SoilCare

SOILCARE FOR PROFITABLE AND SUSTAINABLE CROP PRODUCTION IN EUROPE

A review of soil- improving cropping systems

Editors: Oene Oenema, Marius Heinen,
Rene Rietra, Rudi Hessel

Report number: **07**
Deliverable: -

Report type: **Scientific Report**
Issue date: **01/11/2017**
Project partner: **Wageningen Environmental Research, NL**
Version: **Full Report**

www.soilcare-project.eu

WAGENINGEN UNIVERSITY & RESEARCH

SoilCare

DOCUMENT SUMMARY

Project Information	
Project Title:	Soil Care for profitable and sustainable crop production in Europe
Project Acronym:	SoilCare
Call Identifier:	H2020-SFS-2015-2b
Grant agreement no.:	677407
Starting Date:	01.03.2016
End Date:	28.02.2021
Project duration	60 months
Web-Site address:	www.soilcare-project.eu
Project coordinator:	Wageningen Environmental Research (WEnR)
EU project representative & coordinator of the project:	Dr. Rudi Hessel - (rudi.hessel@wur.nl) +31 317 486 530
Project manager(s):	Erik van den Elsen (erik.vandenelsen@wur.nl), Simone Verzandvoort (simone.verzandvoort@wur.nl), Falentijn Assinck (falentijn.assinck@wur.nl)
Report Information	
Report Title:	A review of soil- improving cropping systems
Principle Author(s):	Oene Oenema, Marius Heinen, Yang Peipei, Rene Rietra, Rudi Hessel (eds.)
Principle Author e-mail:	oene.oenema@wur.nl
Deliverable Number:	-
Work Package:	WP2
WP Leader:	Wageningen Environmental Research, the Netherlands
Nature:	PU
Dissemination:	Document
Editor (s):	Erik van den Elsen
E-Mail(s):	erik.vandenelsen@wur.nl
Telephone Number(s):	+31 317 486533
Report Due Date	31-08-2017
Report publish date:	Full Report - 01-11-2017
Copyright	©2017 the SoilCare project and Partners Copyright notice and disclaimer: http://tinyurl.com/soilcare-disclaimer

No.	Participant organisation name	Abbreviation	Country
1	Wageningen Environmental Research	WEnR	Netherlands
2	University of Newcastle upon Tyne	UNEW	United Kingdom
3	Katholieke Universiteit Leuven	KUL	Belgium
4	University of Gloucestershire	UoG	United Kingdom
5	University Hohenheim	UH	Germany
6	Research Institute for Knowledge Systems	RIKS	Netherlands
7	Technical University of Crete	TUC	Greece
8	Joint Research Centre	JRC	Italy
9	University of Bern	UNIBE	Switzerland
10	Milieu LTD	MLTD	Belgium
11	Norwegian Institute of Bioeconomy Research	NIBIO	Norway
12	Bodemkundige Dienst van België	BDB	Belgium
13	Aarhus University	AU	Denmark
14	Game & Wildlife Conservation Trust	GWCT	United Kingdom
15	Teagasc	TEAGASC	Ireland
16	Soil Cares Research	SCR	Netherlands
17	Instituto Politecnico De Coimbra	IPC/ESAC	Spain
18	National Research and Development Institute for Soil Science, Agrochemistry and Environmental Protection	ICPA	Romania
19	University of Padova	UNIPD	Italy
20	Institute of Agrophysics of the Polish Academy of	IAPAN	Poland
21	Wageningen University	WU	Netherlands
22	University of Pannonia	UP	Hungary
23	Swedish University of Agricultural Sciences	SLU	Sweden
24	Agro Intelligence Aps.	AI	Denmark
25	Crop Research Institute	VURV	Czech Republic
26	University of Almeria	UAL	Spain
27	Fédération Régionale des Agrobiologistes de Bretagne	FRAB	France
28	Scienceview Media BV	SVM	Netherlands

Review of soil-improving cropping systems (SICS)

O. Oenema, M. Heinen, R. Rietra and R. Hessel (eds.)

Contributing authors per chapter:

1 – O. Oenema and R. Hessel
2 – R. Rietra
3 – R. Rietra
4 – M. Heinen
5 – M. Heinen
6 – J. Cuevas, J.J. Hueso, F. del Moral, I. Tsanis and I. Daliakopoulos
7 – J. Stolte, R. Hessel, L. Øygarden, O. Green, A. Ferreira, G. Edwards, J. Poesen and M. Riksen
8 – A. Alaoui, A. Berti, G. Edwards, O. Green, T. Seehusen and P. Yang
9 – X. Yang, V. Silva, E. Huerta and V. Geissen
10 – M.A. Bolinder, F. Crotty, A. Elsen, M. Frac, T. Kismanyoky, T. Kätterer, J. Lipiec, M. Tits and Z. Toth
11 – F. Crotty, A. Termorshuizen, E. Hannula and E. Kandeler
12 – J. Poesen and J. Stolte
13 – G. Wyseure and H. van Helden
14 – A.J.D. Ferreira, J. Cuevas González, G. Wyseure, O. Vizitiu, V. Pinillo, Y. Canton, C. Stoate, A.K. Boulet and C.S.S. Ferreira
15 – P. Cermak, L. Mensik, H. Kusa, E. Kunzova, M. Frac, B. Toth, Z. Toth and J. Lipiec
16 – T. Dalgaard, C. Christen, G.L. Lövei, M. Ferrante, V. Bhaskar, C. Kjeldsen, A. Ferreira, O. Green, I.S. Kristensen, M. Heinen, R. Rietra, P. Yang, R. Hessel and O. Oenema
17 – O. Oenema

SoilCare
SOILCARE FOR PROFITABLE AND SUSTAINABLE CROP PRODUCTION IN EUROPE

Contents

Executive summary

Soils are vital to life on earth. Soils perform many critical functions within ecosystems and societies. Soils serve as media for growth of plants, provide habitat for animals and organisms that live in the soil, modify the atmosphere by emitting and absorbing gases and dust, absorb and purify water, process recycled nutrients, including carbon, so that plants can use them again, and serve as engineering media for construction of foundations, roadbeds, dams and buildings.

