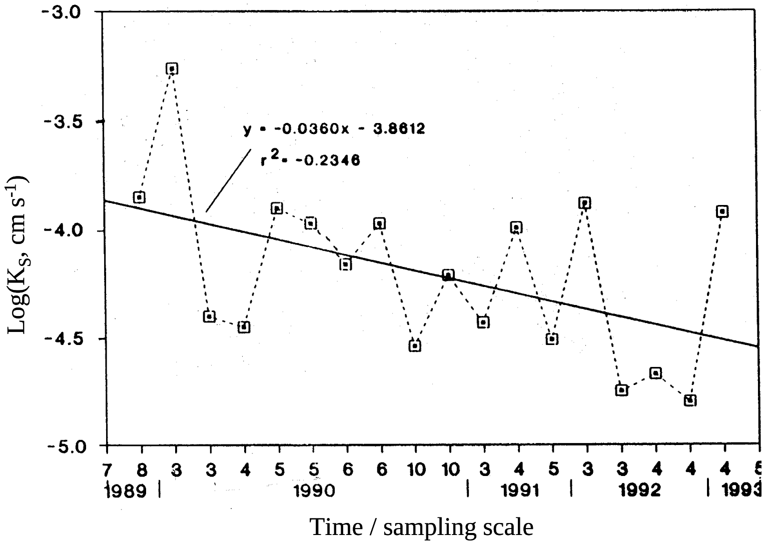

Figure 5.6. Saturated hydraulic conductivity, K_S, of soil at 40 cm depth as measured at intervals in a four-year period with heavy traffic. Please note that the abscissa is not a traditional time axis. (Semmel, 1993; Horn et al., 1995).

Preliminary results from an inter-Nordic research project (POSEIDON, http://www.poseidon-nordic.dk/) show a significant decrease in the volume of >~0.5 mm macropores as a lasting effect of field traffic. Soil cores were sampled in undisturbed condition at 20-40 cm depth in the spring 2009, fourteen years after four times repeated traffic with a heavy sugar beet harvester (Arvidsson, 2001). The macropores in the soil cores (20 cm height, 20 cm width) were made visible by CT-scans performed at a water content of field capacity. The volume of >~0.5 mm macro-pores on the images averaged 0.023 m^3m^{-3} (n=8) for the uncompacted reference soil, which was significantly higher than 0.015 m^3m^{-3} (n=8) for soil trafficked by a heavy sugarbeet harvester fourteen years prior to the sampling. Figure 5.7 shows a typical core collected in reference (not compacted) soil (left), and a typical core that had received traffic (right).

The increased risk of preferential flow for compacted soil was demonstrated by Kulli et al. (2003) with dye tracer studies following wheeling of the test soil. By-pass water flow has important implications because such flow patterns have been shown to facilitate transport of otherwise immobile pollutants such as phosphorus and pesticides to receiving water bodies (Jarvis, 2007).

Soil compaction reduces soil aeration (Czyz, 2004; also see Figure 5.4) and increases emissions of the greenhouse gas N_2O through denitrification at anaerobic sites (Bakken et al., 1987, Simojoki et al., 1991; Hansen et al., 1993). Poor root growth due to dense and poorly aerated soil may reduce crop yield – as discussed in the previous section – and reduce nutrient use efficiency and hence induce leaching of soil nitrogen. Soil compaction is therefore an important cause of many environmental and agronomic problems (flooding, erosion, leaching of agrochemicals to receiving water bodies, emissions of greenhouse gases, crop yield losses), which have significant impacts on ecological services and are expensive for society and the agricultural industry.

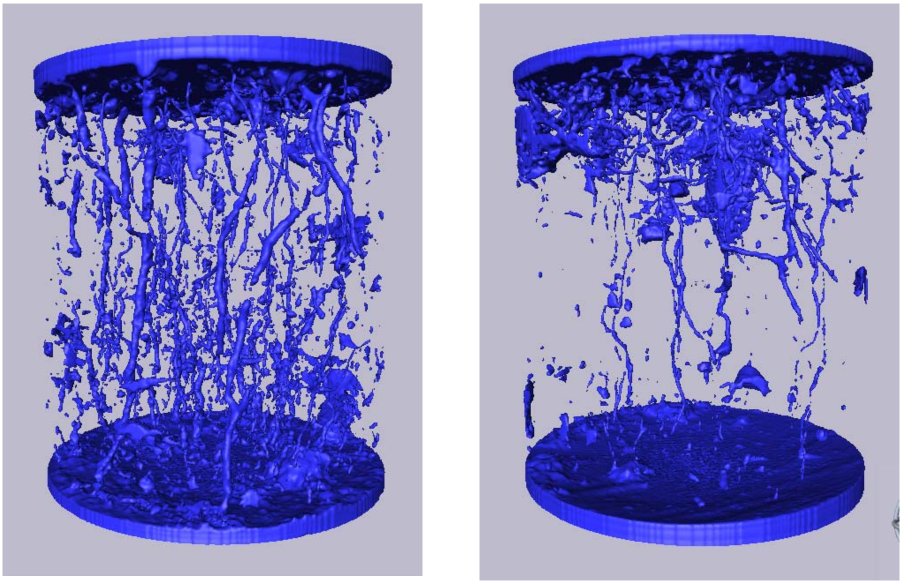

Figure 5.7. Macropores >~0.5 mm in undisturbed soil cores (20 cm height, 20 cm width) collected at 20-40 cm depth of a loamy soil in Skåne, Sweden, either with standard management (left) or trafficked four times by a heavy sugar beet harvester fourteen years prior to sampling (right). Preliminary results from the running inter-Nordic research project POSEIDON (http://www.poseidon-nordic.dk/). Digitized pictures created from Computer Aided Tomography (CT-scans) by Dorthe Wildenschild.

5.3. Identification of areas at risk of soil compaction

Risk areas were defined by the original SFD proposal as '*areas .... where there is decisive evidence, or legitimate grounds for suspicion, that one or more of the ..... soil degradation processes has occurred or is likely to occur in the near future*'. Therefore, two criteria exist for risk area identification for a specific soil: 1) An analysis of the status of compaction that indicates that soil compaction already has occurred and 2) An analysis of the relation between the pressures on the one side and the soil's reaction to those pressures on the other that indicates that the soil is likely to become compacted in the future.

Although we showed in a previous section that a range of Danish soils are already compacted, no systematic monitoring exists. Therefore, we are not able to systematically identify risk areas based on criteria 1) above. Criteria two involves an analysis of how soil (DPSIR elements *state* and *impact*, risk assessment element *system*; Figures 4.2 and 4.4) will react to the mechanical stresses exerted by machinery used in Danish agriculture today (DPSIR element *pressures*; risk assessment element *disturbing agent*).

5.3.1. The stresses exerted to soil by traffic on the cultivated land in Denmark

The weight of agricultural machinery has been increasing during recent decades and is expected to increase further, in line with the structural changes towards fewer and larger farms that need larger and more efficient machines (Kutzbach, 2000). The total weight of tractor and trailer in slurry application may exceed 60-65 tonnes (O. Green, personal information). Although such trailers are equipped with three axles, the load on these may reach 13-15 tonnes and the load on the rear tractor axle will be even higher. No commercially available slurry trailers on the Danish market have axle loads below 10 tonnes (Green & Nielsen, 2006).

Very high axle loads are also affecting soils in harvesting operations (cereals, potatoes, sugar beets). Several investigations have shown that loads even smaller than those mentioned will induce high stresses in the subsoil, exceeding the strength of the soil and hence causing persistent compaction (e.g., Arvidsson et al., 2001). The development in the size of Danish agricultural machinery is reflected in the tractors sold in Denmark in the period from 1995 to 2003 (Fig. 5.8).

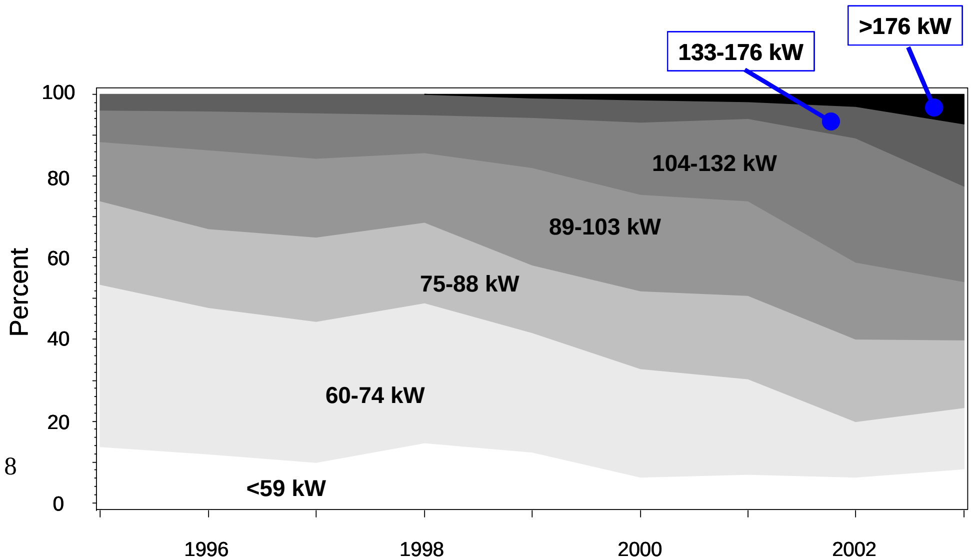

Figure 5.8. Relative sale of differently sized tractors for Danish agriculture in the period 1995-2003, for which comparable data is available. The numbers on the graph are engine effects for the particular size of tractor. Based on data from Danmarks Statistik.

The Danish climate is characterised by wet conditions in the autumn, winter and spring. Soils are most susceptible to compaction in wet conditions. Most traffic situations in arable agriculture therefore involve a high risk of soil compaction. Arvidsson et al. (2000) showed that the risk/probability of subsoil compaction with commonly used machinery in southern Sweden is 100% for spring slurry application and more than 60% after the 1st of October in sugar beet harvesting. Anthropogenic climate change is expected to worsen the problem:

winter precipitation in Denmark is predicted to increase 20-40% in the coming 100 years (Olesen et al., 2006). The largest combine harvesters used in Danish agriculture may carry up to 20 tonnes on the front axle. The precipitation pattern for Denmark in late summer months means that small grain cereal fields frequently have to carry these loads at harvest at water contents where soil is vulnerable to mechanical stress.

5.3.1.1. Stress transmission in the soil profile

Tyres used in agriculture today are much bigger than decades ago. As equipment has increased in size and mass, machine designers have increased tyre sizes to keep the surface unit pressure relatively constant. The concept of radial-ply construction allows reduced inflation pressures and better distribution of the stress in the tyre-soil contact area as compared to cross-ply tyres (e.g. van den Akker, 2003). Nevertheless, even these improved tyres are not able to carry the very high wheel loads at low inflation pressures. This means rather high ground pressures at large contact areas. According to the classical conceptual model for stress transmission in soil (Söhne, 1958), this means higher stresses in the subsoil than for smaller tyres with the same ground pressures (Olsen, 1994). Despite some experimental evidence (e.g. Smith & Dickson, 1990), the classical Söhne model has been

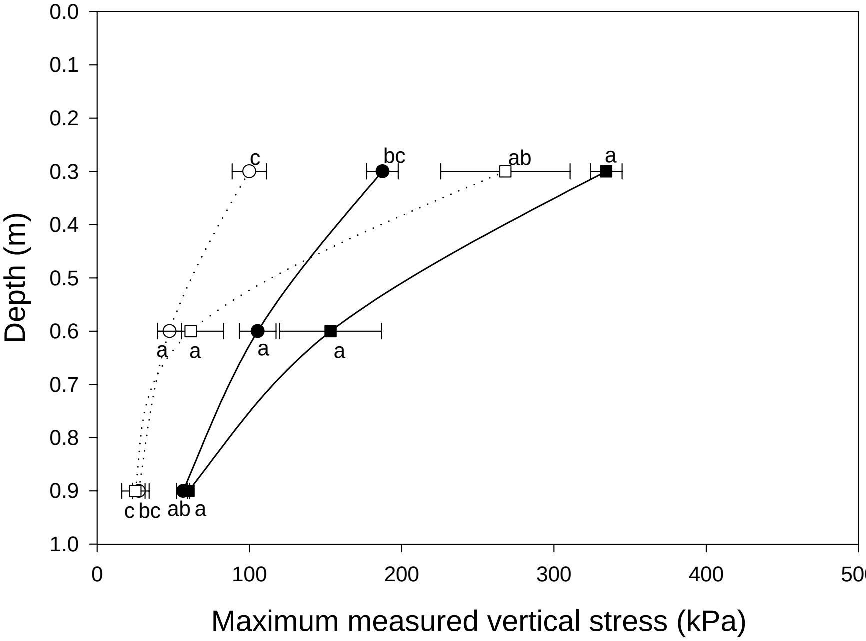

Figure 5.9. Maximum vertical stress measured in the soil profile (0.3, 0.6 and 0.9 m depths) for four compaction treatments in a Stagnic Luvisol at a water content of field capacity (white circles: 800/50R34 - 30 kN; black circles: 800/50R34 - 60 kN; white squares: 560/45R22.5 – 30 kN; black squares: 560/45R22.5 – 60 kN). Error bars represent standard errors. Different letters indicate significant differences between treatments (P=0.05). Reproduced from Lamandé & Schjønning (2009b).

disputed due to observations of prediction failure in some situations (Trautner, 2003; Trautner & Arvidsson, 2003). However, recent research in Denmark has validated the model (Lamandé et al., 2007; Lamandé & Schjønning, 2008b, 2009b).

Figure 5.9 shows the results of a recent, comprehensive investigation of stresses in the undisturbed soil profile in a Stagnic Luvisol (~20% clay) near Research Centre Foulum (Lamandé & Schjønning, 2009b). The soil at the test field had been annually ploughed for years but was not tilled in any way the most recent 1½ year prior to the investigation. A relatively small tyre (560/45R22.5) and a larger one with the same construction (800/50R34) were loaded with either 30 or 60 kN and inflated according to factory-recommendations for low-speed traffic in the field: 560/45R22.5, 30 kN load: 140 kPa; 560/45R22.5, 60 kN load: 340 kPa; 800/50R34, 30 kN load: 50 kPa; 800/50R34, 60 kN load: 100 kPa. The tyres were pulled across test plots, where stress transducers had been installed horizontally from pits at 0.3, 0.6 and 0.9 m depths. The tests were replicated two-three times for each combination of wheel size and load. The results clearly show that the stress in the upper part of the soil profile (~0.3 m depth) is best related to the tyre inflation pressure, while the stresses reaching deep subsoil layers (here 0.9 m depth) are primarily determined by the wheel load. The results in Figure 5.9 are important far beyond the scientific and academic aspects. They demonstrate that the use of big, wide tyres does not solve the problem of high stresses reaching deep subsoil layers when applying high wheel loads in traffic situations.

5.3.1.2. The effect of soil water content

It is common knowledge that a wet soil is more plastic than a dry soil. This means that at least for topsoils, much more soil deformation (compaction) is experienced when the soil is trafficked at wet conditions than at dry. Söhne (1953) anticipated that stress transmission in a wet soil profile will be more concentrated than when the soil is dry. Or in other words that stresses will be much more attenuated in a dry than a wet soil. Although the assumptions by Söhne are often used in modelling of soil compaction, ambiguous results for the so-called concentration factor have been published (Keller & Lamandé, 2009). Horn (1990) found that stress concentration increased with decreasing soil strength (e.g. at wet conditions). However, Trautner (2003) reported measurements that would suggest a rather opposite behaviour, i.e. that stresses were transmitted rather unattenuated the stronger the soil.

Figure 5.10 shows recent results from a Danish study, where vertical stress was measured at three depths in the soil profile and at three water regimes: wet-wet (spring water content), wet-dry (irrigated topsoil on a dry subsoil), and dry-dry (generally dry soil profile in the summer) (Lamandé & Schjønning, 2009c). The results support the assumptions presented by Söhne (1953): the wetter the soil, the more concentrated the transmission of vertical stresses in the soil profile. Very interestingly, the results also show that the stresses in the upper part of the subsoil are much higher at dry than at wet conditions. This means that despite a greater attenuation of vertical stress in the dry soil profile, the stresses reaching 0.9 m depth are actually even higher than when the soil is trafficked at more wet conditions. This may explain the contradictory results discussed above. But the high stresses reaching deep layers in dry soil are – at least for the Danish study presented here – not due to unattenuated stresses transmitted directly to deep layers as suggested by Trautner (2003). It is rather due to very high input stresses in the soil-tyre interface.

The results in Figure 5.10 are also very important for planning of field traffic with minimized soil compaction. The results actually imply that high wheel loads may be a problem for the subsoil also at dry conditions for the upper part of the soil profile. As already mentioned, today's combine harvesters may have wheel loads > 90 kN. The new results indicate that stresses reaching the deep subsoil may exceed the strength of the – often still quite wet – soil at that depth even when the harvesters operate on a dry topsoil (please consult Section 5.3.3. for a detailed discussion of the stress-strength relation). More studies are urgently needed on this aspect of the soil compaction problem.

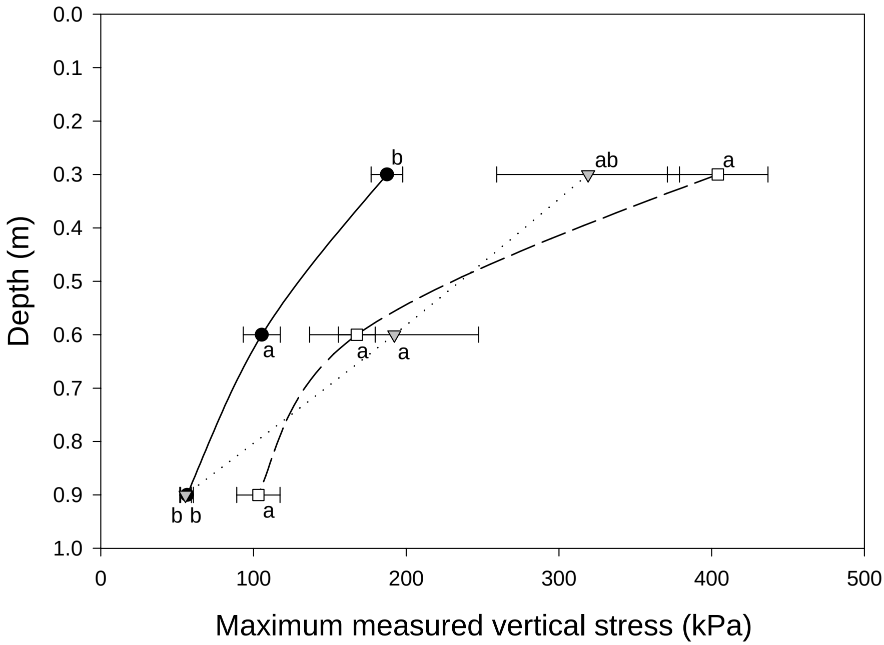

Figure 5.10. Maximum vertical stress measured in the soil profile (0.3, 0.6 and 0.9 m depths) for compaction treatments in a Stagnic Luvisol at three contrasting water regimes (black circles: 'wet-wet', i.e. field capacity [the soil water content in the spring]; gray triangles: 'wet-dry', i.e a summer-dry soil profile with increased topsoil water content after irrigation; white squares: 'dry-dry', i.e. summer-dry soil profile). The test tyre used was a radial ply 800/50R34 loaded with 60 kN. Error bars represent standard errors. Different letters indicate significant differences between treatments (P=0.05). Calculated from data of Lamandé & Schjønning (2009c), where more information on test conditions and exact water contents can be found.

5.3.1.3. Stresses exerted by wheels in the soil-tyre interface

Until recently a main bottleneck in the chain of cause and effect for quantification of stresses reaching specific soil layers has been the lack of realistic data for the stress distribution in the tyre-soil interface. Surprisingly few measurements of these stresses have been conducted. In consequence, even the subtle SOCOMO model for stress transmission in soil created by van den Akker (2004) calculates the stress distribution in the contact area by a standard parabolic formula applied to the mean ground pressure. This parameter in turn is calculated by a rule of thumb not justified by experiments although repeatedly stated in textbooks on agricultural soil mechanics.

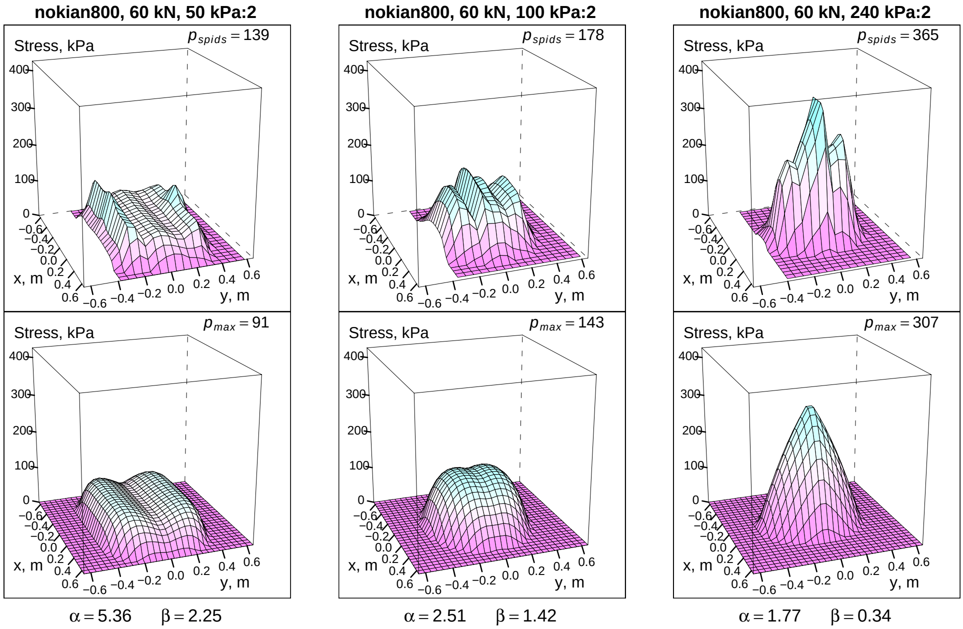

Figure 5.11 Measured (top) and model-fitted (bottom) stress distribution in the contact area between tyre and soil for an 800/50R34 tyre loaded with ~60 kN wheel load and tested at three different inflation pressures. Driving direction is from front to back in the Figures. Selected, individual field tests. p_{peak} is the measured maximum stress, p_{max} is the model-fitted average stress in the part of the of the soil-tyre interface reaching maximum stress, and α and β are parameters describing the stress distribution. Reproduced from Schjønning et al. (2006b).

However, recent research has significantly improved our knowledge of the stresses transferred from agricultural wheels to the soil surface (Keller, 2005; Schjønning et al., 2006ab, 2008; Lamandé & Schjønning, 2008a; Schjønning & Lamandé, 2009). Figure 5.11 shows the stress distribution near the soil-tyre interface for an 800/50R34 radial ply tyre when loaded with ~60 kN and tested at three different inflation pressures (Schjønning et al., 2006b). The tests were performed on a loamy sand soil at a water content slightly below field capacity. The upper row of figures shows the measured distribution, while the lower row of figures shows the stress distribution when fitted with a recently developed model (named FRIDA) and fitting the measured data with a combined exponential (perpendicular to the driving direction) and power-law (along the driving direction) function (Keller, 2005; Schjønning et al., 2008).

Tabel 5.1 gives predicted data for vertical stress in soil exposed to a range of different and differently loaded tyres (Schjønning et al., 2006b). The p_{max} is an output from the FRIDA model and yields the averaged maximum stress in the contact area near the tyre-soil interface. The predictions of vertical stress at 20 and 100 cm depth, in turn, were calculated by the Söhne model, using the FRIDA-fitted stress distribution in the contact area as input. Given the nice fit of the FRIDA model to measured data (Schjønning et al., 2006b, 2008; Fig. 5.11) and the recent validation of the Söhne model, the predictions in Table 5.1 are expected to be rather exact stress estimates.

Table 5.1. The model-fitted maximum stress in the tyre-soil contact area, p_{max}, and the Söhne-predicted max. stress at 20 and 100 cm depth for 20 combinations of wheel load, tyre type, and tyre inflation pressure. The figures are arithmetic averages of individual predictions from each replicate test in the field. For the 60 kN wheel load, analyses of variance for the two tyres 650/65R30.5 and Nokian 800/50R34 estimated the tyre inflation pressure effect on the stress estimate at 100 cm depth. Figures labelled with the same letter within each of these tyres are not significantly different (P<0.01). Reproduced from Schjønning et al. (2006b).

Tyre	Wheel load	Tyre inflation pressure	p_{max}	Stress, 20 cm depth (Söhne)	Stress, 100 cm depth (Söhne)
	kN	kPa		kPa	
Euroband SA 385/65R22.5	30	270	330	208	21
Nokian ELS Radial 560/45R22.5	30	140	181	142	20
Trelleborg TWIN 700/50-26.5	30	50	95	87	19
Mich. CargoXbib 650/65R30.5	30	50	91	66	19
Nokian ELS Radial 800/50R34	30	50	77	64	19
Michelin XEOBIB 650/60R38	30	70	117	92	21
Kleber Topker 650/75R38	30	50	115	101	21
Euroband SA 385/65R22.5	60	550	566	377	42
Nokian ELS Radial 560/45R22.5	60	340	375	282	41
Trelleborg TWIN 700/50-26.5	60	130	218	187	38
Mich. CargoXbib 650/65R30.5	60	50	107	69	30[a]
Mich. CargoXbib 650/65R30.5	60	100	144	129	36[b]
Mich. CargoXbib 650/65R30.5	60	240	290	235	41[c]
Nokian ELS Radial 800/50R34	60	50	97	70	32[a]
Nokian ELS Radial 800/50R34	60	100	142	123	37[b]
Nokian ELS Radial 800/50R34	60	240	299	238	41[c]
Michelin XEOBIB 650/60R38	60	200	254	212	40
Kleber Topker 650/75R38	60	140	207	184	40
Mich. CargoXbib 650/65R30.5	83	200	265	219	54
Nokian ELS Radial 800/50R34	83	200	224	205	52

It appears that the Söhne-predicted stress at 20 cm depth is rather closely related to the FRIDA-predicted maximum stress, P_{max}, in the contact area (Table 5.1). P_{max} in turn correlates well to the tyre inflation pressure. In contrast, the main predictor of stress reaching 100 cm depth seems to be the wheel load, which is in accordance with the results presented in Figure

5.9. Please consult Section 5.3.3. for a detailed discussion of the relative effects of tyre inflation pressure and the wheel load.

The FRIDA model requires six parameters to give a full and quantitative description of the stress distribution: three parameters relating to the contact area (super ellipse half-axes *a* and *b*, and the super ellipse shape parameter *n*), two parameters yielding the shape of the stress distribution (α for the driving direction; β across the driving direction), and finally the wheel load, F_{wheel} (Schjønning et al., 2008). Schjønning et al. (2006b) showed that the parameters *a*, *b*, *n*, α, and β may be predicted for a given tyre from the physical dimensions of the tyre, the actual inflation pressure, the recommended inflation pressure, and the wheel load. This means that it is possible to calculate more realistically the stresses acting in the tyre-soil interface simply from the physical dimensions of the specific tyres in question. This, in turn, provides the basis for realistic modelling of the stresses transmitted through the soil profile. The SoilFlex model developed by Keller et al. (2007) already includes FRIDA-like calculations of the stress distribution in the contact area and uses these as input in a Söhne-modelling of the stresses in the soil profile. A recent Danish project provided predictions of contact area stress distributions and stress transmission in the soil profile for a range of commercially available tyres for use in agriculture (www.planteinfo.dk, issue 'Jord').

5.3.2. The ability of Danish soils to withstand mechanical stresses

The stress exerted by the machinery on the soil has to be counteracted by the mechanical strength of the soil. The concept of pre-consolidation stress originated in civil engineering soil mechanics in relation to the slow consolidation of saturated homogenized clayey soils. In agricultural research, the concept is applied to quick compression of unsaturated soils simulating the loading by soil running gear and wheels, and it is most often labelled pre-compression stress, P_c (e.g. Dawidowski & Koolen, 1994). It marks the transition from the elastic to the plastic compressive behaviour of a soil. Principally, by limiting the imposed stress to below P_c, the risk of soil compaction (i.e. plastic deformation) and undesirable changes to soil structure could be minimized (Dawidowski & Koolen, 1994; Horn & Lebert, 1994).

Uni-axial confined compression tests were applied to undisturbed soil cores sampled in the 30-40 cm depth for a range of Danish soils in the context of an EU-funded project in the 1980s (Schjønning, 1991, 1999). The tests were used to assess the compression index, C_c, indicating the susceptibility to compaction when P_c was exceeded. P_c was not quantified because of uncertainties in the classical procedure of calculation (Casagrande, 1936). Gregory et al. (2006) provided a reproducible calculation procedure that was applied to data for one of the Danish locations: a loamy soil at Thisted in Jutland (Danish classification: JB7; international classification: Albic Luvisol). P_c was estimated to 96, 97, 110, 190, and 473 kPa for each of the matric potentials -50, -75, -100, -160, and -300 hPa, respectively (Gregory et al., 2006). Six replicate soil cores were used at each matric potential. It appears that P_c increased with decrease in matric potential (decrease in water content). In the wet conditions that may be related to a subsoil at field capacity (-50 and -75 hPa matric potential), P_c was slightly less than 100 kPa. In comparison, Gregory et al. (2006) calculated a value of in average 144 kPa for a clay soil drained to a comparable matric potential –60 hPa (Gleyic Cambisol in Switzerland; data supplied by M. Berli). Nissen (1999) found P_c to range from 50 to 95 kPa for a wide range in textures of German soils when drained to -60 hPa. For the same soils, the range changed to 66-107 kPa for cores drained to -300 hPa matric potential. We note that the increase in P_c with decrease in matric potential was much less than observed for the single Danish soil, for which we have data.

A range of studies has indicated that P_c is not an exact limit for distinguishing between elastic and plastic (persistent) strains (e.g. Trautner, 2003; Arvidsson & Keller, 2004; Keller et al., 2004; Mosaddeghi et al., 2007; Keller & Lamandé, 2009). On the other hand, the concept was supported in a recent study in semi-field facilities in Denmark (Lamandé et al, 2007). In that study, plastic strains were never observed at stress values lower than P_c, and strains at higher stresses correlated to the additional increase in stress. There is an urgent need to evaluate, whether P_c reflects a threshold between sustainable and unsustainable conditions.

In addition to the above, we find it important to consider the failure induced at non-isotropic stress conditions. Some of the poor correlations between field-observed strains and the stress exceeding P_c may be due to shear failure not accounted for by the pre-compression concept (Lamandé et al., 2007). Van den Akker (2004) found shear failure to occur before P_c was exceeded for some of the Dutch soils. The minor focus on this failure process is primarily due to the difficulty in measuring soil cohesion and internal friction needed to quantify, whether shear will occur.

Based on the above, it is evident that there are a number of challenges associated with quantification of soil strength parameters relevant for traffic on cultivated land. We cannot be sure that soil functions are not compromised at stresses lower than P_c measured with the existing methodologies. And there are conflicting results as to whether P_c reflects the strength at which the soil will fail. In addition, data on P_c is scarce, and measurements are laborious and expensive. Therefore, several researchers have tried to estimate P_c from more readily assessable parameters (e.g. Lebert, 1989; Lebert & Horn, 1991; Nissen 1999; Fleige et al., 2002). Schäefer-Landefeld & Brandhuber (2001) analysed pedotransfer functions developed for German soils and rejected the approach because of very low prediction accuracies. Nissen (1999) concluded that estimates of pre-compression stresses based on pedotransfer functions would be associated with standard deviations of 50-60 kPa. On top of this should be noted that these pedotransfer functions were based on estimates of P_c determined with coefficients of variance typically ~45% (-60 hPa matric potential; Nissen, 1999).

Due to severe methodological problems in measuring mechanical stress and soil deformation in an undisturbed soil profile in the field, our knowledge of soil behaviour in real loading situations is very poor. The most comprehensive data material was collected in Sweden (Trautner, 2003; Keller, 2004). Keller (2004) reviewed data from tests in Sweden and (a few) in Denmark, in total comprising more than 50 observations of subsoil vertical stress and the associated soil deformation during wheel traffic in agricultural fields (Fig. 5.12). Vertical stress and vertical plastic (permanent) deformation was measured during wheeling at three soil depths (0.3, 0.5 and 0.7 m) and at a range of differently textured agricultural fields.

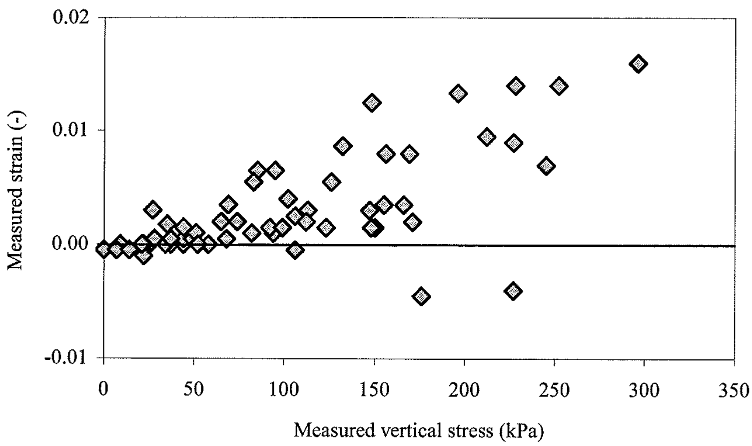

Figure 5.12. Measured strain (relative deformation of given soil layer) as related to the measured vertical stress for a number of soil compaction tests in Sweden and Denmark (Keller, 2004).

Nearly all tests took place at a water content close to field capacity. Vertical strain at 0.3–0.5 m and 0.5–0.7 m depth was calculated as the vertical displacement related to the specific soil layer divided by the height of that layer. Figure 5.12 shows that deformation was seldom observed for vertical stresses less than 50 kPa, while the deformation increased with increased vertical stress above that limit. Based on this observation, Keller (2004) noted as a rule of thumb that soil compaction can be avoided if the applied stress at any given depth is below 50 kPa. As soil strength differs from soil type to soil type – even at a reference water content of e.g. field capacity – it is unsatisfactory to use one fixed value of soil strength across a range of soils. However, the significant uncertainties in the pre-compression concept discussed above may suggest such a fixed threshold as an alternative until more knowledge has been gained regarding the pre-compression stress.

5.3.3. Comparing applied stresses with soil strength

Van den Akker (2004) estimated pre-compression stress, soil cohesion, and soil internal friction for Dutch soils by the use of pedotransfer functions suggested by Lebert & Horn (1991). He then used the SOCOMO model (a version of the Söhne model) to predict the stresses in the soil profile deriving from a 50 cm wide tyre with a tyre inflation pressure of 80 kPa. A comparison of model-predicted stresses and pedotransfer-predicted strengths of the soil for different wheel loads allowed the identification of the wheel load not giving rise to soil failure in the subsoil,- i.e. the wheel load carrying capacity. Subsoil was in this study defined as the part of the soil profile deeper than the ploughing depth, and the predictions of soil strength were for soil drained to a matric potential of -300 hPa.

The wheel load carrying capacity in the Dutch study summarized above appeared to range from 10 to 33 kN (~1 to 3.3 tonnes) across the soils of the Netherland (van den Akker, 2004). Fine-textured (clay-holding) soils had the highest carrying capacities. Soils with textures comparable to most Danish soils (generally rather sandy) had carrying capacities less than 27 kN. This is strikingly small values when considering the wheel loads commonly used in Danish agriculture today (see above). The carrying capacity would be even smaller at higher water contents because of lower pre-compression values, see the former section. Danish soils often display topsoil matric water potentials of approximately -100 hPa at field capacity (Madsen, 1976; Schjønning & Rasmussen, 2000), which is found in the spring and often again in the autumn. The matric potentials would be even higher (less negative) in the subsoil. The estimates mentioned above – using P_c values at -300 hPa matric potential – are therefore too high as wheel load carrying capacities for spring and autumn field operations.

In a former section of this report, we discussed the terms resistance and resilience. In his evaluation of wheel load carrying capacities of Dutch soils, van den Akker (2004) stated as his criterion of sustainability that no soil deformation was allowed for all the subsoil,- from the plough layer bottom and downwards. We agree that this is the optimal situation because the top part of the subsoil is important for most soil functions. Further compaction may increase the tendency that this layer becomes a 'bottleneck' (in a physical as well as a figurative sense) for root growth, and water and air transport etc. On the other hand, the 20-40 cm layer has been shown to be somewhat resilient to compaction effects. This was estimated from a long range of compaction experiments in the field (Håkansson & Reeder, 1994). Biological remediation of compacted soil in these depths was observed by Munkholm et al. (2005ab). Van den Akker's (2004) estimates of wheel load carrying capacity would certainly have arrived at higher values if soil stresses and strengths had been compared at 40 cm depth rather than the 22-32 cm ploughing depths. We will return to this crucial aspect of the compaction problem in the section on risk reduction targets.

Due to the uncertainty in the usefulness of the pre-compression stress as a criterion for soil failure, Schjønning et al. (2006b) suggested a fixed threshold of 50 kPa vertical stress in 50 cm soil depth as the criterion for sustainable traffic on soil. This was based on the previously mentioned comprehensive data set collected in Sweden (Keller, 2004; Fig. 5.12). The approach took into account the low or non-existent resilience to compaction of soil layers below 50 cm. Schjønning et al. (2006b) then modelled the vertical stress in a 'standard' soil profile by the Söhne model (concentration factor = 5,- reflecting a wet soil), using measured stress distributions in the soil-tyre interface for a range of agricultural tyres and wheel loads. Based on 20 combinations of tyre type, inflation pressure and wheel load (59 individual tests and model simulations), a relation could then be established that predicted the depth in the soil profile reached by the 50 kPa isobar of vertical stress, d_{50} (Eq. 1; Fig. 5.13):

$$d_{50} = 32.3 + 7.5 \times F_{wheel} + 7.7 \times \log_2(p_{tyre}), \quad R^2 = 0.960, \quad RMSE = 4.07, \qquad \textbf{(1)}$$

where d_{50} is in cm, F_{wheel} is wheel load in tonnes and p_{tyre} is tyre inflation pressure in bar. Non-SI units were used deliberately because the relation then is straightforward as a 'rule of thumb' for farmers and their consultants: '*The depth for maximum allowable stress (50 kPa) increases approximately 8 cm for each additional tonnes of wheel load and approximately 8 cm for each doubling of the inflation pressure*' (Schjønning et al., 2006b). From this relation it can be calculated that even the currently best low-pressure tyres (50 kPa inflation pressure) available for use in agriculture should not be loaded with more than ~3.5 tonnes in order to keep soil deeper than 50 cm free of >50 kPa vertical stress (at a water content of field capacity). We note that this estimate is close to the recommendation by Håkansson & Danfors (1981) based on crop productivity observed in field experiments. The estimated threshold is also in accordance with Horn & Fleige (2009).

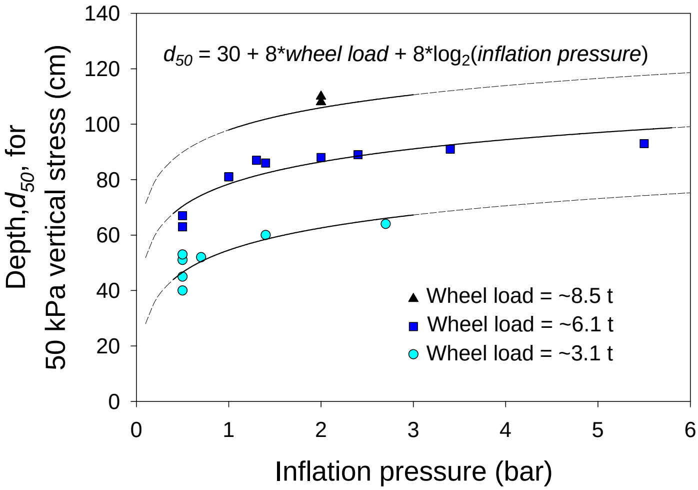

Figure 5.13. Symbols show Söhne-predicted maximum depth of 50 kPa vertical stress in relation to wheel load and tyre inflation pressure. Based on Söhne simulations on average stress distributions for each of 20 tested combinations of tyre type, wheel load and inflation pressure. The lines show the predictions of the suggested, approximate equation (the '8-8 rule'; Equation 1). Notice that the model requires wheel load and inflation pressure in units of tonnes and bars, respectively. Reproduced from Schjønning et al. (2006b).

We note from the two approaches discussed above that the wheel loads used in Danish agriculture today very often will exceed the wheel load carrying capacity. As these wheel loads are potentially applicable to all soils in Denmark, the conclusion in a SFD context is that all soils in Denmark should be classified as risk areas with respect to compaction. Exceptions may be permanent grassland never subjected to traffic or sloping areas that for other reasons cannot be accessed by heavy machinery.

5.4. Decisions on risk reduction targets

As mentioned in the previous section, it has to be established which criterion should be used for sustainability with respect to the soil compaction threat. The EU Soil Thematic Strategy includes a statement that the precautionary principle should be used in the implementation of the SFD (van Camp et al., 2004). In our view, this excludes the approach suggested by Lebert et al. (2007), accepting some deformation of any soil layer independent its depth, provided some threshold levels of simple physical measures are not exceeded. In our opinion, soil layers that are effectively non-resilient to compaction effects should not be compacted. This is because soil is a very complex biomaterial that performs a lot of biological, chemical and physical functions. The latter ones are often easier assessed but this should not be taken as an excuse for using them as criteria for biological functionality. Important soil functions that we are not aware of (as well as – of course – well-known functions) may be reduced by compaction.

Håkansson & Reeder (1994) reviewed the knowledge on the persistence of soil compaction. They concluded that compaction effects are virtually permanent below 40 cm depth even in clay soils in regions with annual freezing. They also noted that complete amelioration by mechanical loosening is usually impossible. Recent research on Danish soils confirms this statement and shows that mechanically loosening of dense soil layers may even reduce root growth and crop yields (Munkholm et al., 2005a; Olesen & Munkholm, 2007). There is also a high risk of recompaction of mechanically loosened soil (Soane et al., 1987; Munkholm et al., 2005ab). If using this as our basis, we conclude that soil layers below 40 cm should be protected from compaction of any degree.

We have noted that the EU Soil Thematic Strategy includes compaction of all soil layers – including the very topsoil – as a concern (van Camp et al., 2004). It is correct that soil may be compressed from the hooves of grazing animals. And traffic even with small-sized machinery may cause compaction of the topsoil. However, the biologically active and frequently tilled topsoil is resilient to compaction effects. Compaction of these layers may be important in terms of its effect on water infiltration and hence surface runoff and water erosion. Nevertheless, we suggest that such compaction effects are considered in the framework of the erosion threat rather than as a compaction effect *per se*.

Combining the considerations above, we arrive at a final question: what is a sustainable situation regarding the soil layer from the depth of frequent tillage (for ploughed soils often approximately 22 cm, for conservation tillage a more shallow depth) to the previously estimated 40 cm depth. The optimal situation would be to follow van den Akker (2004) in his rejection of any soil compaction of non-tilled soil layers. However, the estimates of wheel load carrying capacity previously referenced as a consequence of this approach appear to be so low that it will be very difficult/expensive to follow them in practice. Another solution – which will be promoted here – could be to define wheel load carrying capacities based on no compaction of soil layers deeper than 40 cm,- justified from the documented resilience of soil above this depth. Summarizing, the suggested risk reduction target for Danish soils regarding compaction should be that soil below 40 cm depth should never be exposed to vertical stresses higher than its strength. Until more knowledge is gained regarding the quantification of this strength as related to wheel traffic, we suggest a fixed threshold of 50 kPa vertical stress.

To reach our target, we have to identify a more differentiated criterion than a wheel load carrying capacity. It may be recalled that van den Akker (2004) performed his calculations for a specific tyre (given dimensions and tyre inflation pressure). The recent increase in knowledge on the stress distribution below agricultural tyres (Keller, 2005; Schjønning et al., 2006b, 2008) allows identification of combinations of tyre inflation pressures and wheel loads not giving rise to specific levels of vertical stress at the 40 cm depth (in analogy with Eq. (1)

above) (Schjønning et al., 2006b). Please consult the following section for a detailed discussion of this approach.

5.5. Programme of measures to reach risk reduction targets

The literature on soil compaction includes recommendations on measures to follow in order to reduce or avoid soil compaction. However, most of these recommendations are based on implicit assumptions, general knowledge and practical experience. Tijink (1998) reviewed a number of recommendations for allowable stresses in soil (Table 5.2). The recommendations are all based on a general judgement after field measurements and observations. Söhne (1953) suggested that soil at field capacity should not be loaded with tyres having > 80 kPa inflation pressure. This was supported by Vermeulen et al. (1988) except that they advised less than 40 kPa for traffic in early spring. Petelkau (1984) suggested that traffic on spring-wet soil should not take place with mean ground pressures exceeding 50-80 kPa dependent on soil type. Finally, Rusanov (1994) recommended that the stress in 50 cm depth should not exceed 50 kPa,- even at dry conditions. We note that the latter is in accordance with the Swedish observations referenced above (Keller, 2004).