Generally, crop farmers consider soil as their main capital good that needs to be managed well. Farmers know that there are differences in productivity between soil types and between farms, which are in part related to differences in soil management and soil quality. However, soil management is complex, knowledge and labour demanding and may be costly, while effects on soil quality are often not directly visible, and mismanagement may show up only after several years. Investments in soil quality are therefore often neglected, also because the increased globalization and competition force farmers to lower costs and increase land and labour productivity.

Soil quality is commonly defined as '*the capacity of the soil to function*'. Soil quality depends on a combination of soil physical, chemical and biological characteristics. In crop production, soil quality is often defined as '*the capacity of the soil to sustain high crop yields with a minimum of external inputs and with minimal environmental impacts*'. Differences in crop yields within regions may in part be related to differences in soil quality, although differences in management and micro-climate may also contribute to spatial differences in crop yields.

Soils are under threat of physical, chemical and/or biological degradation due to the intensification, specialization and up-scaling of agricultural production. A total of 11 threats have been defined in Europe: soil acidification, salinization, erosion, compaction, contamination, desertification, flooding & water logging, landslides, loss of organic matter, loss of biodiversity and soil sealing. These threats are caused in part by agricultural activities, but in part also (enforced) by natural processes and/or by industry and citizens. Fortunately, there is also a range of activities that may contribute to the mitigation of soil threats and to an improvement of soil quality and hence to an improvement of soil functioning.

Cropping systems can be considered soil-improving if they result in an improved soil quality, i.e., in a durable increased ability of the soil to fulfil its functions, including food and biomass production, buffering and filtering capacity, and provision of other ecosystem services. Soil improving cropping systems prevent and/or mitigate soil degradation, and contribute to restoring and improving degraded soils. The term 'cropping system' refers to crop type, crop rotation, and the agronomic management techniques used on a particular field over a period

of years. The term '*soil improving cropping systems'* (SICS) is relatively new. Intuitively, the term SICS is well-understood and perceived, but the scientific underpinning as such is still lacking. Yet, there are many examples across the world showing that soils have been improved through 'cropping systems', including so-called man-made soils (e.g., plaggen soils, terra preta soils), fertilized soils, drained soils). Also, conservation agriculture, soil conservation, soil amelioration, and soil improvement are also well-established concepts.

The overall aim of the EU-funded project SoilCare is "*to assess the potential of soil-improving cropping systems and to identify and test site-specific soil-improving cropping systems that have positive impacts on profitability and sustainability in Europe*". SoilCare deals with arable land, with cropping systems that improve soil quality.

This report presents a literature review and assessment of SICS. Two approaches have been applied in the review: (i) SICS for preventing and remediating specific soil threats, and (ii) SICS that improve soil quality in general. The concept of SICS is summarized in the Figure S1 below. An extensive executive summary report has been published separately as deliverable 2.1 of SoilCare.

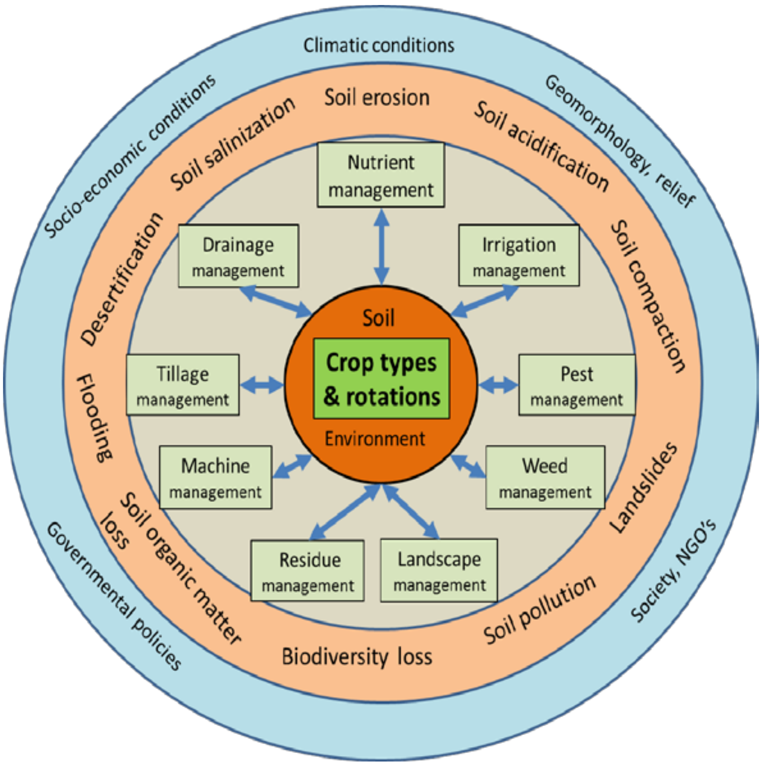


Figure S1. *Concept of Soil Improving Cropping Systems (SICS), with crop rotations and the soil environment in the centre and the nine key agro-management techniques (light-green boxes) surrounding and directly affecting soil quality and the sustainability of cropping systems. Soil threats (light-brown circle) are surrounding the SICS, while the external driving forces for the soil threats and SICS are in the outer (light-blue) circle.*