Table 5.2. Guidelines to prevent soil compaction, expressed in limits for inflation pressure (p_i), average ground pressure (p_c) and vertical soil stresses at 50 cm depth (p_{50}) in spring or in summer/autumn (after Tijink, 1998).

Reference	p_i (kPa)	p_c (kPa) spring	p_c (kPa) summer autumn	p_{50} (kPa) spring	p_{50} (kPa) summer autumn	Remarks
Söhne (1953)	80					Normal moisture conditions
Perdok and Terpstra (1983)	100					
Petelkau (1984)		50	80 [a]			Sand
		80	150 [a]			Loam
		80	200 [a]			Clay
USSR (1986) [b]		80	100	25	30	w.c. (0-30) > 90 % f.c.[c]
Rusanov (1994)		100	120	25	30	w.c. (0-30) > 70-90 % f.c. [c]
		120	140	30	35	w.c. (0-30) > 60-70 % f.c. [c]
		150	180	35	45	w.c. (0-30) > 50-60 % f.c. [c]
		180	210	35	50	w.c. (0-30) < 50 % f.c. [c]
Vermeulen *et al.* (1988)	40	50				early spring, arable land
	80		100			arable land

[a] Moisture content < 70 % of field capacity

[b] Official standard for fine-grained soils, for the whole former Soviet Union. For undriven wheels values are 10 % higher. For two passes in the same rut the values are 10 % lower; for 3 and more passes values are 20 % lower.

[c] w.c. (0-30) = water content (0-30 cm depth); f.c. = field capacity.

We suggest that the measures to be used for regulation of soil traffic in agriculture should be based on the explicit and scientifically based thresholds discussed in the former sections. However, in this exercise the decision on which measures should be used to reach the risk reduction target to a large extent is a political issue. However, if the suggested risk reduction target (Section 5.4) is approved by the political authorities, there is a need for technical tools for farmers to reach the target, and for the political system to assess, whether the farmers meet the thresholds linked to the target. Several possibilities may exist. In this section, we will describe one potential approach.

A group of scientists within the Faculty of Agricultural Sciences at Aarhus University is currently cooperating with stakeholders interested in and involved with the soil compaction problem. These are The Danish Agricultural Advisory Service, The Association of Danish Farm Contractors, the tyre industry (Nordisk Dæk Import A-S), and farmers. The work is also linked to a similar project in Switzerland. The aim is to create an online decision support system, allowing the farmer or any other person to evaluate, whether a specific soil (a specific field) at a specific date (a specific water content) is able to carry a specific machinery without causing persistent compaction deeper than 40 cm depth. The tool is based on recently collected knowledge of the stresses in the soil-tyre interface (Schjønning et al., 2006b, 2008; Lamandé & Schjønning, 2008a) and of the transmission of these stresses through the soil profile (Lamandé & Schjønning, 2008b). The ongoing project includes an extension of data for soil strength as affected by soil water content.

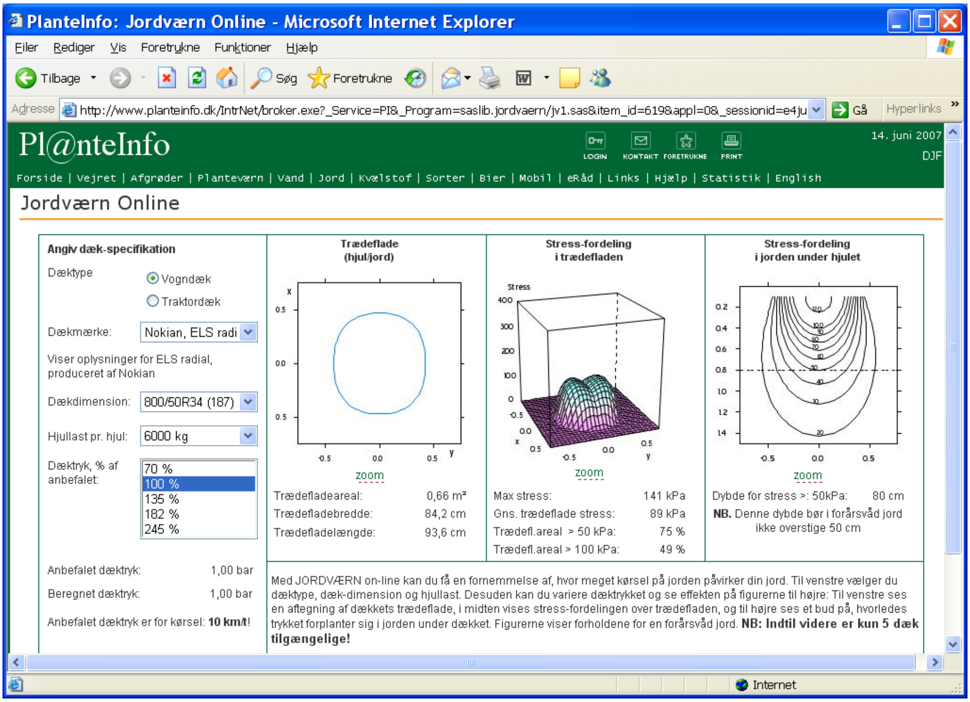

Figure 5.14. Features of the 'JORDVÆRN online' decision support system (prototype) showing diagrams of the contact area, the stress distribution in the contact area, and Söhne predictions of stresses in the soil beneath a selected tyre (www.planteinfo.dk, sheet 'Jord').

The existing knowledge is already available online in a tutorial presentation of stresses acting beneath a long range of tyres but at a "standard" soil type / water content combination (www.planteinfo.dk, sheet 'Jord', Fig. 5.14). Equations for computing the parameters in the

FRIDA model for stress distribution in the soil-tyre interface (Schjønning et al., 2008) are applied to the tyre characteristics for the ~200 different tyres in the system. The user of the system can choose a specific tyre from a list and then select a wheel load and a tyre inflation pressure. The system will then predict the contact area and its form (the left hand Figure on the screen-dump in Figure 5.14) and the stress distribution across this contact area (the middle Figure in Figure 5.14). Next, the vertical stress in the soil profile across the width of the tyre is calculated by the Söhne model (Söhne, 1958), taking the so-called concentration factor as ν=5 (reflecting a 'wet' soil).

The Söhne predictions were evaluated for two tyre types at two wheel loads for a loamy soil at field capacity (using data from Lamandé & Schjønning, 2008b; results not shown but calculations also mentioned in Schjønning et al., 2006b). Expressed in terms of the d_{50}-parameter (Eq. 1), the Söhne predictions of 50 kPa vertical stress fitted well the depth reached by the 50 kPa isobar (RMSE~5 cm; bias~-4 cm). This quantitative test of the Söhne model for stress prediction in real soil is important because data in literature is scarce. The small bias (~-4 cm) reflected a slight underestimation for all four combinations of tyre type and wheel load / inflation pressure, which means that the concentration factor used (ν=5) is slightly too low. Based on these considerations, we conclude that it is possible to predict the stresses in the soil profile below a running wheel by use of the Söhne model with realistic inputs of surface stresses. The predictions shown in the right hand Figure of the screen-dump in Figure 5.14 is of course only valid for a 'wet' soil. This is one point, where the on-going project will include the possibility of predicting also at other water contents, please see below.

Figure 5.15 illustrates the flow of information in the decision support system under construction. Inputs are labelled by light grey. In effect, the system is an implementation of the *SoilFlex* model (Keller et al., 2007) in an environment of existing databases and information services. The soil water content will be calculated from the crop and weather information, and this information already exists in the 'Dansk Markdatabase' and a network of meteorological stations, respectively. The soil type is found in the Danish Soil Database (http://www.djfgeodata.dk) from input of geographical coordinates. The only manual inputs are 1) information on recent tillage, and 2) characteristics of the machinery intended for use in the field.

The system will allow the user to take decisions on traffic from comparison of soil strength and stress at any soil depth. If taking no compaction deeper than 40 cm as the sustainability criterion (Section 5.4), the decision on traffic may be based on that information from the system. However, other criteria may be set up instead. This could be – for example – no compaction of subsoils at all (i.e., including the 20-40 cm soil layer) as suggested by van den Akker (2004).

It may seem unrealistic that farmers before any traffic event should consult a computer system as described. However, in the future it will probably be possible to have measuring systems on the machinery delivering real-time information on wheel loads and inflation pressure. This could then be combined with GPS-systems allowing for a computer on the tractor to perform all calculations automatically at any location in the field. Before such systems have been developed, another option is to create maps of all fields at selected water contents. This will allow an approximate evaluation of the ability of the soil to sustain the loads on the machinery. However, as already mentioned such maps will deviate according to the type of tyre used and thus have to be produced for every type of tyre. The tool may also be used for strategic planning, e.g. when farmers acquire new machinery.

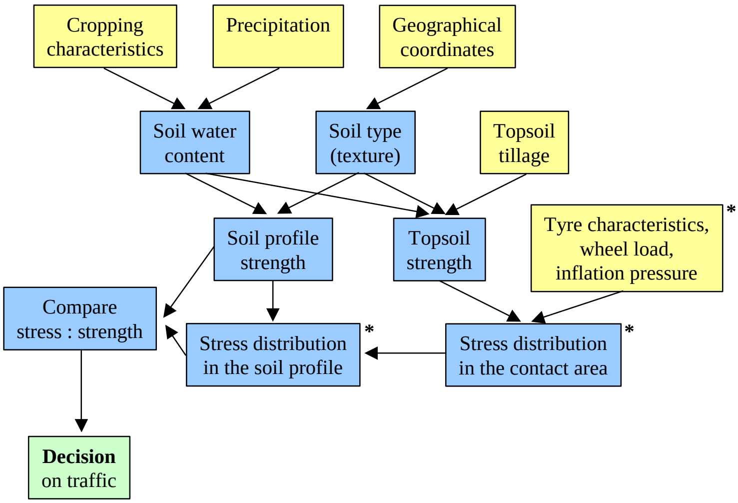

Figure 5.15. Route diagram illustrating the information flow in the planned decision support system ’JORDVÆRN online” allowing an evaluation of the sustainability of an intended traffic event on a specific field in Denmark. Please consult text for explanation of details. The boxes marked with an asterisk constitute the existing web tool (www.planteinfo.dk, sheet ‘Jord’).

5.6. Knowledge gaps and research needs

Although scientists have focused on soil compaction for decades, there are significant knowledge gaps to be filled in order to avoid detrimental soil compaction. Quantitative information on the stress distribution in the soil-tyre interface has been scarce due to major difficulties in measuring these stresses below running wheels. The recent data series collected in Sweden and Denmark (Keller, 2005; Schjønning et al., 2006b, 2008; Lamandé & Schjønning, 2008a) have given some insight mainly into the qualitative nature of the stress distribution. The measurements of stress distribution in the contact area reported by Schjønning et al. (2006b) and those carried out in the new Danish project is just a start in our search for reliable input data to stress transmission in the soil profile. One crucial observation is that the peak stresses in the soil-tyre contact area are approximately proportional to the tyre inflation pressure for a given soil condition. However, much more measurements need to be carried out to reach a satisfactory quantitative basis for modelling the stress in the interface for different soil types, soil consistencies (tillage) and soil water contents. Such information may continuously be added to the decision support system described above in order to improve the accuracy of predictions.

It is more than 50 years ago that W. Söhne in Germany modified the equation for stress transmission in elastic materials for use also in soils. The general principle of adding surface vertical stress vectors to predict the vertical stress at a given soil depth has been confirmed in basic studies in recently tilled soil (Smith & Dickson, 1990), intact lysimeter soil (Lamandé et al., 2007), and to some degree also for soil under field conditions (Lamandé & Schjønning, 2008b, 2009b). Nevertheless, the validity of the model has been questioned (e.g. Trautner, 2003), and the recent achievements suggest the need for a two-layer model (Keller & Lamandé, 2009). Lamandé & Schjønning (2008b; 2009abc) showed that stresses in the soil-tyre interface are transmitted nearly undamped through the plough layer of arable soil,- even 1½ year following the latest ploughing operation. We need much more 3D-studies of wheeling-induced stresses in the soil profile under field conditions. Future studies should include all stress components. Soil may fail due to deviatoric stresses, and we need much more information on these aspects. Again,- the studies have to take place under conditions that reflect the field situation.

In section 5.3.2, we focused the challenges in identifying a soil strength parameter of relevance to short-term mechanical stresses induced by wheels from agricultural machinery. For the time being, we have to rely on the pre-compression stress determined in uniaxial confined compression tests. The pre-compression stress should be quantified for all soil types and at a range of water contents (matric potentials). Nevertheless, we need more field investigations of stress-strain relations in real soil to reveal to what extent we can use that strength expression (Keller & Lamandé, 2009). Soil shear strength should also be focused because deviatoric stresses may account for a considerable fraction of the observed soil deformation.

Although the comprehensive international series of field experiments with compaction by high axle loads (Håkansson & Reeder, 1994) indicated permanent decrease of crop yields, more studies are needed. Individual field experiments have shown no or negligible effects from traffic with high wheel loads (e.g. Arvidsson, 2001). Future studies hopefully may identify the mechanisms responsible for the conflicting results, e.g. by use of the Least Limiting Water Range concept (Da Silva et al., 1994). Given the fact that many fields today have already been compacted, we need more knowledge about the potential alleviation e.g. by growing crops with vigorous root systems. More studies are also needed to quantify the effects of compaction on major ecological soil functions like denitrification, carbon sequestration, and preferential flow in macropores. Loss of nitrous oxides contributes to the greenhouse effect and hence the climate change. Preferential flow in macropores may dramatically increase the transport of pollutants attached to colloids to deeper soil horizons and to the aquatic environment. We need a quantitative evaluation of the compaction effect on these processes.

6. Soil organic matter decline (SOMd)

6.1. Are Danish soils low in organic matter?

Before agricultural use of synthetic fertilizers, soil organic matter (SOM) was important because of its influence on nutrient supply to crops. In the more industrialized nations – as in Denmark – enhanced SOM levels are now seen primarily as a tool for overcoming soil problems other than simple nutrient supply and as a sink for atmospheric CO_2. The growing interest in organic farming though re-focuses the SOM function related to nutrient supply also for Danish conditions. Nevertheless, in this context we will evaluate SOM contents primarily with respect to its interaction with soil mineral particles in creating stable yet friable tilth conditions for soil agricultural use.

The SOM pool encompasses plant, animal and microbial residues in all stages of decay and a diversity of heterogeneous organic substances, mainly of microbial origin and intimately associated with mineral soil constituents (Christensen, 1996). Thus the total organic matter (OM) pool in soil includes organic substances of widely different composition and decomposability.

Based on nation-wide soil profile data, Krogh et al. (2003) estimated an average OM content in 0-100 cm soil profiles corresponding to 144 t C ha^{-1}, the OM storage in Danish soils corresponding to 579 x 10^6 t C of which 69 % is associated with soils under agricultural use. Another study based on soil samples from 336 arable grid points in the nation-wide Square Grid System estimated an average OM content of 111 t C ha^{-1} in the 0-50 cm soil layer (Heidmann et al., 2001, 2002).

A substantial fraction of the OM in soil is relatively resistant to decay and shows turnover rates of decades to centuries. Other fractions turn over within months to years, while the most labile OM fractions are decomposed within days to weeks. Soils may contain charred (charcoal-like) material formed during historic burning of vegetation, and biological inactive OM deposited by geological processes such as podzolation. For Danish arable soils, however, Bruun et al. (2005) found that charred material accounted for < 5 % of the total OM pool residing in the plough-layer.

The addition of OM (e.g. crop residues, animal manure) to soil has an immediate effect on the activity of the soil decomposer populations. This initial phase of decomposition is affected partly by the chemical and physical nature of the added OM, and partly by soil water availability, soil temperature, soil porosity, and the availability of nutrients in the soil. Soil disturbances caused by tillage may stimulate microbial activity during the initial decomposition phase by improving soil air exchange, reducing the size of particulate organic residues, and by breaking up soil aggregates whereby aggregate-protected OM becomes available for decomposition. However, tillage induced effects during this decomposition phase seem to be transient (Kristensen et al., 2003; Thomsen & Sørensen, 2006ab). The initial phase of decomposition is typically completed within the first year after OM addition, and the main effect of tillage during this phase appears to be related to the degree of mixing of soil and OM, and to the seasonal progress in the decomposition of the added OM (Christensen, 1986).

The longer-term effect of added OM is associated with the subsequent turnover of microbial products generated during the initial phase of decomposition. The resulting storage of OM in soil is therefore tightly linked to the quantity of OM entering the soil.

Evaluating the potential for an increased, longer-term storage of OM in agricultural soil is linked to crop and soil management that defines the OM input and the turnover of OM already in the soil. The upper limit for OM accumulation on a given site, also termed the potential storage capacity (Ingram & Fernandes, 2001), is determined by site geology (mineralogy and texture), hydrology (e.g. drainage conditions), and by the climate regime (temperature and precipitation). These overall site factors frame the theoretical potential for OM stabilization in organo-mineral complexes and the potential activity of the microbial decomposers.

The attainable OM storage is defined by the agricultural production system (e.g. cereal dominated cropping, dairy production with grassland dominated cropping), the actual OM storage being defined by the specific management that is adopted in a given production system (e.g. crop sequence, crop residue disposal, use of nitrate catch crops, use of animal manure, soil tillage system). A change in the input of OM to the soil will affect the accumulation of new SOM, whereas a change in soil tillage system may affect its spatial distribution and subsequent turnover rates. Ideally the level of OM in soil under a given production system with fixed management practices will gradually approach a steady-state equilibrium. However, land use and management induced changes in SOM levels are slow and will be manifest and experimentally verifiable only over extended time periods (Smith, 2004), especially in temperate soils cultivated for centuries (Christensen & Johnston, 1997). Even when native and other permanently vegetated soils high in OM are brought under cultivation or when long-term arable soils are converted to permanent grassland or woodlands, long periods (> 100 years) are needed to follow changes in SOM storage. Therefore long-term experimentation combined with simulation models is indispensable to determine subtle changes in the level of OM.

In section 6.3.1., we review the knowledge on land use and management practices for the OM content of Danish soils. SOM has generally declined in Danish soils during the past decades. In the 1930s, permanent grassland accounted for approximately 500.000 ha, which is now reduced to less than 200.000 ha. During the same period, the agricultural area grown with small grain cereals increased from ~43 % to ~55 % (Kyllingsbæk, 2008). In some parts of Denmark, a major part of the agricultural land has been grown continuously with annual cash crops for decades. Today, the agricultural extension service often reports on tilth problems for clayey soils (Leif Knudsen, personal information), which may be due to critical low levels of SOM.

6.2. Why and how low organic matter content influences soil functions

SOM interacts with primary mineral particles in the creation of secondary structural units (aggregates), and this induces inter-aggregate macropores and hence a reduction in bulk density as documented by Franzluebbers et al. (2001). This may significantly influence soil aeration and drainage of surplus water from soils (e.g. Schjønning et al., 2002a). The change in SOM induced by the differentiated fertilization strategy in the long-term Askov experiments has given rise to a difference in soil porosity. Averaged for plough layer soil of all four experimental fields, the porosity was significantly lowest for the unfertilized plots (0.401 m^3m^{-3}), while no significant difference could be observed for the plots receiving fertilizer as either mineral fertilizers or as animal manure (averaged to 0.427 m^3m^{-3}) (Schjønning et al., 1994, 2005). Although the effect is rather small, it may be important in terms of a number of functions related to e.g. the exchange of water and air in soil. Munkholm et al. (2001a) showed that the friability of a soil low in SOM correlated positively to soil porosity. This indicates a direct effect of soil porosity that is important for soil fragmentation in tillage.

SOM is often claimed to enhance soil ability to retain water. However, several investigations in Danish arable soils have shown that the effect is negligible. Ten years of annual incorporation of straw in continuous growing of spring barley has been shown to significantly increase the SOM content (Schjønning, 1986). Nevertheless, another study showed no effect on soil water retained at field capacity in the spring. Neither did the increase in SOM increase the water available to plants (water retained between the 'permanent wilting point' – a matric potential of -1.5 MPa – and the -100 hPa potential) (Schjønning, 1985). Even the significant increase in SOM from a century of animal manure versus mineral fertilizer application did not induce significant differences in water available to plants (Schjønning et al., 1994; Schjønning, 1995). In accordance with other studies (e.g. Miller et al., 2002) the Danish studies indicate that an increase in SOM primarily increase the volume of water retained at -1.5 MPa.

Recent Danish studies have given an important insight in the crucial role of SOM in the creation of soil structure optimizing a range of important soil functions (Table 6.1). Soil was collected from two neighbouring fields at Sjællands Odde, ensuring that they had identical geological origin and soil texture (Munkholm et al., 2001a; Schjønning et al., 2002b; Elmholt et al., 2008). One of the fields had been grown with forage crops in an organic growing system for half a century (labelled High-C in Table 6.1). This included frequent application of animal manure in a crop rotation including grass leys. In contrast, the other soil had been used for continuous growing of small grain cereal crops with only mineral fertilizers, and with no return of organic residues to the soil for at least 25 years prior to the investigation (labelled Low-C in Table 6.1).

The contrasting management gave significant differences in total as well as labile SOM (Table 6.1). The dispersion of clay-sized colloids was higher for the Low-C soil than the High-C soil when field-moist soil was shaken in water for 2 minutes (Table 6.1). Interestingly, however, when air-dried soil aggregates were treated similarly, the amount of dispersible clay was significantly lower for the Low-C than the High-C soil. This may be interpreted as a cementation of dispersed clay in the Low-C soil upon drying (Elmholt et al., 2008). In accordance with this, the tensile strength of dry macro-aggregates was highest for that soil. In essence, the high dispersion of clay in the soil with a low content of SOM may cause the creation of unfortunate strong clods in dry conditions.

The results in Table 6.1 also reveal that even the short-term stability of field-moist aggregates in wet conditions may be higher for a low-C soil than for a high-C soil. Stability of wet aggregates in a relatively short sieving process has been used in many contexts as an indicator of ‘good’ soil structure (e.g. Loveland and Webb, 2003). However, the results in Table 6.1 indicate that a high stability for some soils may rather reflect dense clods with cemented clay as the dominant bonding material. The observation has been confirmed in other Danish studies (e.g. Schjønning et al., 2002b [Fig. 4]; Schjønning et al., 2007). Clay minerals in soil may be regarded as the basic level in the hierarchy of structural elements. If the organization of clay particles is lost, all other hierarchical orders are lost or absent (Dexter, 1988). The implications for daily management are severe. When a soil with low content of biotic bonding agents, such as the Low-C soil, gets wet, clay may readily disperse into the pore water and result in an unstable and muddy soil. When drying up, the dispersed clay will cement the aggregates to mechanically hard clods. A correlation between the amount of dispersible clay and the tensile strength of aggregates has also been observed in other studies (Kay & Dexter, 1992; Watts & Dexter, 1997).