1 Introduction

O. Oenema and R. Hessel

Global crop production is facing the challenge to increase crop yields while at the same time reduce negative environmental impacts (Smil, 2000; Tilman et al., 2010). Increases in yield are especially needed in areas with rapid population growth and increasing food demand, such as in Africa and Asia. Increases in resources use efficiency and decreases in nutrients and pesticides from agricultural land to the wider environment are needed because of their detrimental effects on the environment (Rockström et al., 2009; Steffen et al., 2015), especially in developed and rapidly developing countries. Also, the quality of agricultural land is threatened by a diversity of human actions. These lead possibly to physical, chemical and/or biological degradation of the soil (Karlen et al., 1997; Cassman 1999; De Long et., 2015), which further puts pressure on the aforementioned challenge.

Crop yields in Europe are relatively high, but average wheat yields in several countries are significantly less than what is attainable (Van Ittersum et al., 2013; Boogaard et al., 2013). This holds also for other crops and is likely the result of suboptimal management, adverse wheather conditions due to climate change, and/or impairment of soil quality. Some scientists have argued that production levels in some cropping systems are only maintained by increased inputs (e.g. nutrients and pesticides) and technology, which masks losses in productivity due to reduced soil quality (Jones et al., 2012). Such increased use of agricultural inputs may reduce the profitability of crop production, due to their costs, while also negatively affecting the environment, also due to the unsustainable use of energy and resources in producing these inputs. Soil improvement is therefore necessary to break the negative spiral of degradation, increased inputs, increased costs and damage to the environment (Sørensen et al., 2014). Soil improvement makes crop production more sustainable.

The choice of a cropping system depends on markets, pedo-climatic conditions, crop rotation aspects, availability of genetic varieties, governmental subsidies and farmer's preferences. The term 'cropping system' (CS) refers to crop type, crop rotation, and the agronomic management techniques used on a particular field over a period of years (Nafziger, 2012). Cropping systems can have positive or negative effects on soil quality and the wider environment, depending on crop type, crop rotation, management, and the pedo-climatic conditions.

Crop choice, crop rotation, tillage, planting, irrigation, nutrient and pest management, and harvesting are all part of the cropping system. Choices made on these factors can influence profitability as well as sustainability of crop production systems (Nafziger, 2012; Reckling et al., 2016). Maintaining or improving soil quality and soil health is crucial for crop production, and can especially contribute to remediating subtle forms of soil degradation such as gradual loss

of organic matter and nutrients. In practice, there is often a trade-off between productivity goals and ensuring long-term continuation and provision of ecosystem services (Godfray et al., 2010). Attempts have been made in Europe to achieve soil improvement through for example soil conservation agriculture (e.g. Anken et al., 2004), but soil conservation measures are not adopted to their full potential, and are in some case even abandoned (Lahmar 2010), because conservation measures may have negative effects on crop yield (e.g., Pittelkow et al., 2014) and profitability (e.g. Meinke et al., 2001; Baudry 2014; Brandes et al., 2016).

Cropping systems can be considered soil-improving if they result in an improved soil quality, i.e., in a durable increased ability of the soil to fulfil its functions, including food and biomass production, buffering and filtering capacity, and provision of other ecosystem services. Soil improving cropping systems prevent and/or mitigate soil degradation, and contribute to restoring and improving degraded soils. The term '*soil improving cropping systems*' (SICS) is relatively new. Intuitively, the term SICS is well-understood and perceived, but the scientific underpinning is still lacking, and the scientific, practical and policy relevance not demonstrated. Likely, cropping systems are soil improving under specific soil and climate conditions only.

The overall aim of the EU-funded project SOILCARE is "*to assess the potential of soil-improving cropping systems and to identify and test site-specific soil-improving cropping systems that have positive impacts on profitability and sustainability in Europe*". The project involves 28 organisations and 16 study sites across Europe, and will run from March 2016 to March 2021. The work is organized in 8 work packages (http://www.soilcare-project.eu/).

The current report deals with work package 2 of SOILCAIRE: "Review of soil improving cropping systems (SICS)". Specific objectives of work package 2 are:

1. To review SICS and their key driving forces, in Europe;
2. To analyse the strong and weak points of SICS, using agronomic, environmental, and social-economic criteria;
3. To develop and test a framework for classifying SICS;
4. To derive threshold values for soil quality, and to identify the need for SICS, as function of pedo-climatic zones in Europe; and
5. To develop and test a decision tool to be used for the pre-selection of key SICS.

This report brings together data and information about 'cropping systems that may improve soil quality'. It is based on literature review and assessments. It is the first comprehensive review and assessment of SICS. The focus is on cropping systems that may improve soil quality in the Member States of the European Union (EU-28), but many of the findings in this report have a much broader relevance and applicability than just EU-28.

The term SICS is new, and a search for the term SICS in literature gives no 'hits', apart from the publications of He et al (2012) and Reckling et al (2016), which mentioned the terms for specific potato and rice systems, respectively. A review and assessment of literature on SICS is,

therefore, indirect. It involves examination of cropping systems that change soil threats, properties, functions in a positive manner.