Table 6.1. Soil characteristics measured at two neighbouring soils with contrasting cropping and fertilization management. Please consult text for details. Data from Munkholm et al. (2001a), Schjønning et al. (2002b) and Elmholt et al. (2008).

Soil characteristics	Management system	
	High-C	Low-C
Soil org. C, total (mg g^{-1} soil)	19.7[b]	14.3[a]
Soil org. C, hot-water extractable (µg g^{-1} soil)	232[b]	202[a]
Water-dispersible-colloids of wet soil (mg g^{-1} clay)	98[a]	134[b]
Water-dispersible-colloids of dry aggregates[1] (mg g^{-1} clay)	20.6[b]	18.0[a]
Tensile strength of dry aggregates[2] (kPa)	215[a]	267[b]
Wet macro-aggregate stability[3] (mg g^{-1} soil)	637[a]	873[b]

[1]Averaged across three aggregate sizes, 0.063-0.25, 0.5-1, 4-8 mm
[2]Averaged across four aggregate sizes, 1-2, 2-4, 4-8 and 8-16 mm
[3]>0.25 mm aggregates after Yoder-type wet-sieving for 2 minutes

Recent achievements focus on the role of SOM in re-arranging mineral particles into an open structure (Or et al., 2007). Young & Crawford (2004) suggested that the soil-microbe system should be regarded as self-organized, i.e., that the organization will increase, without being controlled by environmental or other external factors. The results discussed above may be interpreted in this framework. The low level of the carbohydrate-C for the Low-C soil may reflect that this parameter is in minimum for its bonding function of aggregates (Liebig’s law of the minimum). The High-C soil with diversified crop rotation and amble inputs of organic residues may be considered to exhibit a non-limiting level of bonding and binding mechanisms, while hot-water extractable carbohydrates in the carbon-depleted Low-C soil seem to stabilize the clay particles.

Munkholm et al. (2002) showed for another Danish soil that soil depleted in OM had very high tensile strength when dry but weak aggregates when wet (Fig. 6.1). This indicates that more energy is needed to till the soil when it is dry, and there is a greater risk of structural damage when it is tilled in the wet state. We note from Figure 6.1 that for both fields studied, it is especially the soil dressed with animal manure that deviates from the unfertilized treatment, while soil given mineral fertilizers behaved more alike the unfertilized soil. This may reflect differences in the quality of SOM not detectable by the mere level of total SOM. The different slopes in Figure 6.1 imply that the range in water content optimal for tillage is narrower for the UNF than fertilized soils (Munkholm et al., 2002). Another benefit of a high SOM content is that the soil may be tilled at higher water contents (the level of water content optimal for tillage is higher) (Schjønning et al., 1994; Munkholm et al., 2002).

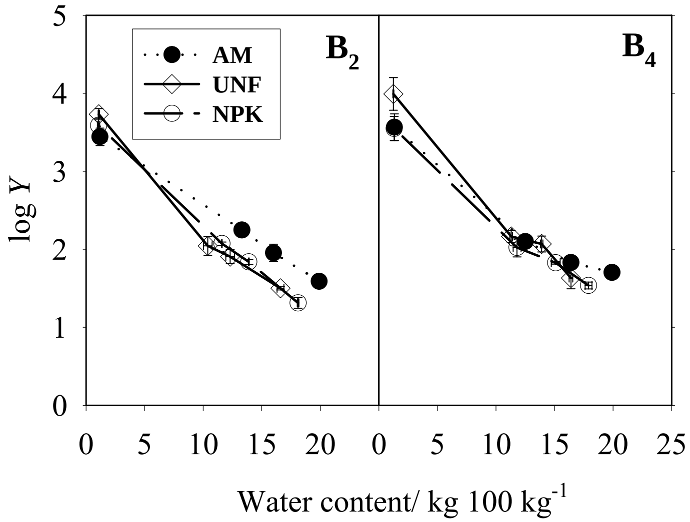

Figure 6.1. Log tensile strength (*Y*) of 8-16 mm aggregates related to gravimetric water content for long-term (~100 years) application of no fertilizers (UNF), animal manure (AM) or mineral fertilizers (NPK). Labels B2 and B4 refer to replicate fields. Reproduced from Munkholm et al. (2002).

The breakdown of clay-OM interaction discussed above may induce distinct changes in the soil pore network. Schjønning et al. (2002a) used a combination of gas diffusion and air permeability measurements to describe the pore system of intact soil cores collected in the Low-C and High-C fields discussed above. They found the Low-C soil to exhibit less tortuous macropores than the High-C soil, while the pore system in the High-C soil had a sponge-like appearance. The observation was interpreted as internal crusting and straining of soil pore walls by dispersed clay particles in the Low-C soil. This has major impact on the transport of water and gases through the soil and within the soil matrix.

Soils in arable rotations have to be frequently tilled. An important characteristic of such soils is the ability to fragment when mechanically disturbed (e.g. in preparation of a seedbed). Watts & Dexter (1997) showed that SOM plays a key role in this characteristic (Fig. 6.2). The results derive from tensile strength measurements of aggregates collected in the long-term Rothamsted experiments. It appears that soil friability is nearly directly correlated to the SOM content.

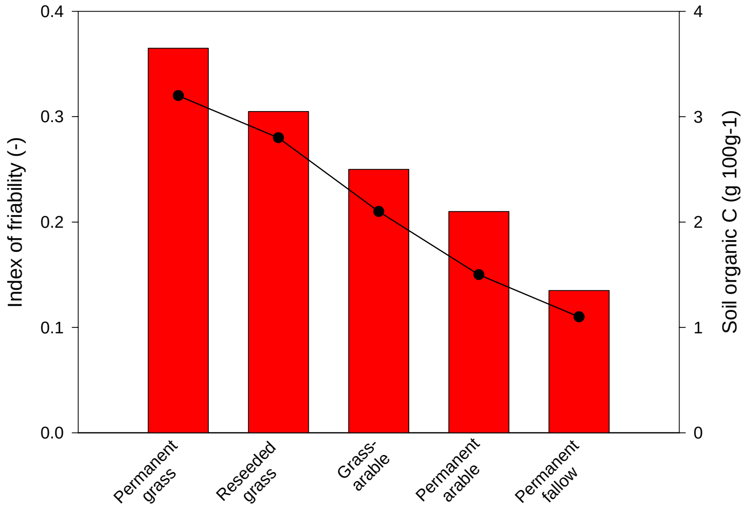

Figure 6.2. Index of friability calculated from tensile strength of dry aggregates (bars) and soil organic C (dot symbols and line). The friability index is calculated as ½×F1, where F1 is the index presented by Watts & Dexter (1997) in their study of aggregates from plots of the long-term Rothamsted rotation experiments.

Munkholm et al. (2002) studied the unfertilized (UNF), the mineral fertilized (NPK), and the animal manure dressed (AM) plots in two replicate fields of the Askov long-term fertilization experiment. The SOM content was significantly different between treatments in both fields, and averaged for the two fields the SOM content was 1.02, 1.22, and 1.45 g C $100g^{-1}$ soil for the UNF, NPK and AM treatments, respectively (Munkholm et al., 2002). The range in SOM obtained from the contrasting fertilization treatment is thus much less than found in the Rothamsted rotation comparisons (Fig. 6.2). For both Askov-fields, Munkholm et al. (2002) found the lowest index of friability for soil collected in the unfertilized plots, and for one of the fields soil friability was significantly higher for the AM treatment. Soil from the High-C field shown in Table 6.1 had a higher friability than soil collected in the Low-C field. The difference was, however, not statistically significant (Munkholm et al., 2001a). Nevertheless, the two studies of friability for Danish soils confirm the trend shown in Figure 6.2.

6.3. Identification of risk areas regarding SOMd

The working group involved with SOM decline during the work towards the EU Soil Thematic Strategy concluded that no universal target or lower threshold level of SOM existed across all soils and climates (van Camp et al., 2004). They suggested an alternative procedure searching for target SOM values for a number of well defined regional soil units that should be delineated on the basis of the important factors determining SOM levels in soil, namely a combination of: climate type (which will differentiate between geographical regions but also altitude), soil type (texture), and drainage. They acknowledged that this would probably lead to a set of hundreds of target SOM levels for Europe. They also realized that it is very difficult to derive such target values from monitoring activities, since SOM changes have to be observed on a representative sample covering the various management practices and site factors on a long term basis (new steady state for SOM after change of management practices in agriculture might last up to 50-100 years (van Camp et al., 2004)).

We agree in the problems addressed above. We further add the important aspect that no fixed optimum can be obtained even for a given soil because there may be different optima for different soil functions (Sojka & Upchurch, 1999; Letey et al., 2003; Schjønning et al., 2004). Nevertheless, in section, 6.3.2, we will attempt to identify a pattern in the thresholds for SOM found for soil functions in Danish soils. Section 6.3.1 is a review of the knowledge on management effects on SOM for Danish conditions. Section 6.3.3 is dedicated to identify the management options most likely to keep the soil's OM above a lower threshold for sustained functionality.

6.3.1. Soil use and management affecting SOM content

The subsequent sections illustrate the effect of various management options on levels of OM in typical Danish soils that have been in arable cultivation for long periods. The sections draw mainly on field experiments with factorial design that have been continued for several years. The effect of the production system on OM levels is illustrated by results from the nation-wide Danish Square Grid System (Heidmann et al., 2001, 2002).

6.3.1.1. Effects of specific management practices

Figure 6.3 illustrates the C content in soil (0-20 cm) from long-term field experiments with different annual disposal of straw for a period of 29-36 years. The experiment was placed on three sites with different soil types.

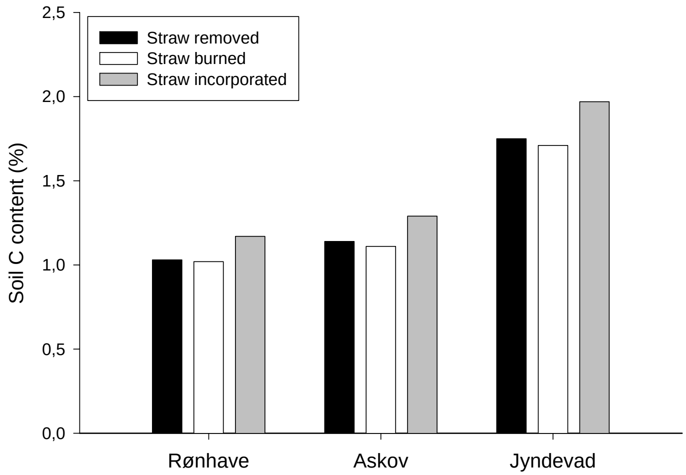

Figure 6.3. The effect of annual removal, burning and incorporation of straw on the C content in 0-20 cm soil sampled in 2002/2003 at Rønhave (JB7), Askov (JB5) and Jyndevad (JB1). The experiment grew continuous spring barley dressed with mineral fertilizer and were initiated in 1974 (Askov and Jyndevad) or in 1967 (Rønhave). From Schjønning (2004).

Spring barley supplied with mineral fertilizers was grown every year. The annual straw production (4-5 t straw ha^{-1}) was burned in the field, baled and removed, or incorporated into the soil (Schjønning, 2004). Straw burned and straw removed treatments were not significantly different in soil C content while annual incorporation of straw increased soil C by 0.17 %, corresponding to a relative increase in soil C storage of 13 %. The experiment at the Rønhave site showed a long-continued loss of SOM even when straw was incorporated (Fig. 6.4). Similar results have been obtained in farm trials under the auspices of the Danish Agricultural Advisory Service (Table 6.2). These results also indicate that continuous spring cereal cropping with mineral fertilizers causes a long-term decline in OM levels even when straw is incorporated.

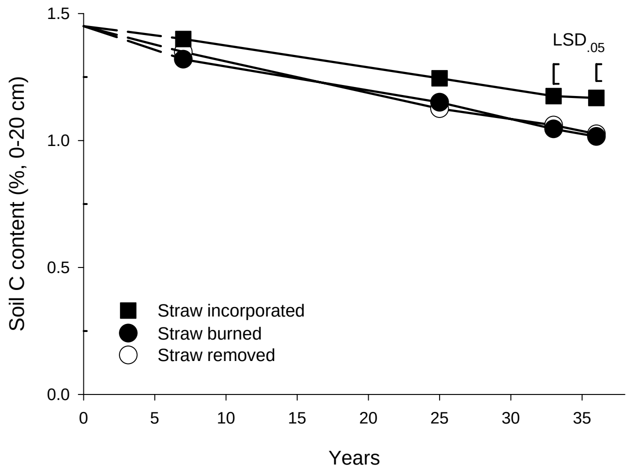

Figure 6.4. The effect of annual removal, burning and incorporation of straw on the C content in 0-20 cm soil at Rønhave (JB7). The experiment grew continuous spring barley dressed with mineral fertiliser. From Schjønning (2004).

Table 6.2. The effect of straw incorporation on soil C content in the plough layer in continuous spring barley cropping with mineral fertilizers. Nine experimental sites with straw removed or incorporated over a period of 10 years (The Danish Agricultural Advisory Service; Skriver, 1984).

	At experiment start	After harvest 1983	
	1974	Straw removed	Straw incorporated
% C (0 – 25 cm)	1.98	1.77	1.86
Relative	100	89	94

The effect of straw incorporation on SOM levels depends on the amount of straw. Figure 6.5 shows results from a field experiment with continuous spring barley and mineral fertilizers in which 0 (straw removed), 4, 8 or 12 t straw ha^{-1} was incorporated every year for a period of 18 years (Thomsen & Christensen, 2004). During the last 10 years of the experiment, the treatments were combined with a nitrate catch crop of ryegrass, split into two sub-treatments of which one was given pig slurry (35 t slurry ha^{-1} annually) while the other was left without slurry. Straw combined with catch crop growing gave more SOM than when straw alone was incorporated while the addition of pig slurry contributed little to OM accumulation. Compared to straw removal, annual incorporation of 4, 8 and 12 t straw ha^{-1} over a period of 18 years caused a relative increase in the SOM level of 12, 21 and 30 %, respectively.

Figure 6.5. Soil C content (0–20 cm) after annual incorporation of 0 (straw removed), 4, 8 and 12 t straw/ha over a period of 18 years. The experiment was carried out at Askov (JB5) with spring barley and mineral fertiliser. Treatments were straw only, straw combined with catch crop (ryegrass), and straw combined with catch crop and pig slurry (35 t/ha/year included for the last 10 years; Thomsen & Christensen (2004)).

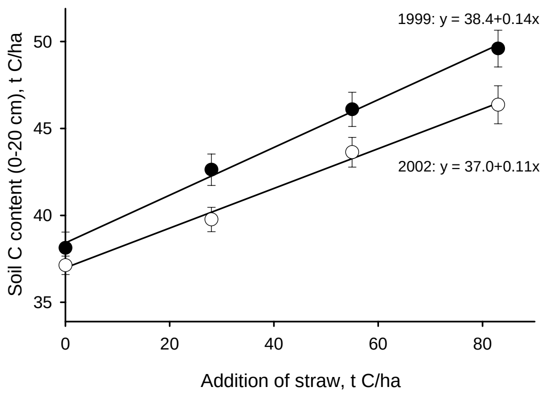

Figure 6.6. Soil C content (0-20 cm) one year (1999) and four years (2002) after the last addition of straw as a function of the amount of C added in straw in the experimental period 1989-1998. See Figure 6.5 for additional explanation (Thomsen & Christensen, 2004).

Figure 6.6 shows the relationship between the amounts of C added with straw during the 18 years period (1981-1998) and the soil C level 1 year (1999) and 4 years (2002) after the last addition of straw. The amount of C retained in soil after 1 and 4 years corresponds to 14 % and 11 % of the straw-C added during 1981-1998, respectively, and does not depend on the annual straw rate. Thus the soil's capacity to store OM was not a limiting factor in this experiment even though large quantities of straw were added over a relatively long time period.

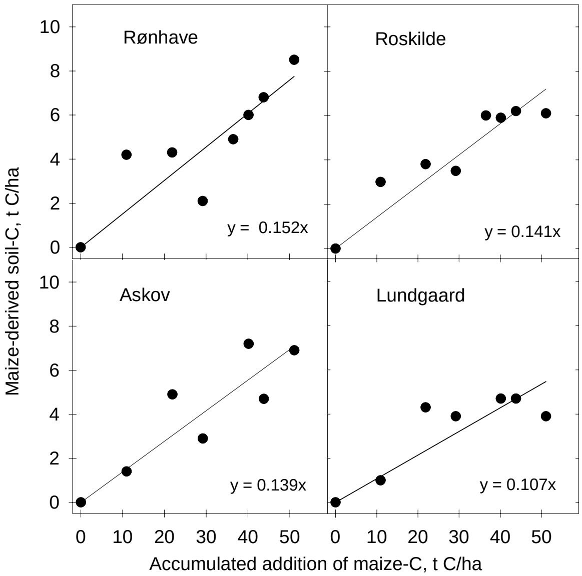

Figure 6.7. Changes in the soil C pool (0-20 cm) of maize-derived C as a function of added above-ground maize biomass. The experiment included four soil types, with continuous silage maize for 14 years. The annual addition of maize biomass corresponded to 8 t DM/ha (Kristiansen et al., 2005).

The retention of C added in maize biomass was studied in a confined small-plot experiment where silage maize was cropped continuously on four soil types over a period of 14 years (Kristiansen et al., 2005). The maize received mineral fertilizers and included two treatments. In one treatment maize C came only from roots and stubbles while in the other treatment roots and stubbles were supplemented with chopped aboveground maize biomass equivalent to 8 t DM ha^{-1}. Changes in the natural abundance of ^{13}C showed that the content of maize derived OM in the soils was directly related to the amount of C added in maize biomass (Fig. 6.7). On the coarse sand site (Lundgaard), 11 % of the added maize-C was retained in the soil while the retention in the more clayey sandy loams (Askov, Roskilde and Rønhave) averaged 14 % of the added maize-C. The annual increase in maize derived OM in the Rønhave and Lundgaard soils corresponds to 0.9 and 0.7 t C ha^{-1}, respectively, and aligns with changes in the overall OM contents (Fig. 6.8). For the Askov and Roskilde soils, changes in their overall OM content were smaller than the increase in maize derived OM, and an additional annual input of 8 t maize DM ha^{-1} did not significantly affect their overall OM storage. At least for the Askov soil with a relatively high initial soil C level, the treatments applied in this study were not able to provide a further increase OM storage, suggesting that the soil's storage capacity could be near saturation.

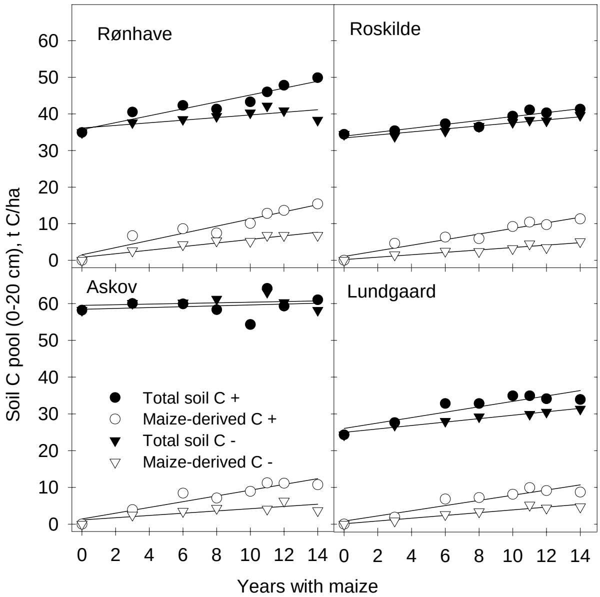

Figure 6.8. Changes in soil C and maize-derived soil C contents on four soil types (0-20 cm) with continuous silage maize for 14 years. Above-ground maize biomass was removed at harvest (-) or added (8 t DM/ha/year) immediately following harvest (+) (Kristiansen et al., 2005).

The effect of different organic amendments on the formation of OM in soil was measured over a 30-year period in a confined small-plot experiment. The soil was a coarse sand subsoil (CS-subsoil) with a very low initial C content, dug up from 50-100 cm depth and used as topsoil (Christensen & Johnston, 1997). Every year, farmyard manure, cereal straw, sphagnum peat and sawdust of known C content and corresponding to 6.5 t DM ha^{-1} was added to the soils. Soil without amendment served as reference treatment. The soil was cropped with a four-course crop rotation (spring barley, fibre flax, winter cereals, and silage maize) and dressed with mineral fertilizers. The C content in the reference soil increased from 0.27 to 0.57 % due to inputs of crop roots and stubbles (Fig. 6.9). Similar additions of DM in straw, sawdust and farmyard manure caused almost the same increase in the soils' OM level, whereas peat gave rise to a considerable higher increase. The fractional retention of C added in cereal straw, sawdust, farmyard manure and peat was 20, 23, 36 and 49 %, respectively (Fig. 6.10).

In a Swedish experiment with addition of cereal straw, green manure, farmyard manure and sawdust every second year over a period of 35 years, the retention of added C was 16, 15, 29 and 24 % (Witter, 1996). The retention of C added in peat and sewage sludge was considerably higher (53 and 46 %, respectively). The soil used in the Swedish experiment was much more clayey (37 % clay) and had an initial C content of 1.5 % (corresponding to 40 t C ha^{-1} in 0-30 cm soil depth).

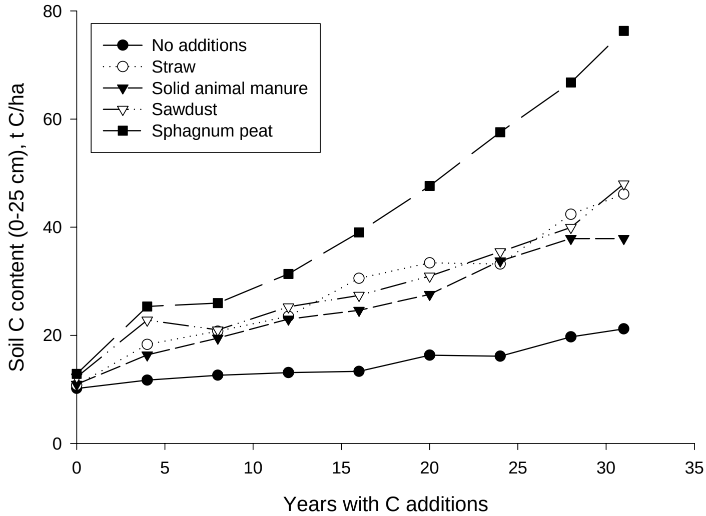

Figure 6.9. Change in the C content of a sandy soil (0-25 cm) in a 30-year experiment with annual incorporations of 6.5 t DM/ha in straw, solid animal manure, sawdust and sphagnum peat. In the reference treatment "No additions" only roots and stubble were incorporated (Christensen & Johnston, 1997).