The focus of SOILCARE and this review are on arable land. Originally, arable land was defined as 'growing crops in ploughed soil', in contrast with permanent grassland and orchards. This definition does not fit well for arable farms with zero tillage (which some would consider a SICS). Yet, we will use the terms arable land and arable cropping in this report for growing crops in both ploughed soil and minimally disturbed soil. Crops include arable crops, vegetables, temporary grass for seed, hay, silage or green manure production, as well as perennial feed crops like alfalfa, which may be grown for 4 or more years. The focus on arable land is related to the fact that the risk of soil degradation is greater in arable land than in grasslands and orchards. Also, there is a large variety of crop types, crop rotations, machines and management, which provide opportunities for optimization and thereby improving soil quality.

Before reviewing the state of the art of soil improving cropping systems, we provide in Chapter 2 a brief introduction of current farming systems and cropping systems in EU-28, based on statistical data.

Chapter 3 introduces the driving forces of cropping systems; cropping systems continuously develop in response to changes in market conditions and developments in science and technology, policy, education and farmers' preferences.

Chapter 4 provides an overview of soil quality and soil threats in EU-28, and how they relate to spatial variations in climate, geomorphology and soil types.

Chapter 5 introduces the analytical framework for analysing and classifying SICS, as developed for the purpose of this study.

Chapters 6 to 15 provide comprehensive reviews and assessments of cropping systems that improve soil quality and/or prevent and mitigate soil threats. Each of these 10 chapters deals with cropping systems for a specific soil threat, i.e., cropping systems that prevent and mitigate a specific soil threat. The literature data and information greatly differs between soil threats, and therefore the the length of the chapters too.

Chapter 16 deals with an overview of main soil improving cropping systems, focused on preventing and mitigation of various forms of subtle and gradual soil degradation, and/or improving soil quality.

Chapter 17 deals with the selection of most promising SICS, to be discussed by stakeholders and tested further at the 16 study sites across EU-28.

A glossary of terms was prepared as part of the review of SICS; this glossary can be found at the website of SOILCARE (http://www.soilcare-project.eu/).

1.1 References

Anken T, P Weisskopf, U Zihlmann, H Forrer, J Jansa, K Perhacova. 2004. Long-term tillage system effects under moist cool conditions in Switzerland. Soil and Tillage Research, 78, 171-183.

Baudry, J. 2014. Optimising profitability of crop production through Ecological Focus Areas. EIP-AGRI Focus Group. Discussion paper. European Commission, 18 pp.

Boogaard, H. ; Wolf, J. ; Supit, I. ; Niemeyer, S. ; Ittersum, M.K. van 2013. A regional implementation of WOFOST for calculating yield gaps of winter wheat across the European Union. Field Crops Research 143, 130 – 142.

Brandes, E., GS McNunn, LA Schulte, IJ Bonner, DJ Muth, BA Babcock, B Sharma and EA Heaton. 2016. Subfield profitability analysis reveals an economic case for cropland diversification Environ. Res. Lett. 11 (2016) 014009.

Cassman KG. 1999 Ecological intensification of cereal production systems: Yield potential, soil quality, and precision agriculture. Proc Natl Acad Sci 96: 5952–5959.

DeLong, C., R. Cruse, and J. Wiener. 2015. The Soil Degradation Paradox: Compromising Our Resources When We Need Them the Most. *Sustainability*, *7*, 866-879; doi:10.3390/su7010866

Godfray, H. C. J. *et al.* 2010. Food security: the challenge of feeding 9 billion people. *Science* 327, 812–818.

He Z, R Larkin, and W. Honeycutt (eds.) 2012. Sustainable Potato Production: Global Case Studies. Springer.

Jones A, P Panagos, S Barcelo, F Bouraoui, C Bosco, O Dewitte, C Gardi, M Erhard, J Hervás, R Hiederer et al. (2012) The State of Soil in Europe: A Contribution from JRC to the European Environmental Agency's Environment State and Outlook Report—SOER 2010; Publications Office: Luxembourg.

Karlen D L, MJ Mausbach, JW Doran, RG Cline, RF Harris and GE Schuman. 1997. Soil Quality: A Concept, Definition, and Framework for Evaluation. Soil Sci. Soc. Am. J. 61:4-10.

Lahmar R. 2010. Adoption of conservation agriculture in Europe. Lessons of the KASSA project. Land Use Policy 27, 4-10

Meinke, H., W.E. Baethgen, P.S. Carberry, M. Donatelli, G.L. Hammer, R. Selvaraju, C.O. Stockle. 2001. Increasing profits and reducing risks in crop production using participatory systems simulation approaches. Agricultural Systems 70, 493–513.

Nafziger. 2012. Cropping Systems. In: Illinois Agronomy Handbook, pp. 49-63. Available at: http://extension.cropsciences.illinois.edu/handbook/pdfs/chapter05.pdf (last access: 13 Dec-2016).

Pittelkow CM, X Liang, BA Linquist, KJ van Groenigen, J Lee, ME Lundy, N van Gestel, J Six, RT Venterea, C van Kessel. 2014. Productivity limits and potentials of the principles of conservation agriculture. Nature. Doi:10.1038/nature13809

Reckling, M, JM Hecker, G Bergkvist, CA Watson, P Zander, N Schläfke, FL Stoddard, V Eory, CFE Topp, J Maire, and J Bachinger (2016). A cropping system assessment framework - Evaluating effects of introducing legumes into crop rotations. Eur. J. Agronomy 76: 186–197.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin III, F. S., Lambin, E., Lenton, T., Scheffer, M., Folke, C. & Schellnhuber, H. J. 2009. Planetary boundaries: exploring the safe operating space for humanity. *Ecology and society,* 14.

Smil, V. 2000. Feeding the World: A Challenge for the 21st Century, The MIT Press, Cambridge.

Sørensen CG, N Halberg, FW Oudshoorn, BM Petersen, R Dalgaard. 2014. Energy inputs and GHG emissions of tillage systems. Biosystems Engineering, 120, 2-14.

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., Biggs, R., Carpenter, S. R., de Vries, W. & de Wit, C. A. 2015. Planetary boundaries: Guiding human development on a changing planet. *Science,* 347, 1259855.

Stolte J, M Tesfai, L Øygarden, S Kværnø, J Keizer, F Verheijen, P Panagos, C Ballabio, R Hessel (Eds). 2016. Soil threats in Europe; EUR 27607 EN; doi:10.2788/488054 (print); doi:10.2788/828742 (online).

Tilman, D., Balzer, C., Hill, J. & Befort, B. L. 2011. Global food demand and the sustainable intensification of agriculture. Proc. Natl Acad. Sci. USA 108, 20260–20264.

Tóth G, L Montanarella and E Rusco (eds.). 2008. Threats to Soil Quality in Europe. JRC Scientific and Technical Reports (JRC 46574 / EUR 23438. DOI 10.2788/8647

Van Ittersum MK, R Rabbinge. 1997. Concepts in production ecology for analysis and quantification of agricultural input-output combinations. Field Crops Research 52, 197-208.

Van Ittersum, M.K., Cassman, K.G., Grassini, P., Wolf, J., Tittonell, P., Hochman, Z. 2013. Yield gap analysis with local to global relevance a review. Field Crops Research 143, 4-17.

2 Overview of farming and cropping systems in Europe

R. Rietra

This chapter provides an overview of current farming systems and cropping systems in EU-28 in 2013. It offers also a basis for upscaling results of for example study sites, and helps in selecting SICS to be explored further in WP3 and WP6 of SOILCARE.

2.1 Farming systems

European scale data about farming systems are collected by Eurostat through the Farm Structure Survey (FSS) and presented as "agricultural production systems. The classification of farm types is detailed with 62 particular types, which are aggregated to 22 principal types and 9 general types (EC, 2008).

In 2013, 29% of EU-28 farms were specialised in field crops and 18% in permanent crops (Table 2.1). Specialist grazing, pig, poultry and mixed crop-livestock holdings account for 45% of the holdings. While specialist field crops and specialist grazing livestock together account for 46% of the holdings, they account for 74% of the land.

Table 2.1. *Agricultural holdings by farm type in the EU-28 in 2013.*

#	General farm type	% of holdings	% of land
1	Specialist field crops	29	42
2	Specialist horticulture	2	1
3	Specialist permanent crops	18	6
4	Specialist grazing livestock	17	32
5	Specialist granivores (pig, poultry etc.)	9	2
6	Mixed cropping	5	2
7	Mixed livestock	4	2
8	Mixed crop-livestock	14	11
9	Non-classifiable holdings	2	1

The economic size varies between farm types. Specific data about the economic performance of farms is collected through the Farm Accountancy Network (FADN) (EU-FADN, 2016). Anderson et al (2016) developed a farm typology for EU agriculture using FFS and FADN data, on the basis of:

- Specialisation: Measured as the output value from the main activity; 10 farm specialization types.

- Size: Measured as the economic size of the farms; 3 classes: <16; 16-40; >40 ESU[1]
- Intensity: Measured as the total output in Euro per ha; 3 classes: <500; 500-3000; >3000 euro/ha
- Land use: Measured as the proportion of the agricultural area covered by specific types of crops; 9 different land use types were distinguished.

In total 189 farm types were identified; the aggregate of 3 size types, 3 intensity types and 21 combined specialisation/land use types (Table 2.2). The farm typology presented in Table 2.2 is a useful framework for characterizing farm types, as farm size, intensity, specialization and land use are important determinants for the pressure on soil quality (Chapters 6-16).

Table 2.2. *Share of the farms, area, livestock units (LU) and output covered by the different size types, intensity types and specialization/land use types (Anderson et al., 2006).*

	Share of farms %	Share of area %	Share of LU %	Share of output %
Small scale	49.5	15.2	6.3	10.7
Medium scale	24.4	24.3	16.6	16.7
Large scale	26.0	60.5	77.1	72.7
Low intensity	11.8	23.6	6.2	2.9
Medium intensity	53.4	61.6	39.0	37.9
High intensity	34.8	14.8	54.9	59.2
Arable/Cereal	12.4	18.6	2.2	9.4
Arable/Fallow	4.4	8.8	0.5	2.5
Arable/Others	6.3	6.1	1.1	4.8
Arable/Specialised crops	5.0	3.7	0.6	4.4
Beef and mixed cattle/Land independent	0.3	0.1	0.9	0.5
Beef and mixed cattle/Others	1.8	1.6	2.2	1.0
Beef and mixed cattle/Permanent grass	3.8	6.5	5.4	1.6
Beef and mixed cattle/Temporary grass	0.9	1.6	1.7	0.7
Dairy cattle/Land independent	0.4	0.2	1.4	1.3
Dairy cattle/Others	6.9	7.6	10.8	10.2
Dairy cattle/Permanent grass	5.7	7.5	10.1	7.7
Dairy cattle/Temporary grass	2.2	2.8	3.4	3.3
Horticulture	27.2	7.7	0.4	14.1
Mixed farms	6.6	11.0	11.8	8.5
Mixed livestock	2.3	2.7	8.9	4.2
Permanent crops	3.7	0.5	0.0	8.9
Pigs/Land independent	0.9	0.6	10.8	4.4
Pigs/Others	1.2	1.3	10.7	4.7
Poultry and mixed pigs/poultry	1.0	0.5	9.0	3.7
Sheep and goats/Land independent	1.1	0.1	1.1	0.6
Sheep and goats/Others	6.0	10.4	7.2	3.5