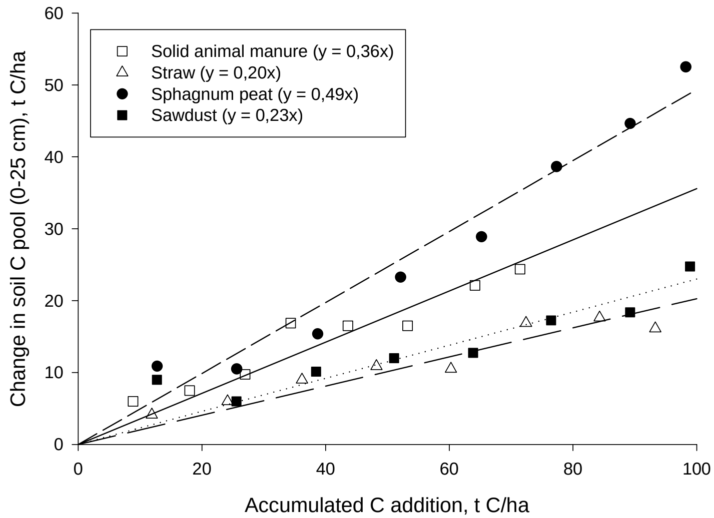

Figure 6.10. Change in soil C pool (0-25 cm) as a function of C added in straw, sawdust, solid animal manure and sphagnum peat. Annual addition was 6.5 t DM/ha over a period of 30 years (Christensen & Johnston, 1997).

These results show that the build up of SOM is similar for primary plant materials (straw and sawdust) despite their very different concentration of lignin. This is in accordance with Hofmann et al. (2009) who studied the turnover of lignin added in maize biomass and native lignin already in the soil. It was found that the native as well as the maize-derived OM pool turned over more slowly than their corresponding lignin fractions. Thus lignin cannot *per se* be considered a plant residue component with particular resistance to decomposition in soil. For animal manure, sewage sludge and peat that have been exposed to microbial processing before being added to the soil, a larger fraction of the added C was retained in the soil. Such organic amendments will therefore contribute more to the build up of OM in soil than primary plant residues.

Table 6.3. Annual change in soil C content (0-25 cm) in a 30-year experiment with different crop rotations and additions of straw and animal manure on a C-rich sandy loam soil (SL-topsoil), and C-poor coarse sand (CS-subsoil) and sandy loam (SL-subsoil) soils. Initial soil C contents are given in brackets (Christensen, 1988).

Management practice	Annual change in soil C content (kg C/ha)		
	SL-topsoil (3% C)	CS-subsoil (0.1% C)	SL-subsoil (0.2% C)
Vegetation-free fallow, unfertilised	-1249	+90	
Cereals, straw removed, MF[1)]	-851	+371	
Cereals, straw incorporated, MF	-319	+765	
Crop rotation[2)], MF	-574	+514	+578
Crop rotation with SMA[3)]	-308	+724	+960
Root crops, MF	-926	+240	
3-yr grass-clover + 1-yr turnips, MF	-458	+596	

[1)]MF = mineral fertiliser
[2)]winter crop, turnips, spring-sown cereals, grass-clover
[3)]SMA= solid manure

Table 6.3 shows results from another 30-year small plot experiment based on the CS-subsoil and a clayey subsoil (SL-subsoil) again with a very low initial OM content. Also included was a SL-topsoil taken from the 10-30 cm layer of a field that had been in permanent grassland for a very long period and thus had a high initial OM level. The treatments included different cropping systems and tested effects of crop residue disposal and use of animal manure (Christensen, 1988). All treatments introduced an increase in the OM content of the CS-subsoil and a decrease in the SL-topsoil. When this soil was left in vegetation-free fallow for 30 years, 34 % of the SOM pool was lost. This loss corresponds to an average annual decline of 1.25 t C ha^{-1} (Christensen, 1988). The systems with incorporation of crop residues and addition of animal manure showed an annual loss of 0.31 t C ha^{-1} from the OM rich SL-topsoil. The same cropping systems showed average annual accumulations of 0.72 to 0.96 t C ha^{-1} in the OM poor CS- and SL-subsoils. Compared with cropping systems based on cereals and removal of crop residues, the four-course crop rotation had a positive effect on the OM level in the soil. A more frequent presence of grass-clover leys in the rotation increased the OM storage. In a 6-year field experiment with four-species grass-only leys, dressed with mineral fertilizers and used for cutting, Christensen et al. (2009) observed an average annual accumulation of 1.1 t C ha^{-1}. Table 6.3 also shows that the system with root crops contributed less to the OM pool than the cereal based system with removal of crop residues, whereas addition of animal manure had a clear positive effect on the build up of OM in the soil.

The experiment described above illustrates that the initial SOM level has a decisive influence on the potential for further accumulation of OM. Clearly there appears to be an upper limit for OM storage in soils subject to a given land use. However, the time span for management-induced changes in SOM is very long. Even after 30 years, the OM levels in soils exposed to different management had not yet reached steady-state equilibrium (Christensen, 1988).

The effect of crop rotation on the development in OM levels is linked to characteristics of the root system, the loss of aboveground plant biomass during the growth period, the length of the active growth period, and the frequency and intensity of tillage associated with the crops grown in the rotation. Grass with a long active growth period and a high root density provides more root-derived OM than cereals and root crops. Crop rotations with a high proportion of cash crops (e.g. cereals, oilseed rape, and peas) and row crops (beet roots, potatoes and silage maize) involve more frequent tillage than cropping systems dominated by perennial grass leys.

Table 6.4. Change in soil C content (0-20 cm) in a 30-year field experiment (1956-1986) at Askov Experimental Station with vegetation-free fallow and in three crop rotations with added mineral fertiliser (Christensen, 1990).

Crop rotation[1)]	% C		Annual change[2)], kg C/ha	Relative decrease over 30 years, %
	At start	After 30 years		
Vegetation-free fallow	1.66	1.11	-589 c	34
Wi-Tu-Sp-Cl	1.56	1.35	-269 a	16
Wi-Tu-Sp-Fl	1.58	1.30	-320 ab	19
Wi-Ma-Sp-Fl	1.65	1.32	-362 b	21

[1)] Wi = winter wheat, Tu = turnip, Sp = spring cereals, Cl = grass-clover, Fl = flax, Ma = maize
[2)] Values followed by the same letter are not significantly different at the 95% level

Table 6.4 shows the change in SOM over a 30 years period in a field experiment at Askov Experimental Station (Christensen, 1990). The initial soil C content was moderate (1.7 % C) and the experiment compared three crop rotations with a tilled, vegetation-free fallow soil that was kept free of vegetation during the experimental period by tillage operations. The fallow soil lost 34 % of its initial soil C content (0-20 cm). The difference between beets root and silage maize as a row crop in the rotation was not statistically significant, whereas the rotation with one-year grass-clover maintained a slightly higher C content. However, all treatments caused a decline in the SOM pool, and even after 30 years the OM pool had not yet reached steady-state equilibrium between build-up and loss of soil C, underpinning the very long term nature of management induced changes in SOM contents.

The use of nitrate catch crops and green manure crops in breaks between main crops when the soil would otherwise have been left without vegetation, contributes to an increased above-and below-ground input of OM to the soil. Figure 6.5 shows the effect of ryegrass used as a nitrate catch crop in continuous spring barley. The effect of the grass corresponds to annual incorporation of 4 t straw ha^{-1} (Thomsen & Christensen, 2004). Similar results have been found in other studies (Tables 6.5 and 6.6).

Table 6.5. Soil C content at 0-10 cm and 10-20 cm after more than 10 years of continuous cereal cropping (mainly spring barley) with and without ryegrass as a catch crop (Rasmussen, 1991).

	% soil C			
	Jyndevad		Højer	
	0-10 cm	10-20 cm	0-10 cm	10-20 cm
Without catch crop, ploughed	1.86	1.85	1.58	1.55
With catch crop[1)], ploughed	2.07	1.92	1.73	1.69
With catch crop, no tillage	2.15	1.88	2.03	1.75

[1)]Spring-sown ryegrass, fertilised after cereal harvest, one cut at beginning of November

Table 6.6. Effect of catch crop on soil C content at 0-20 cm after 23 years of continuous spring barley cropping on a coarse sandy soil (Hansen et al., 2000).

	% soil C[1)]
Autumn-ploughed, no catch crop	1.52 c
Autumn-ploughed, ryegrass catch crop[2]	1.67 b
Spring-ploughed, ryegrass catch crop[2]	1.88 a
Spring-ploughed, no catch crop	1.70 b
$LSD_{.95}$	0.17

[1)]Values followed by the same letter are not significantly different.
[2)]Undersown in spring barley in the spring.

Table 6.7. The effect of tillage (MP=mouldboard ploughing, ST=shallow tillage) on % C in soil from eleven field experiments, listed in order of clay content. Top and bottom under ST refer to the upper and lower part of the former plough layer. Numbers followed by the same letter are not significantly different (P=0.05) (Schjønning & Thomsen, 2006).

Location	Trial age (year)	ST tillage depth (cm)	MP Whole soil layer	ST Top	ST Bottom
Jyndevad	36	3-5	2.20[a]	3.05[a]	2.17[a]
Lund	3	3-5	1.98[a]	2.15[b]	1.88[a]
Dronninglund	4	3-5	5.84[a]	6.58[a]	5.98[a]
Bygholm LS	4	3-5	1.72[a]	2.05[b]	1.73[a]
Bramstrup	8	10-15	1.00[a]	1.34[b]	1.09[a]
Bygholm FC	4	3-5	1.65[a]	1.89[b]	1.63[a]
Jerslev	5	5-8	3.82[a]	3.94[a]	3.55[a]
Vasebæk	3	10	1.32[ab]	1.54[b]	1.30[a]
Malmø	30	10-15	1.78[a]	2.05[a]	1.92[a]
Nakskov	5	10	1.52[a]	1.76[b]	1.60[a]
Kløvested	2	10	2.41[a]	2.40[a]	2.29[a]
Average	-	-	2.29[a]	2.64	2.28[a]

These studies also indicate that soil tillage can have quantitative effects on the vertical distribution as well as the quantity of the OM. Table 6.7 shows the effect of introducing plough-less, shallow tillage on the soil C content in the former plough layer. The study was based on 11 field trials that compared mouldboard ploughing (MP) with non-inversion shallow tillage (ST). Two experiments had been continued for 30-36 years, the other nine for 2 to 8 years (Schjønning & Thomsen, 2006). The trials were located on soils of different geological origin, texture and OM content. Under shallow tillage, the topsoil (the tilled zone) increased significantly in soil C at six of the sites, but the change in soil C content could not be related to age of experiment, texture or depth of the tilled zone under ST treatment. In contrast to topsoil, the soil from below the tilled zone (the previously ploughed bottom layer) under ST treatment and corresponding MP treated soils did not differ in soil C content at any of the 11 sites (Table 6.7).

Table 6.8. Effect of cattle slurry (7% DM) and solid cattle manure (23% DM) on soil C contents (% C at 0-25 cm) after annual additions of, on average, 25, 50 and 100 t manure/ha over a period of 12 years. The crop rotation was turnips (maize on JB1), barley, ryegrass and barley.

	Lundgård (JB1)		Askov (JB5)	
Fresh weight/ha/year	Slurry	Solid manure	Slurry	Solid manure
Mineral fertiliser[1)]	1.45		2.21	
25 t animal manure	1.53	1.55	2.26	2.37
50 t animal manure	1.59	1.69	2.30	2.53
100 t animal manure	1.60	1.98	2.39	2.74

[1)]Control without animal manure

The effect of animal manure on SOM in field experiments on Askov (loamy sand) and Lundgaard (coarse sand) soils is illustrated in Table 6.8. Cattle slurry (with 7 % DM) and solid cattle manure (with 23 % DM) was added annually at rates of 25, 50 or 100 t fresh weight ha^{-1} over a period of 12 years. Compared to the reference treatment given mineral fertilizers only, the addition of solid manure showed the largest effect when compared to slurry on fresh weight basis. When compared on DM basis, slurry and solid manure had a similar effect on the SOM content. The initial C content in the Lundgaard soil was 1.5 % C, and all treatments (except for the mineral fertilizer reference treatment) increased the content of soil C. At Askov the initial soil C content was 2.7 %, and this soil showed a decrease for all treatments except the treatment with 100 t fresh weight ha^{-1} in solid manure. In accordance with results shown in Figure 6.5, annual addition of slurry has but a small effect on SOM content.

Soils receive significant belowground inputs of OM from the root system. The OM input arises from root exudates and root turnover during the growth period, and from the fraction of the root system that decays after crop harvest. Thus the quantity of root-derived OM input to the soil depends on crop type and management (e.g. grass cutting strategy), the length of the growth period, and to some extent of the productivity of the crop. However, there is no simple proportionality between a given crop's above- and belowground production (Jensen & Christensen, 2004). Table 6.9 shows the amount of root derived C in spring and winter barley grown to maturity. The results are based on $^{14}CO_2$ pulse-labelling of the crops under field conditions (Jensen, 1993, 1994; Jensen & Christensen, 1993). Throughout the growth period of spring barley, about 1.7 t C ha^{-1} became allocated to the root and soil. Five days after a labelling event, 23 % was released again as CO_2 by root and microbial respiration, while another 18 and 59 %, respectively, was recovered in macro-roots (> 0.425 mm) and soil. For winter barley, the total belowground transfer of C during the growth period was 2.4 t C ha^{-1} of which 40 % was released again as CO_2 within the first five days after labelling.

Table 6.9. Above-ground and below-ground biomass production in spring barley and winter barley. Results are based on $^{14}CO_2$-labelling under field conditions (Jensen & Christensen, 1993).

	1990, Spring barley			1991, Winter barley		
	kg C/ha	%		kg C/ha	%	
Above-ground production	4703		100	7433		100
Below-ground production	1652	100	35	2372	100	32
Respiration (day 0-5)	394	23	8	936	40	13
Macro-roots (root wash)	303	18	6	472	20	6
Soil	955	59	20	964	40	13

For both cereal crops, the root derived C corresponded to about 1/3 of the C present in aboveground biomass at harvest. Not considering the root derived C that was converted to CO_2 within the first five days after deposition, the root derived C input to the soil accounted for about 1.4 t C ha^{-1}. The amount of root derived OM inputs to the soil may be less in row crops such as beets root and silage maize whereas perennial leys most probably leave more root-C in the soil.

6.3.1.2. Effects of production systems

The effect of various productions systems on the OM content in Danish agricultural soils has not been studied with the same intensity as effects of management options. On a national scale, it is often assumed that the total pool of OM in agricultural soils is at equilibrium. A possible loss of OM from soils under continuous cereal cropping and other cash crops dressed with mineral fertilizers is generally considered to be compensated for by gains in OM in soils under forage production (frequent grass-clover leys) and intensive use of animal manure. It is well known that the N budget (N inputs in fertilizers and manure minus N output in harvested crops) is positive under grass based ruminant production systems, and that cereal cropping based on mineral fertilizers and straw removal shows a negative N budget. These differences in N accumulation in soil suggest corresponding changes in soil C pools.

Heidmann et al. (2001, 2002) studied the changes in the amount of soil C under different agricultural production systems. The study was based on soils from 336 grid points in the nation-wide Square Grid System, sampled in 1986/87 and again in 1997/98. Soil was sampled in the 0-25 and the 25-50 cm soil depths. The content of C in the 0-25 cm and the 25-50 cm ranged from 1.5 to 2.3 % and from 0.9 to 1.6 %, respectively. Figure 6.11 show the changes in soil C content over the 10-12 years period for the two soil depths in relation to the Danish soil type classification (JB1 to JB7). For the sand soils the soil C content increased significantly for JB 1 in the 0-25 cm and 25-50 cm depth, and for JB 2 to JB4 in the 25-50 cm depth. In contrast, the C content decreased over the period in the more clayey soils (JB5 to JB7). For JB7 the C content decreased significantly in both soil depth, and for JB6 in the 0-25 cm depth. Table 6.10 shows the total SOM pool in the 0-50 cm across the various soil types. The amount of C in the soil in 1998/99 varies from 92 to 145 t C ha^{-1}, with an average of 112 t C ha^{-1}. The changes from 1987/88 to 1998/99 show increases on the sandy soils (from 5 to 24 t C ha^{-1}) while the C content decrease in the more clayey soil (from 7 to 15 t C ha^{-1}). The average annual changes in soil C is shown in Figure 6.12. The changes observed for JB1 and JB7 are statistically significant. When combining results from all grid points, no significant change was observed in the C in 0-50 cm depth over the 10-12 years period.

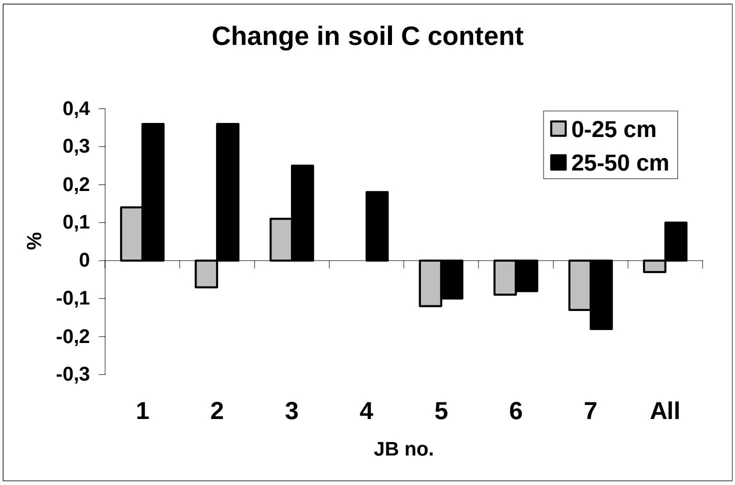

Figure 6.11. Changes in C concentration (% C) in soil sampled in 1986/87 and in 1997/98 in the Danish Square Grid System, divided into soil types. Sampling from 0-25 cm and 25-50 cm depths (Heidmann et al., 2001).

Table 6.10. Average C contents in samples from the Danish Square Grid System (t C/ha at 0-50 cm), divided into soil types (Heidmann et al., 2001).

Soil type, JB no.	1	2	3	4	5	6	7	Avg.
No. measuring points	49	28	20	80	13	93	45	328
C-content 1987/88	121	93	124	117	105	100	116	110
C-content 1998/99	145	105	130	122	92	93	101	112
Change in period	+24	+12	+6	+5	-13	-7	-15	+2

Based on information from the landowners regarding the production system that had dominated the 10-12 years period, the data from the grid points were divided according to fertilization type (mineral fertilizer, cattle manure, pig manure, mixed animal manure, and rest). The fertilization type was "mineral" when animal manure had not been used during the period. When animal manure was applied, the fertilization type was termed "cattle" or "pig" if more than 90 % of the manure was from cattle or from pigs, respectively. The category "mixed" includes mixed animal manure (cattle and pig). The grid points "rest" include mainly soils with occasional inputs of manure from fur and poultry productions. Figure 6.13 shows changes in the soil C pool (0-50 cm) in grid points over the 10-12 year period when these are arranged according to fertilization types which was taken as an approximation to production systems. Although the changes were not statistically significant, there was a clear trend with increased soil C storage under cattle manure, mixed animal manure and rest, and a decline in soil C under mineral fertilizers and pig manure.

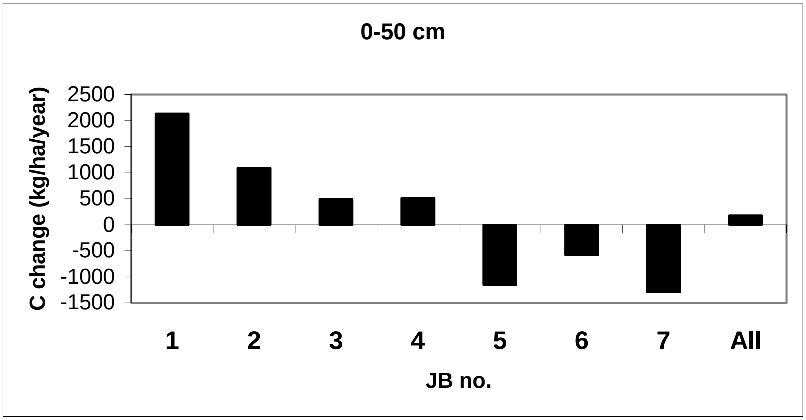

Figure 6.12. Annual average changes in soil C content (kg C/ha/year) at 0-50 cm depth in soil samples from the Danish Square Grid System divided into soil types (Heidmann et al., 2002).

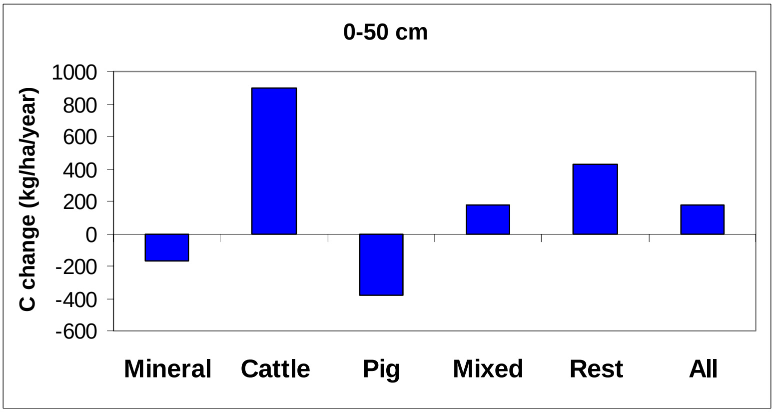

Figure 6.13. Annual average changes in soil C content (kg C/ha/year) at 0-50 cm depth in soil samples from the Danish Square Grid System divided into fertilization type (Heidmann et al., 2001).

A regression analysis of the change in soil C content shows a decline in C content with increasing initial values (in 1986/87). The number of years under grassland and the number of applications of animal manure in the 10-12-year period had a positive effect on soil C contents, as did the sum of N applied in mineral fertiliser. The various soil management factors incorporated in the regression analysis had less effect at 25-50 cm depth than at 0-25 cm (plough layer). There was a more frequent use of the mineral fertiliser on clayey soils and a prevalence of cattle farms on sandy soils. The effects of soil type and management practice are therefore likely to be confounded, whereby the isolation of effects derived from fertilization type and effects associated with soil type becomes complicated. Neither has it been possible to carry out a quality control of the management information supplied from individual farms. The division of the data material into soil types is, however, considered to be reasonably certain.

From analyses of samples from the Danish Square Grid System it appears that only small changes in the national soil C pool have taken place over the 10-12-year period investigated. This general result does, however, hide the considerable and contrasting changes that have taken place on different soil types subject to different production systems.

6.3.2. Thresholds of SOM contents for sustained soil functions?

Which SOC levels are critical to arable farming? Loveland & Webb (2003) reviewed the British literature on SOC and soil quality. They found little evidence for any general threshold, but cited findings in the USA and England that aggregate stability declines seriously at levels of SOC below 2%. Riley & Bakkegard (2006) discussed the problems in identifying a relevant criterion for sustainability. They mentioned that a threshold based on yield reductions would point out a higher SOC threshold for their Norwegian soils. Schjønning et al. (2007) focused on a range of tilth parameters and observed satisfactory tilth conditions for a soil with ~2% SOC and tilth problems for a soil with ~1% SOC. Other Danish studies have shown satisfactory tilth characteristics in soils with low SOC, e.g. ~1.2% (Munkholm et al., 2002) and on the other hand poor tilth conditions in a soil with ~1.4% (Table 6.1; Munkholm et al., 2001a; Schjønning et al., 2002a; Elmholt et al., 2008). Therefore, we agree with Loveland and Webb (2003) in their rejection of a common critical level of SOC. Other factors than the level of SOC seem to influence critical tilth conditions.

In Figure 6.14 we have plotted the data on SOM and soil friability from the Rothamsted experiments (also presented in Figure 6.2) together with recent Danish studies of the same variables. In accordance with Figure 6.2, there is a fine linear correlation between the index of friability and SOM content (the circle symbols and the dash-dot regression line). We also note that the data from the long-term Askov experiment appear to confirm a linear correlation between SOM and index of friability for that soil type. However, the two soil types display this correlation at distinctly different levels of SOM. This is a quantitative expression of the general recognition that no universal level of SOM for sustainable soil functions exist across soil types.

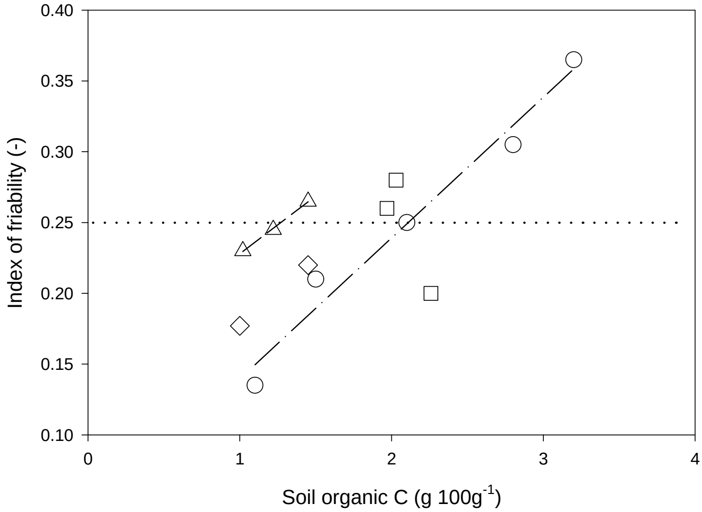

Figure 6.14. Index of friability calculated from tensile strength of dry aggregates as related to the soil organic C for a range of soils. Circles: soil from the Rothamsted experiment (Fig. 6.2), triangles: soil from the Askov experiment, squares: soil from Danish forage cropping systems, diamonds: soil from Danish cash crop systems. The index of friability for the Danish locations was calculated as the F3 index (Dexter, 2004b), while that for the English location was calculated as ½×F1. Please consult Dexter (2004b) for an explanation that this makes the indices comparable. The dash-dot line indicate linear regression of the Rothamsted data, while the dash line indicate regression for the Askov data (Watts & Dexter, 1997; Munkholm et al., 2001a, 2002; Schjønning, unpublished data).

Dexter (2004ab) used the so-called S-theory to show that a friability index F1~0.5 might serve as a lower boundary of satisfactory tilth conditions. In Figure 6.14, we combined the so-called F3 and F1 indices of friability, which means that the sustainability criterion just summarized corresponds to an index of 0.25 in the Figure (the horizontal dotted line). Although Dexter's S-theory and its suitability for expressing soil functions is still disputed, it is interesting that the threshold index of 0.25 corresponds to the friability measured for the mineral fertilized soil at the Askov experiment (the middle value) and the grass-arable rotation at the Rothamsted experiment (also here the middle value of the investigated plots). The soil dressed with animal manure at Askov and the reseeded and permanent grass treatments at Rothamsted all appear above this threshold in spite of quite different levels of SOM for the two soil types. We also note that the two Danish cash-cropped soils appear well below the threshold in close agreement with the tilth condition at these locations. We finally note that two out of three forage cropping systems display friability indices above the threshold. The soil at all these sites had satisfactory tilth conditions and the low level for the latter relates to living roots giving high tensile strength of the largest aggregates used in the tensile strength tests (Munkholm et al., 2001a). This is thus rather focusing a caveat in test conditions. Generally, the combined data in Figure 6.14 indicate that friability as quantified in these studies (Watts

& Dexter, 1997; Munkholm et al., 2001a, 2002) is a good reflection of soil tilth but that there is no unique relation to SOM across soil types.

The examples above – generally disqualifying SOM as a soil quality indicator for sustained soil physical and mechanical functions – points to the ambition to identify the hundreds of combinations of climates and soil types, each combination giving rise to a target value (van Camp et al., 2004). Thus there is room for research efforts that can isolate fractions of SOM that are more relevant to physical and mechanical soil properties. Thomsen et al. (2009) found that near infrared spectroscopy (NIRS) allowed prediction of labile SOM across 37 soils including soils from long-term experiments, gradients of texture/SOM in individual fields and soils from sites with very contrasting geological origin and management histories. However, in addition to being a tedious task, within field variability may represent a challenge even to such distributed approach. Moreover, the evaluation of the sustainability of a given management system based on SOM levels requires monitoring over longer time spans (van Camp et al., 2004).

Another potential approach would be to monitor and evaluate SOM independent properties related to the specific physical and mechanical soil functions. One example would be the friability index discussed above. However, soil friability will be dependent on other soil properties than SOM (e.g. bulk density differences derived from soil compaction). Direct evaluation of soil functions would therefore integrate several of the threats addressed in the Soil Thematic Strategy.

Based on concepts by Hassink (1997), Dexter et al. (2008) recently showed that arable soils often display a ratio, n, between <2 μm mineral particles (clay) and organic carbon (OC) close to or higher than 10 (n = clay/OC = 10). Their results further indicated that for such soils, some selected soil physical properties correlated to the content of OC. For mechanically undisturbed soils like pasture, clay/OC ratios were typically below 10. Such soils have often been considered as having passed their 'capacity factor' for carbon sequestration (Ingram & Fernandes, 2001; Carter et al., 2003). For Danish agricultural soils with soil clay/OC ratios above this limit, the current level of OC appears to depend mainly on the amount of OM added in crop residues, animal manures and other organic amendments (Thomsen et al., 1999, 2001). Even when large quantities of OM are added to soil, the soil appears to retain a fixed proportion of the added C. For plant biomass an average of 15% of added C is retained (Thomsen & Christensen, 2004; Kristiansen et al., 2005), while some 30 to 40 % of the C supplied in animal manure is retained (Christensen & Johnston, 1997). Experiments with applications of organic matter over periods up to 30 years show no sign of OM saturation. However, some experiments indicate that it is difficult for soils under conventional arable management to maintain a C concentration above 2.5-3% (e.g. Kristiansen et al., 2005). A higher concentration can probably only be achieved under permanent vegetation. There is, however, insufficient documentation for the level of SOM that can be reached under long-term grass (Christensen et al., 2009). Also the effect of soil tillage intensity on the turnover of the more stable SOM pools is not currently settled.

Selected physical properties for some French soils with SOM levels beyond the 'capacity factor' were related to soil clay content rather than SOM content (Dexter et al., 2008). This observation is ground-breaking because it adds a functional dimension to the ongoing debate that has been dominated by the sequestration of carbon *per se*. Ingram & Fernandes (2001) acknowledged that soils receiving high inputs of OM (additions from external sources or root exudates / plant debris) may well have SOM contents higher than expected from their mineralogy. In accordance with Hassink (1997), Dexter et al. (2008) defined this 'pool' as the non-complexed OC (NCOC). For soils not having reached their 'capacity factor', Dexter et al. addressed the clay not "coated" with OC as the non-complexed clay (NCC). Considering arable soils relatively low in SOM, the interesting point is Dexter et al.'s conclusion that OC is determining essential structural characteristics until NCC≤0 (i.e., until the 'capacity factor' is reached) <u>across soils with different clay contents</u>.

Dexter et al. (2008) showed that non-complexed clay is more easily dispersed in water than is clay complexed with OC. We have previously studied a range of arable soils for their content of dispersible clay (Schjønning et al., 2002a). For six of these soils having satisfactory tilth conditions the clay/OC ratio averaged ~9.7. Data from the long-term fertilization experiment at Askov, Denmark, indicates that soil receiving either animal or mineral fertilizer at adequate rates for a century had a clay/OC ratio of ~9.5 (Schjønning et al., 1994; Munkholm et al., 2002). According to Dexter et al. (2008), the clay of all these soils is virtually saturated with OC (clay/OC~10). In contrast, soil kept unmanured for a century had an average clay/OC ratio of 11.7, indicating a pool of NCC. The unmanured soil displayed severe signs of structural degradation (e.g., weak in wet conditions, mechanically strong in dry conditions). One of the soils studied by Schjønning et al. (2002a) was also depleted in OC due to long-term continuous growing of small-grain cereals without any return of organic residues and manure to the soil. For that soil, the clay/OC ratio was as high as 13.7, and significant signs of degradation of soil structure were observed (Munkholm et al., 2001a; Schjønning et al., 2002a). A detailed study of water dispersibility of clay (WDC) by Elmholt et al. (2008) indicates, for soil samples collected across this Low-C soil – and only this soil – a correlation between WDC and the hot-water extractable C fraction. The above supports the theory suggested by Dexter et al. (2008) of a lower threshold of OC for sustaining the self-organization process in soil. Their data actually revealed a clay/OC ratio for 'saturation' of complexed OC on the clay particles in the range 8-11 (hence suggesting 10 as a suitable limit). This is in rather close agreement with the clay/OC ratios for the Danish arable soils reviewed above that have no tilth problems, while those displaying poor tilth conditions had higher values. The relation between clay and OC for the soils discussed here is shown in Figure 6.15.

6.3.3. Comparing management options with SOM-related soil functions

The simple Dexter et al. (2008) clay/OC concept discussed in the former section may be one tool for identifying sustainable soil physical conditions in relation to SOM content. The Dexter concept provides an indicator for soils that may exhibit satisfactory tilth conditions despite low OC contents as compared to structurally degraded soils with higher OC contents (e.g., the fertilized Askov soils compared to the Low-C soil of Group III). However, more studies are needed to reveal causal relationships behind the concept and to investigate the general validity of one specific clay/OC ratio across soil types. Except from some Australian soils, the capacity of a soil to preserve OC by its association with clay particles were not affected by the dominant type of clay mineral in the study by Hassink (1997). This was supported by the promising agreement between Polish and French soils (Dexter et al., 2008) and Danish soils (Fig. 6.15). Nevertheless, more studies are needed on this aspect. Furthermore, it is most likely that differences in SOM characteristics may compromise the universality of the clay/OC=10 ratio as an index of the physical/mechanical sustainability of arable soils. Some studies have also considered the silt fraction (2-20 μm) as being active in carbon sequestration (e.g. Hassink, 1997). Comprehensive reviews have shown that the silt sized fraction accounts for 20 to 40 % of the total OM content in Danish arable soils and that 2 to 10 % of the SOM is associated with the sand-sized fraction (Christensen, 1992, 1996). Moreover, it was found that the SOM enrichment of clay and silt was inversely related to the proportion of these size fractions in soil.

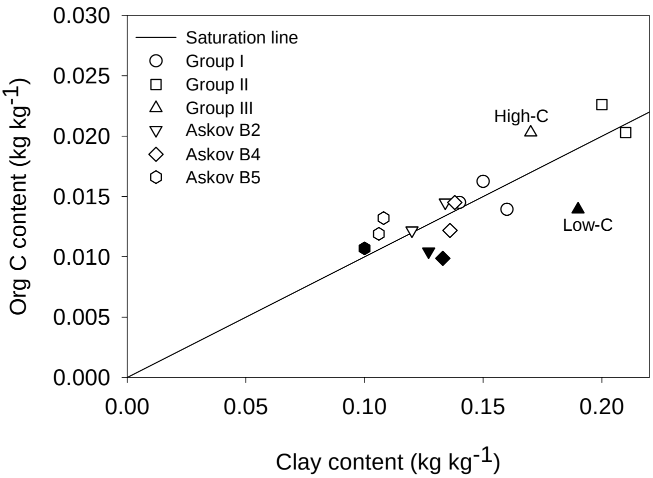

Figure 6.15. Relation between the content of clay and OC for a range of Danish soils with different soil management. Results on soil physical characteristics were reported for the Group I through Group III soils by Munkholm et al. (2001a) and Schjønning et al. (2002a), the Askov B2 and B4 fields were studied by Munkholm et al. (2002), and the Askov B5 soil by Schjønning et al. (1994). Soils with poor tilth conditions are shown by closed symbols. The soils labelled with 'Low-C' and 'High-C' are those also addressed in Table 6.1. The 'saturation line' represents n=clay/OC=10. Reproduced from de Jonge et al. (2009).

The clay/OC ratio concept and the derived NCC term should rather be regarded as an empirical and functional approach than as a true reflection of the interaction between mineral and organic particles in soil. We have previously emphasized the caveats in introducing indices in soil science (Schjønning et al., 2004). On the other hand, the whole exercise with the Soil Thematic Strategy and the upcoming Soil Framework Directive calls for workable approaches in combating the threats to a sustained soil quality. We find the Dexter et al. concept – despite its empirical nature – so promising that we suggest a dual approach: use of the concept in parallel with the 'management threshold' term defined by Schjønning et al. (2004) as "*the most severe disturbance any management may accomplish without inducing significant changes towards unsustainable conditions*". The management threshold idea relies on the apparent fact that for a given combination of climate and soil type, a given management system will induce a SOM level that ensures the soil functions and services that we want the soil to deliver. This was exemplified for friability of soils with different SOM contents (Fig. 6.14), where we identified management systems yielding satisfactory soil conditions: satisfactory fertilization and/or diversified crop rotations. However, the management threshold approach will probably not be applicable across large differences in climates, but it seems easier to cope with e.g. 3-5 climates within Europe rather than with hundreds of climate/soil combinations. In addition, management characteristics are directly assessable in an evaluation system.

More specifically, the results on management effects on SOM content reviewed in section 6.3.1 point to the following trends for Danish soil and climate conditions. Up to 30% of the SOM pool is turned over within a period of 20-30 years (Christensen, 1988, 1990; Bruun et al., 2003; Hansen et al., 2004; Kristiansen et al., 2005). However, the time taken for a new equilibrium in soil C pools to be reached following changes in soil management is often considerably longer (>50-100 years; Christensen & Johnston, 1997). In line with this, a number of long-term experiments with intensive cropping demonstrate a continuous decline in the level of SOM.

On sandy soils, and in particular the coarse sandy JB1 soils, SOM contents appear to increase. These soil types are dominated by cattle production systems that have a high frequency of grass in their crop rotation and intensive use animal manure as fertiliser. For the clayey soil types (for Danish conditions, soils with >10% clay are considered clayey), and for clay contents higher than 15% in particular, SOM contents appear to decline. The clayey soil types often support pig production systems typically dominated by cereal cropping and other cash crops and using mainly mineral fertiliser and pig slurry. This development questions the sustainability in straw removal on these soil types, and points to the potential of management measures such as the incorporation of catch crops to compensate for straw removal.

The burning of vegetation can result in the formation of recalcitrant black carbon compounds that subsequently accumulate in soil. When podzolic soils are cultivated, the chemically stable carbon from the B-horizon can become mixed with the plough layer. This carbon is also assumed to be biologically inert and combined with the black carbon from earlier burnings of the heathland it may form an important part of the relatively high carbon content often registered on coarse sandy JB1 soils. The significance of the historical burnings of vegetation, the pre-1990 burning of stubble and straw in the field, and the ploughing up of the B-horizon is at present uncertain, but may account for < 5 % of the total SOM pool.

The scope for increasing the accumulation of OM in agricultural soils via changes in land management is linked mainly to an increase in the recycling of plant residues, an expansion of the incorporation of straw, and a more frequent use of catch crops. Whether a more general transition to no-till practices in Denmark would induce a significant increase in the more stable SOM pools is not yet sufficiently clear. Although the issue of biologically inert C in

Danish soils is not yet sufficiently clarified, current agricultural management practices are not expected to lead to a significant and irreversible accumulation of SOM within an agronomically relevant timeframe.

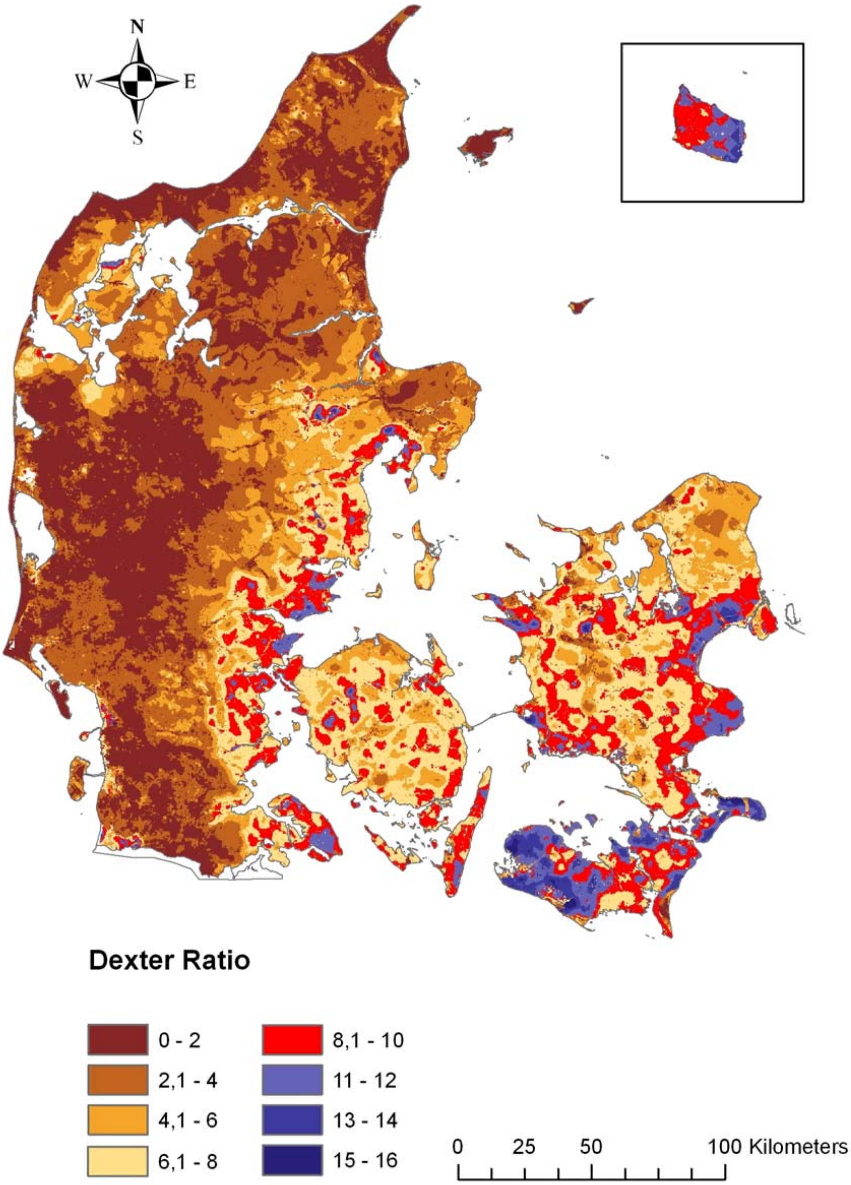

Figure 6.16. The ‘Dexter-ratio’ calculated as the ratio between the topsoil (0-20 cm) content of clay and organic C. Created from the Danish Soil Database (http://www.djfgeodata.dk) at Aarhus University by Mogens H. Greve.

6.4. Decisions on risk reduction targets

As outlined in the previous section, the use of a ratio between soil clay and OC contents as an index of the physical/mechanical sustainability of a soil calls for further scrutiny. Figure 6.16 shows the results of a preliminary study in which we have calculated this 'Dexter-ratio' for Denmark. The major part of the land displays values quite below the critical limit of clay/OC = 10 (yellow and the different shades of brown). These areas generally coincide with the sandy areas across most of western Jutland. In contrast, several areas with soils of morainic origin (generally the eastern part of the country), displayed in the different shades of blue indicate that the soil probably exhibits non-complexed clay (cf definition in section 6.3.2). This in turn is expected to reflect tilth problems. The areas in red are close to the critical limit, where a management strategy decreasing the content of SOM might turn the soil into a critical category. As mentioned previously, research is needed before the 'Dexter-ratio' would be safe for regulation of soil management. Nevertheless, we believe that a successful – perhaps modified – index would reduce significantly the difficulties in coping with the decline in SOM in Danish agricultural soils.

6.5. Programme of measures to reach risk reduction targets

It is primarily a political issue how land use and management should be regulated to ensure sustainable levels of SOM in Danish soils. As described in previous sections, recent achievements in our understanding of soil type differences in soil physical behaviour as related to its SOM content might serve as the basis for targets to be obeyed in agricultural use of the land. More research is needed to confirm the importance/viability/relevance of the approach. The Dexter et al. (2008) concept (Fig. 6.16) may turn out to be a useful tool to identify soils which can be used for the production of bioenergy crops and soils where straw for heating purposes can be removed without compromising their SOM-related quality properties.

As an alternative, we suggest using 'management thresholds' identified from expert judgement and modelling. This includes prescriptions for different management options that should be followed. Given the overwhelming problems in dealing with SOM *per se* as a universal indicator, we would find such management guidelines the most powerful approach in coping with the threat of decline in SOM.

6.6. Knowledge gaps and research needs

As has become evident from the discussion in the sections above and in accordance with the final report on the work with the EU Soil Thematic Strategy (van Camp et al., 2004), SOM *per se* is problematic as an indicator across soil types for the level of OM needed to secure the soil physical and mechanical functions and services needed in the agricultural use of soil. The study by Dexter et al. (2008) indicates that the ratio between clay and soil OC may rather be suitable as a predictor of soil structural conditions and tilth for arable soils low in SOM. Several previous studies have addressed the 'capacity factor' or 'organic matter saturation' terms. However, these studies have been performed with other objectives: 1) the desire to understand how organic matter is stabilized in soil, and 2) the need for quantification of the potential C sequestration in soil. Dexter et al. (2008) rather adopted a functional approach and found that soil tilth will not be much affected by soil organic C provided it exceeds the ‘capacity factor’. In contrast organic C contents below this ‘capacity factor’will have huge negative effect on soil structural properties. Dexter et al.’s study should be regarded primarily an ‘eye-opener’. There is an urgent need for its evaluation over a range of soil types and clay mineralogies. The concept should be developed towards the best expression of the ‘capacity factor’, which might require the inclusion of some estimate of mineral surface area in the silt and perhaps even in the sand fraction. It is also an open question to what extent the concept can be used across climatic regions or even across local variation within individual countries. Last but not least: the concept has as yet only been tested for soil physical functions and properties. We need research of soil biological and chemical properties as related to the clay/OC ratio.