[1] ESU is European Size Units (ESU), where 1 ESU corresponds to 1,200 Euro. It refers to the value of output from the farm less the cost of variable inputs required to produce that output, based on 3 years averages.

2.2 Crop rotations

Crop rotations are important for the sustainability of agricultural systems (Mudgal *et al.*, 2010). However, empirical data are scarce about crop rotations, because there is little or no monitoring of crop rotations in EU countries (Schönhart *et al.*, 2011; Lorenz *et al.*, 2013). Main crop rotations are six, four, three, two and one year crop rotations. Typical four year crop rotations in Western Europe may consist of "winter wheat-sugar beet-winter wheat-potato", or "winter wheat-silage maize-winter wheat-sugar beet". A typical three year rotation may consist of "winter wheat-winter barley-sugar beet/silage maize" or "winter wheat-winter wheat-sugar beet". A typical two year rotation may consist of "winter wheat-silage maize/sugar beet" (Leteinturier *et al.*, 2006). One-year rotations are also called monocultures.

On the basis of series of aerial photos taken during the period 1992 to 2003, crop sequence patterns were analysed for France (Xiao *et al.*, 2014). The major crop sequences in France are three-year crop rotation of "wheat-barley-rapeseed", a two-year crop rotations "maize-wheat", "rapeseed-wheat", monocultures of maize, wheat and barley, long-term fallow, temporary pasture, and sequences of different two-year rotations with the three year rotation or one year wheat.

Intensive crop rotations have a relatively high share of root crops (potatoes, sugar beet, carrots, onions, flower bulbs) and/or a high share of vegetables (often 2 or more crops per season). Such intensive crop rotations often require intensive soil cultivation (ploughing, crumbling), relatively high inputs of nutrients, pesticides, and irrigation, and often heavy machinery for harvesting (combine harvesters), which in part happens late in autumn. Conversely, extensive cropping systems have a relatively high share of cereals and perennials, and have relatively little soil cultivation. Permanent cropping systems (fruit orchards, olive yards) commonly have only some soil cultivation near the trees, to remove weeds.

The crop statistics of Eurostat distinguishes 17 categories for cereals and 29 for other main crops, 40 categories for vegetables, 41 for permanent crops. Within each crop large differences can exist. Cereals can be managed intensively, such as in northern France, Germany and United Kingdom, but can also be important for nature conservation such as in parts of Spain.

2.3 Agronomic management techniques

A one-off survey was the Survey on Agricultural Production Methods (SAPM). The legal basis for both the Farm Structure Survey (FSS) and the Survey on Agricultural Production Methods (SAPM) is Regulation 1166/2008 (EU, 2008). The data collected in SAPM include: tillage methods, soil conservation, landscape features, animal grazing, animal housing, manure application, manure storage, manure management and treatment facilities and irrigation. Tillage practices were derived from the share of arable areas under conventional, conservation

and zero tillage (Table 2.3). Both databases, FSS and SAPM, have been linked (EUROSTAT, 2016f) and can be accessed via the online publication on agro-environmental indicators.

Conservation and zero tillage are used on 9 to 29% of the farms (Table 2.3). Eurostat identifies only three types of tillage practices (conventional, low and zero tillage), which masks subtle but often important differences. For example, a recent study in England identified a wide range of tillage practices (Townsend *et al.*, 2015), and a large percentage of farms practiced a variety of reduced tillage practices, which differ depending on crop type (Figure 2.1). The benefits of the three ploughing frequencies were analysed on the basis of changes in crop yield and in the costs of seed, fertiliser and pesticides.

Table 2.3. *Share of arable land under soil conservation tillage EU27 in 2010 (%) (Eurostat, 2016c). Tillage and type of tillage in % of farm type.*

	Principal farm type	**Total, %**	**Tillage practices, %**		
				Conservation or zero tillage	**Conventional**
1	Specialist cereals, oilseed and protein crops	36.8		29	67
2	General field cropping	14.6		23	64
3	Specialist horticulture - indoor	0.2		6	36
4	Specialist horticulture - outdoor	0.5		12	73
5	Other horticulture	0.1		14	64
6	Specialist vineyards	0.8		20	55
7	Specialist fruit and citrus fruit	0.5		18	58
8	Specialist olives	0.4		18	49
9	Various permanent crops combined	0.3		18	63
10	Specialist dairying	10.6		13	59
11	Specialist cattle-rearing and fattening	4.0		9	42
12	Cattle-dairying, rearing and fattening combined	1.5		12	59
13	Sheep, goats and other grazing livestock	3.1		12	49
14	Specialists pigs	2.9		21	74
15	Specialist poultry	0.8		23	61
16	Various granivores combined	0.3		14	74
17	Mixed cropping	3.4		21	70
18	Mixed livestock, mainly grazing livestock	2.3		14	72
19	Mixed livestock, mainly granivores	1.3		16	71
20	Field crops-grazing livestock combined	9.1		24	65
21	Various crops and livestock combined	5.6		19	73
22	Non-classifiable holdings	0.6		24	30
	Total EU-27	**100**		**22**	**64**