We also see a need for much more studies on the effect of different management strategies on SOM. The ‘management threshold’ concept introduced by Schjønning et al. (2004) may be an operational tool not compromised by soil type differences. However, the present knowledge – as summarized for Danish conditions in this report – should be combined with models in the development of decision support systems.

7. Soil erosion by water

7.1. Is water erosion occurring in Denmark?

In this section water erosion is understood to be a process of accelerated erosion due to man's management of the soil resource, in contrast to the natural process of landscape formation. In Denmark spectacular soil erosion events are rare and erosion risk is generally perceived to be low due to a relatively low relief and low rainfall erosivity (e.g. Hansen, 1989; Van der Knijff et al., 2000; Veihe et al., 2003). Summarizing the few erosion studies, Veihe et al. (2003) gave a typical soil erosion rate of <3 t ha^{-1} yr^{-1} for Denmark. Erosion occurs on most soil types, typically in the autumn and winter after prolonged periods of rainfall, in connection with snowmelt and with rainfall on frozen soil.

In a systematic plot experiment, the effect of cropping and tillage on erosion was investigated at two sites in central Jutland, a loamy sand (Danish classification: JB4) and a sandy loam (JB6), over a period of three years (Schjønning et al., 1995). The plots had linear slopes of ca. 10% and corresponded to USLE (Universal Soil Loss Equation) erosion plots. Annual erosion rates in winter wheat sown along the slope varied between 1.7 and 26 t ha^{-1} at the sandy site and 0.2 and 1.7 t ha^{-1} at the loamy site. The differences were explained by soil texture, soil structure and, to a lesser extent, somewhat different rainfall patterns. Lower erosion rates were measured in different plot experiments in western Jutland and eastern Denmark (Hasholt, 1990). However, in those experiments the slope gradients were either <4% and the soils coarse sandy, or ploughed plots lay bare in winter, substantially reducing the erosion risk due to larger depression storage capacity (Hansen et al., 1999).

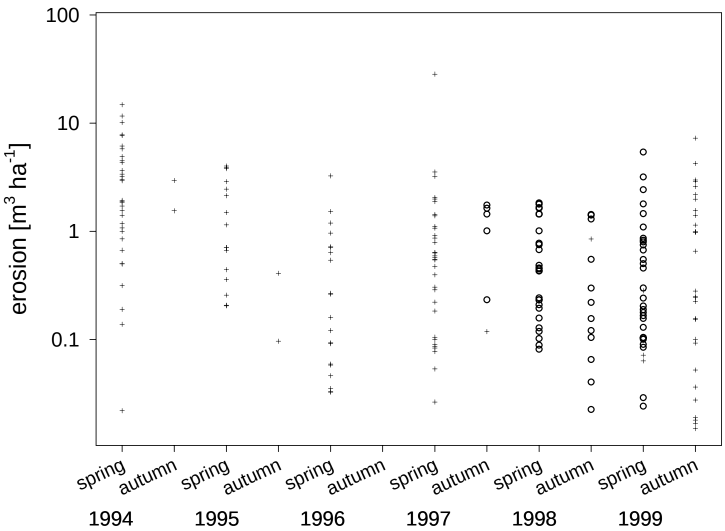

Figure 7.1. Measured rill erosion rates in spring and autumn on 213 slope units in Denmark (Djurhuus et al., 2007).

In an extensive field study, rill erosion was surveyed on 189 slopes units in different parts of Denmark in the autumn and spring between 1994 and 1999. Loamy sands and sandy loams were the dominant soil types. In only 20% of the surveys could rill erosion be detected. The mean annual erosion rate on all slopes was 0.6 t ha^{-1}. The non-zero erosion rates were highly right-skewed, with a median of 0.7 t ha^{-1} yr^{-1}, a 75% quantile of 1.9 t ha^{-1} yr^{-1} and a maximum of 37 t ha^{-1} yr^{-1} (Fig. 7.1; Djurhuus et al., 2007). Where erosion occurred, the distribution of rill erosion rates was broadly similar to those reported from other northern and western European field studies (e.g. Alström & Bergmann, 1990; Broadman, 1990; Govers, 1991; Chambers & Garwood, 2000).

In absence of a soil erosion model for Denmark, the relative erosion risk in the country was mapped with the spatially distributed soil erosion model WaTEM (Van Oost et al., 2000), which is based on the USLE (Renard et al., 1996). Accordingly, 10% of the agricultural land was vulnerable to soil erosion by water in 2003, assuming that winter wheat was grown on the whole area (Heckrath et al., unpublished data). Unlike the traditional USLE, WaTEM calculates the upslope contributing area of a given point instead of the slope length as part of the topographic factor (L) in the model (Desmet & Govers, 1996). Therefore WaTEM accounts more accurately for runoff patterns on complex topography. Both the erodibility (K) and the erosivity (R) factors of the USLE were adapted for Danish conditions.

7.2. What is the impact of water erosion to the soil and the environment?

Accelerated water erosion is both a threat to the soil resource and may cause pollution, especially eutrophication, of surface waters. In Denmark only the latter has been of concern and was the prime motivation for initiating erosion studies. The adverse effects of soil erosion on soil quality and agronomic productivity are well-documented in the literature (e.g. Lal, 2001; Govers et al., 2004). Similar to elsewhere, the most severe effects at eroding sites in Denmark comprise the loss of fine-textured material, organic matter, nutrients and available water capacity as well as a general decline in soil structure. Colluvial deposits on sites with sandy, weakly structured soils are often depleted in clay and organic matter. Although these processes impair soil productivity in the long term, it is difficult to quantify the productivity loss due to soil erosion, because of the confounding effects of climatic factors, landscape position and management. Measures other than soil productivity may need to be included to assess the potential threat to the soil.

To our knowledge, there are no specific reports on water erosion-induced soil degradation for Denmark. Likewise, a critical erosion rate, beyond which lasting damage of the soil occurs, has not been defined. Compared with the rate of soil development, which in temperate Denmark is substantially less than 1 mm yr^{-1}, even small erosion events can impair the integrity of the soil pedon. On the basis of the extensive rill erosion survey, however, we conclude that water erosion in Denmark presently is a minor threat to the agronomic productivity of soils.

7.3. Identification of risk areas regarding water erosion

Models are commonly used for identifying areas vulnerable to erosion. Most of those combine expressions of erosivity and erodibility with topographic and crop management functions. In Denmark the USLE has been used as a qualitative indicator of erosion risk

(Olsen & Kristensen, 1998). However, as water erosion is a relatively infrequent and highly variable event under Danish conditions, the probability of erosion occurring ought to be explicitly considered in modelling. To this end, data from the rill erosion survey (see Section 7.1) were used to develop an expert system for predicting water erosion in Denmark (Djurhuus et al., 2007).

7.3.1. Water erosivity for Danish climatic conditions

Climatic or rainfall erosivity is a cause of regional variations in water erosion potential. In a recent study the magnitude and frequency of erosive rainfall was determined for Denmark for the period 1954 to 1996 (Leek & Olsen, 2000). To this end, an empirical submodel of the USLE, the rainfall erosivity index 'R', was calculated based on the rainfall energy and intensity. The model quantifies the net effect of the kinetic energy of the raindrops on impact with the soil, and the amount and rate of runoff likely to be associated with the rain.

Table 7.1. Average annual rainfall erosivity for four selected years at six weather stations in Denmark (Leek & Olsen, 2000).

Year	1967	1990	1993	1994
Mean erosivity, R, (MJ mm m^{-2} hr^{-1})	0.026	0.045	0.030	0.053

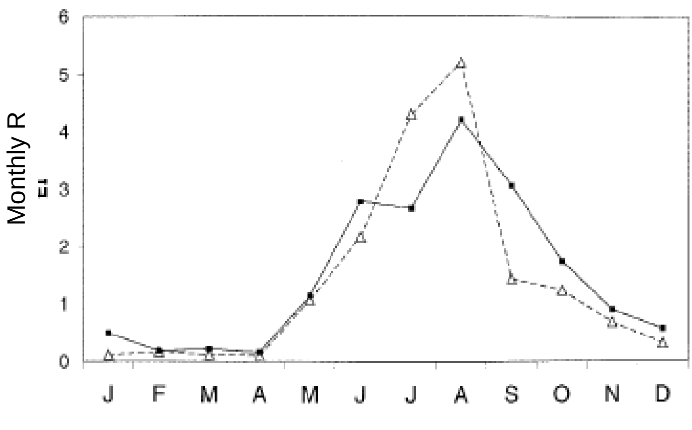

Figure 7.2. Annual distribution of monthly rainfall erosivity (R) for the linear least square trend 1954-96 for six weather stations. The dotted line represents 1954 and solid line 1996 data (adapted from Leek & Olsen, 2000).

There was a considerable but random regional variation of erosivity during the period (Leek and Olsen, 2000). That is, the geographic location of high erosivity will vary considerably from year to year. Table 7.1 shows average annual values from six weather stations for four years. Rainfall erosivity in Denmark was low compared with other parts of northern and western Europe, (Morgan, 1995, Chapter 4) and an order of magnitude lower than erosivity in the eastern half of the USA (Wischmeier & Smith, 1978). Erosive rainfall increased between

1954 and 1996 and so did erosivity, especially in the autumn (Fig. 7.2). This rising trend in annual precipitation in Denmark is expected to accelerate according to the latest projections of climate change (e.g. Jeppesen et al., 2009) and hence rainfall erosivity in autumn. This coincides with the time of the year when soils tend to be rather vulnerable to water erosion.

The newly-developed expert system for predicting water erosion consists of three parts: a logistic model predicting the probability of erosion, a model predicting a conditional erosion rate only in cases where erosion occurs, and a third model predicting the soil surface roughness. Surface roughness, in turn, is a variable in the probability model. The number of storms with >20 mm rainfall only had a minor effect on the probability of erosion. In contrast, the magnitude of erosion depended strongly on the annual amount of erosive rainfall. This underlines the importance of the climate change-induced rising rainfall erosivity for the erosion potential in Denmark.

Crop and soil management are decisive for the extent of water erosion. Plant cover and crop residues protect the soil from raindrop impact, while roots contribute to the mechanical strength of soils. In the long term, crop management affects soil structure and hence both soil strength and water infiltration, which, in turn, are important factors for water erosion (see Section 5.2). Tillage and seedbed preparation determine soil surface roughness and therefore depression storage capacity (Hansen et al., 1999). Ploughed land with its large capacity to store water delays runoff initiation. Contour tillage and planting is often reported to reduce soil loss under low rainfall intensity, as micro-topographic features reduce runoff velocity. Experimental evidence both from controlled plot experiments and the rill erosion survey stresses the vulnerability of winter cereals to water erosion in Denmark, especially when sown along the slope (Schjønning et al., 1995). In contrast, erosion was practically absent on fields with grass, catch crops or cereal stubble and intermediate for ploughed soil (Djurhuus et al., 2007).

7.3.2. The ability of Danish soils to withstand the mechanical stresses from surface water runoff

Erodibility describes a soil's inherent resistance to erosion, i.e. both detachment and transport. Hence, erodibility is defined independently of other factors controlling water erosion like rainfall, topography and crop management. Erodibility varies with soil texture, soil structure, soil strength, and soil chemical composition. The least resistant particles to detachment and transport are silt and fine sand. Coarse sand requires greater forces for entrainment, while clay makes soils more cohesive and resistant to detachment. Accordingly, loamy sand and sandy loam soils prevalent in Denmark are typically moderately erodible (Renard et al., 1996). An erodibility field test on 11 representative Danish soils showed that clay was the single most important parameter explaining soil loss for a range of clay contents between 4 and 20% (Schjønning et al., 1995, Chapter 11). In this study, clay content also correlated positively with soil porosity, which in turn increased infiltration and reduced runoff. Similarly, both the probability and the magnitude of erosion declined in soils with increasing content of the combined clay and fine silt (2-20 µm) fraction in the Danish rill erosion survey. This could be explained by higher aggregate stability and surface roughness with increasing clay and fine silt content, both of which reduce erodibility.

One of the most commonly used erodibility indices is the K factor of the USLE (Wischmeier & Smith, 1978), which describes soil loss on a standard erosion plot per unit of the erosivity factor R. From a large amount of American experimental data, a soil texture-based pedotransfer function for the K factor was developed that greatly facilitates erodibility estimates. However, although the K factor pedotransfer function is well established for agricultural soils in the USA, employing it for other parts of the world may be subject to error due to variations in soil properties. A weakness of the K factor, in general, is that it does not account for seasonal patterns in erodibility related to, for example, tillage-induced changes in bulk density and hydraulic conductivity or freeze-thaw cycles.

Despite these limitations, we calculated the K factor according to Renard et al. (1996) from data of the Danish Soil Database (http://www.djfgeodata.dk), because soil conditions in Denmark were considered sufficiently similar to those in the USA. The use of the K factor pedotransfer function thus permitted the mapping of erodibility in Denmark at 250 m-resolution and the comparison with published data. Organic lowland soils were excluded from the analysis. The K factors for agricultural land were broadly similar on the two main islands, Sealand and Funen (Table 7.2), which are dominated by sandy loams developed on glacial till. Geology and soils are somewhat more versatile in Jutland. Fine sands in the north, coarse sands on the outwash plain in the west and loamy soils on glacial till in the east result in a broader distribution of K factors in Jutland (Table 7.2). The K factor is commonly divided into erodibility classes where K factors below 25 and above 45 indicate low and high erodibility, respectively (Römkens, 1985). Thus, most agricultural soils in Denmark are classified as moderately erodible. Jutland also had a substantial proportion of soils with low erodibility and the only highly erodible soils in Denmark, Aeolian sands, were found in the north of Jutland and occupied about 3% of its agricultural land.

Table 7.2. Summary statistics of the K factor (kg hr MJ^{-1} mm^{-1}) calculated for agricultural land in different parts of Denmark in 2003 excluding organic lowland soils (unpublished data).

	25^{th} percentile	median	75^{th} percentile	90^{th} percentile
Jutland	20	28	33	38
Funen	30	33	35	36
Sealand	32	34	36	38

Topography often exerts a large control on water movement on the soil surface. The topographic effect is here understood as the system's resistance to an external pressure, namely rainfall erosivity. Water erosion usually increases both with slope steepness and size of the upslope drainage area as a result of respective increases in velocity and volume of surface runoff. One of the most well-known attempts to describe this combined effect is in the form of the length-slope (LS) factor of the USLE (Wischmeier & Smith, 1978), which is linearly correlated with soil loss. To characterize the topographic effect on water erosion in Denmark, we used a modified, two-dimensional form of the LS factor. This modified LS factor incorporated upslope drainage area rather than slope length and, therefore, better represented surface runoff on complex topography (Desmet & Govers, 1996). On the basis of a 10-m digital elevation model, LS factors were calculated for field blocks, with administrative units each comprising one or more agricultural fields and surrounded by permanent landscape features. This procedure set a natural limit for the size of the upslope drainage area. The average field block size was 11.6 ha. The distributions of LS values were broadly similar between the three major regions in Denmark (Table 7.2). In general, LS values were rather low, consistent with the countries' mostly low relief. There are very few reports in the literature with which compare. In a Belgian catchment in the loess belt with widespread erosion problems, average LS values on moderately eroding sites were larger than the LS 90th percentiles in Denmark (Desmet & Govers, 1996). In some parts of Denmark, especially the eastern half of northern Jutland, eastern Jutland and western Funen, moderately high LS values >15 occurred on footslopes, comparable with observations from Belgium.

Table 7.3. Summary statistics of the dimensionless 2D LS factor calculated for agricultural land (2003 data) in different parts of Denmark (unpublished data).

	median	75th percentile	90th percentile	99th percentile
Jutland	0.3	0.9	2.3	9.5
Funen	0.4	1.1	2.6	9.1
Sealand	0.3	0.9	2.1	7.8

In the Danish rill erosion survey, both the probability and the magnitude of erosion were found to increase with increasing LS values for the slope units. However, the effect was rather weak compared to other variables such as cropping system and presence of an impermeable layer in the root zone. By and large, topography had little influence on the soils' vulnerability to water erosion in Denmark.

7.3.3. Relating climatic erosivity to the soil's resistance to degradation

There is no simple spatial pattern of erosion events on sloping land in Denmark. Rainfall erosivity varies randomly between regions, erodibility is rather uniform and the relief is generally low. On the other hand, crop management and subsoil conditions have a strong influence on both the probability and the magnitude of water erosion (Djurhuus et al., 2007). Therefore, the assessment of erosion risk ought to be done at the field scale. Most erosion models predict erosion risk by relating expressions of erosivity to the soils' ability to resist soil loss. This approach was also used for the development of the water erosion expert system in Denmark (Djurhuus et al., 2007), which makes it directly suitable for the purpose of identifying risk areas. Some important variables of the expert system have been described in the previous sections. Figure 7.3 shows as an example the effects of different variables on predictions of median erosion rates for selected scenarios representing typical variable combinations. For the first scenario (Fig. 7.3 a), the variability of the predicted erosion rates is also shown as quantiles (Fig. 7.3b).

The largest median erosion rate of up to 3 m^3 ha^{-1} (c. 4 t ha^{-1}) was predicted for winter cereals on a south-facing slope with an impermeable layer and 700 mm precipitation accumulated over days with >8 mm precipitation (Fig. 7.3 c). This scenario shows the relatively minor effect of aspect compared to the importance of impermeable layers. As impermeability often is the result of soil compaction (Chapter 5), this is an example of how one aspect of soil degradation amplifies another. Whereas topography had little influence on a soil's vulnerability to water erosion (Fig. 7.3d), the effect of soil texture on erodibility was more marked, especially for small grains (Fig. 7.3a). The expert system also predicts the variability of water erosion risk for given variable combinations. The example in Figure 7.3b shows substantial tailing towards higher erosion risk. This information is particularly useful for identifying high-risk areas due to a larger differentiation of risk than the prediction of median erosion is able to.

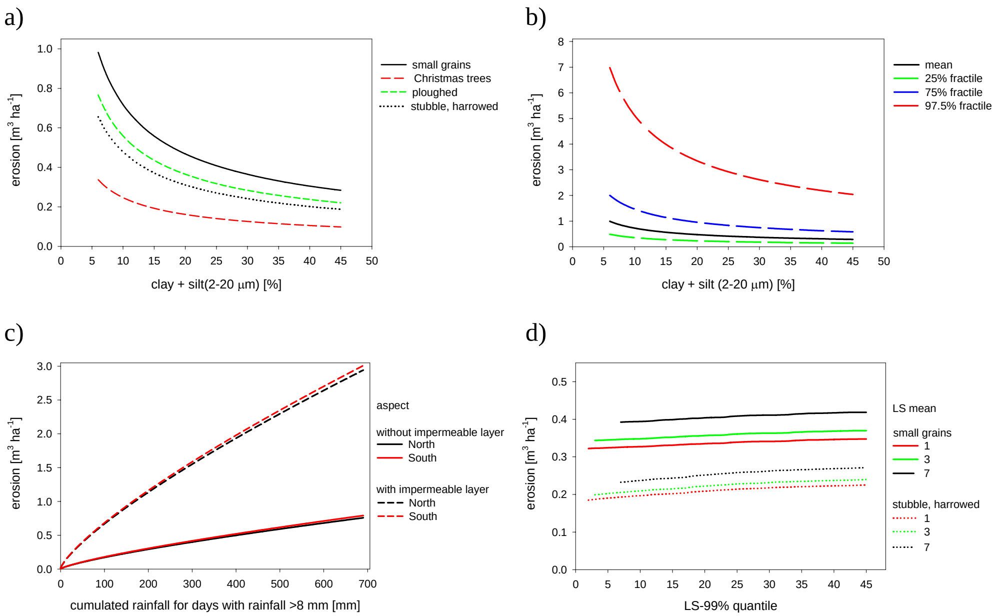

Figure 7.3. Predicted median erosion (m^3 ha^{-1}) as a function of clay+silt (a), accumulated precipitation for days with precipitation > 8 mm (c) and LS-99% quantile (d). Also, (b), the mean and the 25, 50 and 97.5% quantiles from the distribution of predicted erosion rates for small grains (winter cereals) corresponding to (a). For the variables that are not explicitly listed in the figures the following settings were used: clay+silt: 30%; water impermeable layer: not present; aspect: north; precipitation and snowmelt on frozen soil: 12 mm (a) and 0 mm (c, d); days with precipitation > 20 mm: 2 (a) and 0 (c, d); accumulated precipitation for days with precipitation > 8 mm: 230 mm; LS-mean: 2.75 (a) and 1.5 (c); LS-99% quantile: 15 (a) and 7 (c) (unpublished data).

7.4. Decisions on risk reduction targets

The extent of soil degradation due to water erosion has been little investigated in Denmark and, to our knowledge, there has barely been a technical debate regarding tolerable levels of soil loss. As pointed out above, water erosion was only seen as a potential threat to the aquatic environment. Lacking scientifically documented critical thresholds, we argue that the preservation of the long-term integrity of the soil resource ought to take precedence over a more short-term abatement strategy based on documented soil productivity loss. This is consistent with the precautionary principle manifest in the EU Soil Thematic Strategy. Therefore, water erosion ought to be prevented on agricultural land where it is likely to frequently cause rill erosion. The critical factor combinations are assessed by employing models like the rill erosion expert system. This categorical approach to risk reduction implies that the vulnerability to rill erosion is confirmed locally.

7.5. Programme of measures to reach risk reduction targets

Water erosion reduction measures are widely reported in literature (e.g. Morgan, 1995) and well tested. Principally, conservation strategies aim at increasing water infiltration to reduce runoff volumes and erosivity, strengthening topsoil resistance to detachment of soil particles and protecting the soil surface with plant or residue cover (e.g. Govers et al., 2004). Soil cover both enhances infiltration and reduces detachment. The same effects are achieved by maintaining an adequate soil organic matter (SOM) content and lime potential, which results in good soil structure, macroporosity and aggregate stability (Chapter 6). A well-developed root system furthers both soil cohesion and macroporosity and therefore affects infiltration and detachment. A rough soil surface has a large depression storage capacity for rainfall allowing more time for infiltration and dissipating runoff energy. Surface roughness is controlled by tillage practices. Wheel tracks typically have a low infiltration capacity and are important for runoff generation, which is why they should be removed mechanically.