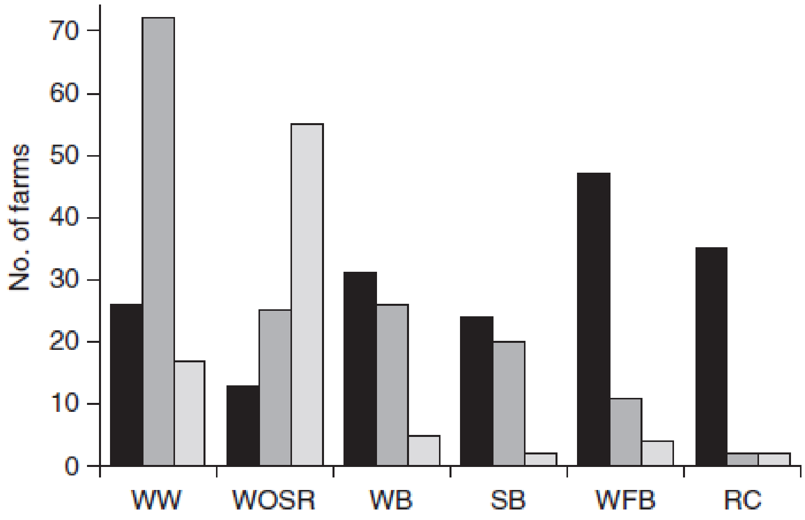


Figure 2.1. *Frequency of ploughing after different crops. Always plough (black bars), sometimes plough (dark grey bars), never plough (light grey bars). WW (winter wheat); WOSR (winter oilseed rape); WB (winter barley); SB (spring barley); WFB (winter field beans and peas); RC (root crops) (Townsend et al., 2015).*

Soil cover refers to the fraction of the land covered by crops during the winter season. Soil cover is important for preventing loss of nutrients and pesticides by runoff, and reduce the risk of soil erosion. Currently it is estimated that 25% of the arable land in EU-28 is not covered during the winter season (Table 2.4).

Table 2.4. *Arable land with soil cover EU28 in 2010*

Soil cover	%
Normal winter crop	44
Cover or intermediate crop	5
Plant residues	9
Bare soil	25
Not recorded	16

2.4 Manure management

The data collected in SAPM also include: manure application, manure storage, manure management and treatment facilities (EU, 2008). The Manure Management Inventory was carried out by Eurostat in 2012. Also the obligatory country reports on the Nitrate Directive are used as the basis for a 4-yearly report by the EU commisions on the implementation of the Directive (EC, 2013). Data are derived from different sources and are therefore not harmonised across countries (Eurostat, 2013b).

Manure manegement is relevant for the emissions of ammonia, but also for nitrous oxide and methane (GHG) and for the leaching and run off of nutrients to groundwater and surface water and consequently eutrophication and biodiversity (loss). The current uptake of Best Avaliable Technology (BAT) by European farmers is only patchy (Loyon *et al.*, 2016).

The number of livestock farms with manure storages are presented in Table 2.5. In some EU countries all holdings with liquid manure storage facilities have to use a cover to minimize NH_3 emissions (Belgium, Germany, the Netherlands, Slovakia). In Romania, Bulgaria and Cyrus 28%, 27% and 15% of the holdings with storage for liquid manure are covered, respectively.

Manure treatment has become relevant in some countries in Europe. It is estimated that 7.8% of the manure was treated in 2010-2012. The most important treatments are separation and anaerobic digestion. In total 3.1% of the total livestock manure was separated in liquid and solid fractions, and 6.4% of the livestock manure in EU was anaerobically digested for producing biogas. In Germany, 29% of the livestock manure was digested anaerobically. Separation was mostly used in Italy; 24% of the livestock manure was separated. (Foged *et al.*, 2012). The treatments of livestock manure can have effects on agricultural production, GHG emissions and also soil quality (Möller & Müller, 2012).