Poulsen & Rubæk (2005) listed a number of suitable conservation options for Denmark, which practically eliminate the erosion risk. The simplest and most cost-effective are adapted crop and soil management to minimize erosion risk. On vulnerable areas, winter cereals ought to be omitted unless sown by direct drilling. Alternative options are catch crops or harrowed stubble during the runoff season. As a measure of landscape engineering, grassed waterways can be established across the steepest slopes breaking runoff connectivity.

7.6. Knowledge gaps and research needs

In Denmark research needs regarding water erosion mainly evolve in the context of protecting the aquatic environment. However, the following issues also are relevant in view of soil protection:

- Quantitative analysis of water erosion impact on long-term soil fertility and soil functions in different agro-landscapes.
- Assessment of climate change impact on water erosion risk including anticipated changes in cropping systems.
- Implementation and evaluation of the Danish expert system as a practical, web-based tool for erosion control and conservation planning.

8. Soil erosion by tillage

8.1. Is tillage erosion occurring in Denmark?

Whenever soil is cultivated, tillage moves very substantial amounts of soil in the cultivation layer. Spatial variations in the magnitude of soil movement during tillage along a hillslope cause net gain or loss of soil locally. This process is referred to as tillage erosion (Govers et al., 1999). Characteristically, tillage erosion removes soil at convexities such as crests and shoulder slopes and deposits it again at the concavities of footslopes and hollows. The linear slope sections remain stable. Hence tillage-induced soil redistribution primarily depends on the change in slope gradient in tillage direction. This means that the rate of tillage erosion often is described as a simple linear function of slope curvature (Govers et al., 1994). Field boundaries will act as a line of zero transport, so that soil loss will take place at upslope field boundaries, whereas accumulation occurs at downslope field borders, leading to the formation of soil banks. The spatial signature of tillage erosion differs fundamentally from that of water erosion: soil loss by tillage will be most intense on landscape positions where water erosion is minimal. Another important difference between water and tillage erosion is that the latter is solely a process of soil redistribution within fields. In contrast to often unreliable data on the extent of water erosion, tillage erosion estimates depending only on topography and tillage management are considered robust (Van Oost et al., 2006).

Given the nature of the process, tillage erosion must be assumed to be widespread on rolling topography in Denmark. However, only a few field surveys (Van Oost et al., 2003; Heckrath et al., 2005) and controlled tillage experiments (Heckrath et al., 2006) have been conducted in Denmark. This research was supported by findings from other countries and has fundamentally contributed to the development of tillage erosion and soil property evolution models (Van Oost et al, 2005; Van Oost et al., 2007). Systematic tillage experiments involving physical tracers showed tillage erosion rates between 10 and 20 t ha^{-1}, depending on tillage direction on typical slopes in central Jutland (Heckrath et al., 2006). In recent years the spatial patterns of the fallout radionuclide caesium-137 (^{137}Cs) have increasingly been analysed to trace soil redistribution on agricultural land over the last 40 to 50 years (Quine, 1999). This methodology was employed on a field with representative hummocky topography in northern Jutland. In summary, eroding sites averaged soil losses of 20 t ha^{-1} yr^{-1} due to tillage, whereas depositional sites received about 10 t ha^{-1} yr^{-1}. One third of the field exhibited tillage erosion rates larger than 15 t ha^{-1} yr^{-1}, while another third correspondingly had substantial rates of soil deposition. Areas with maximum tillage erosion had lost about 0.15 m of soil over a period of 45 years (Heckrath et al., 2005). These results were confirmed at another field site in eastern Jutland (Heckrath, unpublished data).

It is therefore concluded that tillage operations have caused severe soil redistribution on arable fields in Denmark during the past decades – and still do – and that tillage erosion rates are at least an order of magnitude higher than for water erosion.

8.2. What is the impact of tillage erosion to the soil and the environment?

Tillage erosion is today recognized as an important process of soil degradation affecting soil productivity (Lal, 2001) or landscape evolution (Quine et al., 1997). Close relationships between tillage erosion and the spatial pattern of soil organic carbon (SOC), total soil N, P, soil depth, available water capacity and above-ground biomass have been reported from different parts of Europe (e.g. Kosmas et al., 2001; Quine & Zhang, 2002; Heckrath et al., 2005). These studies have provided evidence that tillage erosion operates like a conveyor belt, transporting soil and soil constituents from convexities to concavities. Tillage causes severe soil truncation and loss of plough layer soil on convexities. Subsequently, as ploughing depth is maintained, less fertile subsoil material is incorporated into the plough layer and eventually leads to its degradation. Accordingly, areas of lighter soil colour around convexities, commonly observed in Denmark, are evidence of tillage-induced SOC depletion. Conversely, soil accumulates in concavities through downslope translocation from the upslope landscape elements. These relatively small areas, therefore, develop deep A horizons due to perpetual burial of the former plough layer. Hence, soil movement by tillage erosion is a major contributor to within-field variability of soil properties (Quine & Zhang, 2002; Heckrath et al., 2005) and has an adverse impact on soil productivity (Schumacher et al., 1999).

The effect of tillage erosion on soil properties was comprehensively studied at the field site in northern Jutland (Heckrath et al., 2005). Soil organic carbon and phosphorus contents in soil profiles increased from the shoulder towards the slopes. The significance of tillage erosion for soil profile anisotropy at this site was illustrated by a comparison of averaged soil property values in different erosion classes (Table 8.1). Stable areas were represented by tillage erosion rates of –5 to +5 t ha^{-1} yr^{-1}. While SOC contents in the 0-0.25 m layer were 13% higher on aggrading compared to eroding areas, the difference was 38% in the 0.25-0.45 m layer. The ratio between SOC contents in the 0-0.25 m and the 0.25-0.45 m layer was higher on eroding compared to aggrading areas. Ignoring dynamic processes of SOC turnover, a first approximation of SOC redistribution due to tillage erosion between erosion classes was calculated based on the plough layer SOC concentrations and the soil redistribution rates. We obtained SOC changes of -220 and 150 kg SOC ha^{-1} yr^{-1} on eroding and aggrading areas, respectively (Table 8.1).

These results suggest that tillage erosion has important implications for SOC storage at the field scale. Eroded SOC is deposited in a subsoil environment with assumed much longer turnover times (Gill & Burke, 2002). Additionally, denuded shoulderslope positions may bind extra atmospheric carbon (Liu et al., 2003). To further investigate SOC fluxes induced by soil redistribution at this site, C dynamics were incorporated in a spatially distributed model including both water- and tillage-induced soil redistribution (SPEROS-C; Van Oost et al., 2005). The SOC patterns predicted by SPEROS-C were in good agreement with field observations. The model results confirmed that in fields with gently rolling topography, tillage erosion and deposition exert a large influence on SOC redistribution and soil profile evolution at a timescale of a few decades. The formation of new SOC at eroding sites and the burial of eroded SOC below plough depth provided an important mechanism for C sequestration on sloping arable land in the order of 30–100 kg C ha^{-1} yr^{-1}. These findings have been supported subsequently by results from a number of arable field studies in different parts of the world (Van Oost et al., 2007). Therefore, any attempt to manage agricultural land to maximize C sequestration must fully account for tillage erosion and the fate of eroded and buried SOC across the landscape. Increasing variability of SOC contents in soils will directly affect soil properties such as soil structure and aggregate stability and, in turn, nutrient cycling water retention and erodibility by water (Chapter 7). With declining SOC contents in topsoils, clay is to a lesser degree associated with SOC and therefore more readily dispersed. This may give rise to a larger colloid mobilization and translocation in soil profiles (Dexter et al., 2008).

Like SOC, total P was another soil property that evidenced a spatial distribution that appeared to be strongly affected by tillage erosion. There was a lower rate of decline of total P in soil profiles on aggrading compared to eroding areas and therefore a larger P enrichment of the subsoil. The overall evidence also suggested that crop productivity was affected by tillage-induced soil redistribution (Table 8.1). However, tillage erosion effects on crop yield were confounded by topography-yield relationships, and the effects could not be clearly separated.

Table 8.1. Arithmetic means of spatially interpolated (block-kriged) soil properties for eroding, aggrading or stable areas at a field site in northern Jutland. Changes in SOC and total P are the product of erosion rate and the concentration of the respective soil property (Heckrath et al., 2005).

Property		Eroding areas 48%	Stable areas 20%	Aggrading areas 32%
Soil redistribution rate	t ha^{-1}	-20.2	0.4	11.6
SOC $_{0\text{-}0.25}$ †	g kg^{-1}	11.5	13.3	13.2
SOC $_{0\text{-}0.25}$	t ha^{-1}	38	43	43
SOC $_{0.25\text{-}0.45}$	t ha^{-1}	13	17	18
Change SOC‡ $_{0\text{-}0.25}$	kg ha^{-1} yr^{-1}	-224	7	152
Total P $_{0\text{-}0.25}$	mg kg^{-1}	446	556	645
Total P $_{0\text{-}0.25}$	kg ha^{-1}	1480	1820	2100
Total P $_{0.25\text{-}0.45}$	kg ha^{-1}	840	940	1230
Change Total P‡ $_{0\text{-}0.25}$	kg ha^{-1} yr^{-1}	-8.6	0.4	7.6
A_h	m	0.26	0.31	0.34
Grain yield	t ha^{-1}	6.1	6.8	7.2

† label indicates soil depth in metres; ‡ soil redistribution rate times soil content.

8.3. Identification of risk areas regarding tillage erosion

Unlike water and wind erosion, the effects of which can often be easily identified in the landscape, the extent and severity of tillage erosion only become apparent after several decades of tillage through variations in soil properties and the development of tillage-related landforms like soil banks. This is why attention has focussed on wind and water erosion and why tillage erosion has only been sporadic investigated in the last 20 years.

The extent of tillage erosion has not yet been mapped for the arable land in Denmark. However, the modelling concepts, the technology and the data are available for simple risk assessment tools to be developed.

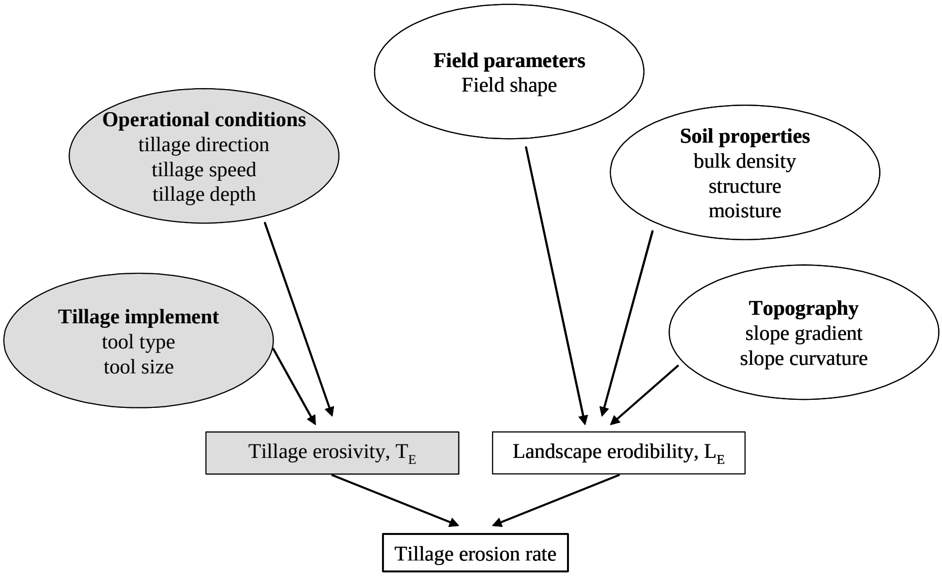

Figure 8.1. Factors affecting tillage erosion. The process of tillage erosion can be seen as a function of the erosivity of a given tillage operation (TE) and the erodibility of the cultivated landscape (LE). This simple general concept is the basis for the following discussion. Adapted from Lobb et al. (1999).

8.3.1. Which tillage operations induce downslope movement of soil?

Compared to other soil erosion processes, tillage erosivity arises solely from soil management. Tillage erosivity – the potential of a given tillage operation to erode soil within a landscape – depends on several physical and human factors (Fig. 8.1). These include the type and the size of the tillage implement and the operational conditions such as tillage speed and depth, as well as direction. The practical relevance of tillage erosivity is that it corresponds to the proportionality factor relating slope curvature to tillage erosion rates (Govers et al., 1994). The mouldboard plough is the primary tillage implement in Denmark and in general the most erosive. However, some types of chisel ploughs are only slightly less erosive when used at tillage depths similar to the mouldboard plough (Van Oost et al., 2006). Tillage erosivity for particular implements has been studied intensively in the 1990s by measuring the translocation of physical or chemical tracers in controlled experiments on various landscape positions. Tillage erosivity was shown to increase with tillage speed and depth (Van Muysen et al., 2002; Heckrath et al., 2006). Pooling the results from 34 published international studies with mouldboard ploughs, Van Oost et al. (2006) found that different erosivities were to a large degree explained by the variation in tillage depth and to a lesser degree by speed. For example, an increase in tillage depth from 0.2 to 0.3 m resulted in a 150% rise in erosivity. Tillage direction also exerts an important influence on mouldboard erosivity on rolling topography. Heckrath et al. (2006) concluded from controlled tillage experiments in Denmark that erosivity increased in the order contour tillage, slantwise tillage turning the soil upslope, and up- and downslope tillage.

8.3.2. Which topographies are critical

Landscape erodibility is the propensity of a landscape to be eroded by tillage and depends on topography, physical properties of the soil as well as field size and shape (Fig. 10.1). The latter affect the generation of soil banks. The static variable topography exerts the dominant effect on landscape erodibility by tillage and hence its resistance to displacement. Soil translocation during tillage is a gravity-driven process, where translocation rates depend on the slope gradient. The rates are highest on steep slopes tilling in downslope direction and vice versa. With changing slope gradient in tillage direction, the masses of soil transported from and to a given point differ, causing either net soil loss or gain at this point (Govers et al., 1994). Therefore, the variation of slope curvature determines the magnitude of tillage erosion. A fragmentation of such topography into small fields increases the risk as soil banks become more abundant. The potential for tillage erosion has not been mapped for Denmark and the precise area of arable land affected by the process is unknown. However, circumstantial evidence suggests that tillage erosion is prevalent and extensive in all moraine landscapes and, hence, on the majority of arable land in Denmark.

In Danish tillage experiments, tillage erosivity declined with increasing bulk density as the soil became more compact and by extension more cohesive. The effect was smaller than that of tillage speed, and for a rise in bulk density from 1300 kg to 1700 kg m^{-3} erosivity declined by 20% (Heckrath et al., 2006). However, following primary tillage, loose soil was shown to be much more vulnerable to tillage erosion than compact soil (Van Muysen et al., 1999).

8.3.3. Critical combinations of tillage operations and topography

Tillage erosion is ubiquitous on rolling topography in Denmark. The more undulating the landscape, the larger is the vulnerability to tillage erosion. Mouldboard ploughing parallel or in a steep angle to the aspect is the most erosive tillage operation. Therefore, the most erosive tillage operation also is the most common. Tillage erosion is exacerbated by variable tillage speed and depth, especially when downslope tillage is faster and deeper than upslope tillage. As tillage erosivity essentially only depends on the tillage operator, the variation of erosivity has a large random component. This introduces a degree of uncertainty regarding the predictability of tillage erosion. On the other hand, the operator’s control over erosivity also provides him with control over the magnitude of tillage erosion and suggests simple mitigation options. Field borders across a hillslope give rise to the formation of soil banks. Therefore, the partition of long slopes into separate fields increases the potential for tillage erosion.

As with water erosion, thresholds for critical soil truncation or soil burial have not yet been defined. Hence, we are currently unable to determine a tolerable range of tillage erosion rates. To show the effect of different operational conditions on tillage erosion rates we used a simple model (Van Oost et al., 2006) to calculate scenarios for mouldboard tillage parallel to the aspect on two convexities with constant curvature (Fig. 8.2). The large curvature implied a change in slope gradient from 11.3 to 5.7 degrees over a distance of 12.5 m, while the distance was 50 m for low curvature. Both curvature values are typical for rolling topography in Denmark; the lower value is more common. Even on the minor convexity, tillage erosion rates reached 5 t ha^{-1} for the typical ploughing depth and speed of respectively 0.25 m and 5 km h^{-1} (Fig. 8.2). These erosion rates were high compared to water erosion rates in Denmark (see 9.1). Only by reducing tillage depth to 0.15 m and speed to 3 km h^{-1} did tillage erosion rates remain at about 1 t ha^{-1}. Under the same low erosivity conditions, tillage erosion rates were about 5 t ha^{-1} for the large curvature.

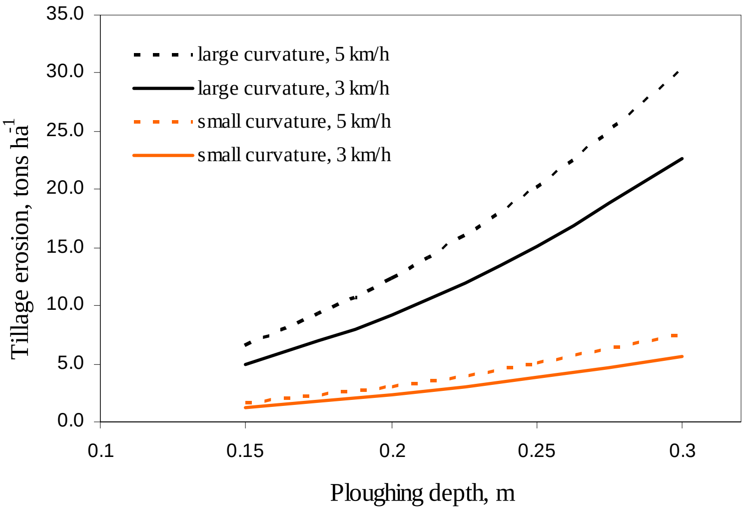

Figure 8.2. Predicted tillage erosion rates for mouldboard ploughing as a function of tillage depth, speed and topography. The tillage direction is parallel to the aspect and the soil bulk density 1400 kg m^{-3}. Based on Van Oost et al. (2006).

8.4. Decisions on risk reduction targets

Since both tillage-induced soil loss and soil accumulation within fields are much more severe and widespread than for water erosion, tillage erosion must be considered a substantial long-term threat to soil productivity in Denmark. Measured data and model scenarios from Denmark and other parts of northern Europe with similar topography and tillage intensity provide clear evidence that tillage erosion rates frequently exceed 20 t ha^{-1} yr^{-1} on eroding sites within fields. Next to land levelling, tillage erosion is the most severe process of human-induced soil redistribution in Denmark. In other words, tillage in its current intensity is 10 to 100 times as erosive as water erosion and it is much more widespread. Hence, it is incontrovertible that tillage erosion will inflict substantial cost on the agricultural sector due to loss of productivity on eroded sites or the implementation of fertility–enhancing measures in the long term. Consequently, we stress that concerted efforts should be made to minimize tillage erosion.

Despite the potential threat, surprisingly few have tried to quantify the impact of tillage erosion on soil productivity and other soil functions in agro-landscapes. One specific problem is that the effects of erosion and topography on crop yields are confounded. There is extensive evidence that different soil quality parameters are impaired (see 8.2), especially on eroding sites. However, a framework is lacking for holistically assessing the long-term impact on such central aspects as soil fertility and productivity, SOC storage and nutrient cycling and losses in a landscape context.

Defining risk reduction targets also requires consideration of the practicalities and costs of mitigation strategies and their monitoring. Intensive tillage and mouldboard ploughing is an integral part of modern Danish agriculture. Changing cultivation systems drastically may reduce crop productivity, affect land use and will inflict costs on the economy in general. Hence, the definition of reduction targets requires a comprehensive cost-benefit analysis of sustainable tillage in a given landscape. Therefore, we cannot set a qualified reduction target for tillage erosion at present.

Policymakers generally have two options for managing risk reduction targets. The first option leaves the reduction target unspecified and seeks to reduce tillage erosion through concerted and comprehensive national information campaigns and volunteer measures backed by incentives. The second option defines a mandatory upper limit of tolerable annual tillage erosion rates on eroding sites.

8.5. Programme of measures to reach risk reduction targets

Reducing tillage erosion in Denmark calls for regulatory measures and an administrative framework for implementing and assessing mitigation strategies over a longer period of time. To identify and map risk areas, model tools have to be employed that assess tillage erosion for different tillage operations in certain cropping scenarios. Irrespective of the choice between volunteer and mandatory measures, the mitigation options listed below apply. Characteristic for these options is that they all represent adapted tillage practices and take immediate effect. Some measures are simply good agricultural practice.

- The most effective measure is to convert from conventional to reduced tillage systems; with no-till, tillage erosion is eliminated.
- Reducing tillage speed and especially tillage depth substantially reduces tillage erosivity of conventional implements. Care should be taken not to increase speed for downslope operations.
- After contour tillage, slantwise tillage, turning the soil upslope, is the least erosive.
- The frequency of tillage operations should be reduced and the loosening of soil before mouldboard ploughing avoided.
- Long slopes should not be partitioned into separate fields to avoid the formation of soil banks.

8.6. Knowledge gaps and research needs

In Denmark the following major research needs follow from the discussions above:

- Holistic and quantitative analysis of tillage erosion impact on long-term soil fertility and productivity, SOC storage and nutrient cycling and losses at the landscape scale. This is a prerequisite for defining risk reduction targets.
- Comprehensive cost-benefit analyses of sustainable tillage and cropping systems in Denmark.

- Development of a practical, interactive tool for predicting and mapping tillage erosion for different tillage scenarios at the field scale. The web-based tool would serve educational and planning purposes mainly for land-users and advisors.

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SUMMARY

The EU Commission is preparing a proposal for a Soil Framework Directive with the purpose of protecting the soil resources in Europe. The proposal identifies six major threats to the sustained quality of soils in Europe. This report addresses the threats that are considered most important under the prevailing soil and climatic conditions in Denmark: compaction, soil organic matter decline, and erosion by water and tillage. For each of these threats, the relevance and damage to soil functions as well as the geographic distribution in Denmark are outlined. We suggest a procedure for identifying areas at risk. This exercise involves an explicit identification of: i) the disturbing agent (climate / management) exerting the pressures on soil, and ii) the vulnerability of the soil to those stresses. Risk reduction targets, measures required to reach these targets, and the knowledge gaps and research needs to effectively cope with each threat are discussed.

Our evaluation of the threats is based on soil resilience to the imposed stresses. Subsoil compaction is considered a severe threat to Danish soils due to frequent traffic with heavy machinery in modern agriculture and forestry. The soil content of organic matter is critically low for a range of Danish soils, which should be counteracted by appropriate management options. Soil erosion by tillage, and to a lesser degree by water, adversely affects soil quality on much of the farmland because degradation rates are much higher than generation of soil.

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