Table 2.5. *Holdings with manure storage facilities in EU-27 in 2003*

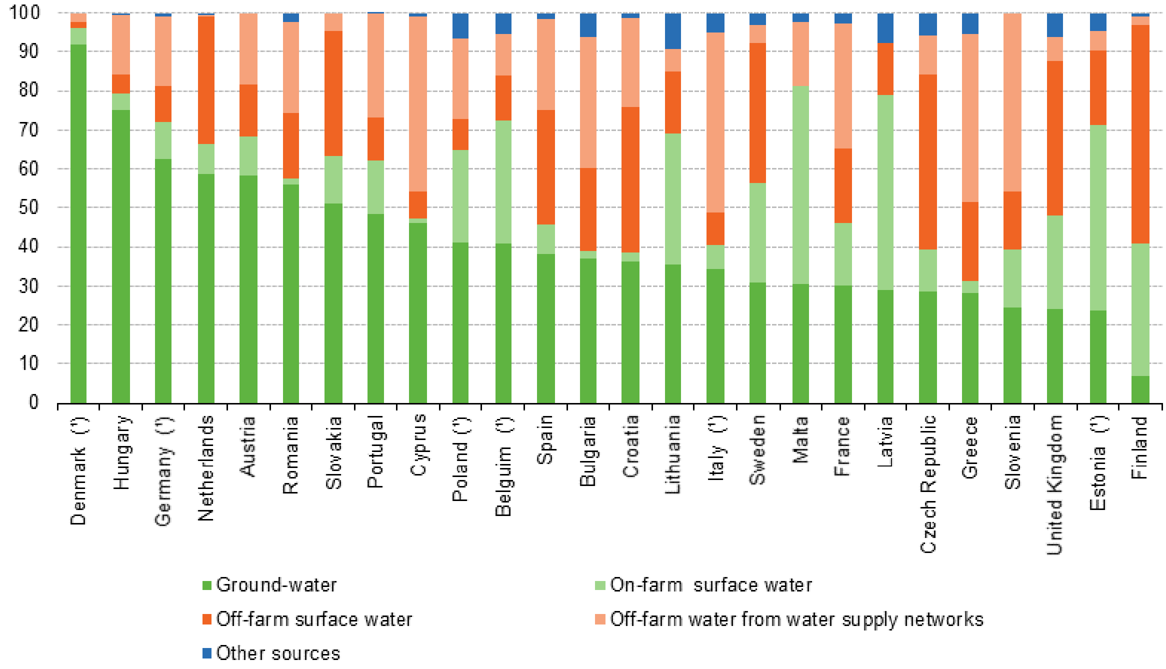
Storage facility	number	%
for solid manure	1591 830	80%
for liquid manure	959 290	49%
for slurry	610 910	31%
Total number of holdings with storage facitlities	1 977 530	

2.5 Irrigation

Irrigation is relevant for increasing crop production in drought-prone areas. In special cases, irrigation may be practiced to enhance crop quality. The amount and source of irrigation water can have positive and negative effects on salinization, erosion, soil structure (García-Ruiz, 2010), transfer of pathogens (Pachepsky *et al.*, 2011) and on the emission of GHG and nitrogen (Eagle & Olander, 2012). Data on irrigation have been collected in the Survey on Agricultureal Production Methods (SAPM) in Europe SAPM.

In total, 15.6% of the agricultural land in Portugal was irrigated in 2003. Interestingly also 16.8% of the agricultural land in Denmark was irrigated. However the volume of water used for irrigation in Portugal was more then 10 times higher than in Denmark (7371 versus 685 m^3 per ha) (Eurostat, 2016a), because in Denmark irrigation is only used to bridge dry periods during the growing season. The Eurostat data also show that groundwater seems to be an important source of irrigation water in almost all Member States (Figure 2.2) (Eurostat, 2016a). This kind

of information might be combined with quality data of surfacewater or groundwater in relation to saliniation or risks for of transfer of pathogens.



Note: Ireland:data considered not existing or non-significant; Luxembourg: data not available.
(¹) Only main water source for irrigation used on farms was reported.

Figure 2.2. *Water source use for irrigation (Eurostat, 2016a).*

2.6 Animal grazing and housing

The data collected in the Survey on Agricultural Production Methods (SAPM) also includes the number and density of livestock on agricultural land. The use of livestock manure is relevant for Soil-Improving Cropping Systems (SICS) such as soil organic matter (Söderström *et al.*, 2014). The Eurostats data have been used to describe the use of livestock manure across the EU, and agricultural nitrogen and phosphorous balances: surplus or deficits (EEA, 2015), and also for soil organic matter (Gobin *et al.*, 2011).

Emissions of methane and ammonia depend on the type of animal housings and grazing systems (Bellarby *et al.*, 2013). Various grazing systems exist in the EU but data are often based on estimation (van den Pol-van Dasselaar *et al.*, 2015). The percentage of full grazing dairy cows, versus very limited grazing, varies between 20% and 25% resp. in Poland and Denmark and more than 90% in France, Ireland and Sweden. One inventory is available of the type of animal housings (Eurostat, 2013a). Animal housing is relevant for the calculation of the ammonia emission from animal housings (Amann *et al.*, 2012).

It is estimated on world scale that livestock accounts for half of the ammonia and methane emission technical mitigation potential of the agriculture and other land-use sectors. An important tool to do so is turning degraded lands into grassland (Herrero *et al.*, 2016). Inventarisations of land degradation and erosion are avaliable at EU level from the Joint Research Centre (Panagos *et al.*, 2015).

2.7 Landscape features

Current Rural Development payments are used to encourage farmers to apply agricultural methods that comply with improvement of the landscape. The data collected in SAPM also include: state and diversity of landscape. This indicator can provide information about changes in the landscape (EU, 2012). Biodiversity is relevant for agriculture production by crop protection and pollination (Rands & Whitney, 2011; Tschumi *et al.*, 2015; Jeanneret *et al.*, 2016). The use of corridors, or landscape features can enhance biodiversity (Grashof-Bokdam & van Langevelde, 2005, Opdam, 2013).

On the other hand, management of agricultural land is also important for the biodiversity in EU. Inventories of biodiversity are available (Petit *et al.*, 2001; Maiorano *et al.*, 2015).

Diversity is related to the land-use intensity in Europe (Kleijn *et al.*, 2009). A description of the biodiversity of soils, also in various cropping systems is given in the *Atlas of Soil Biodiversity* (Artz *et al.*, 2010). In terms of SICS it is found that soil biodiversity is often related to the agricultural intensity (Tsiafouli *et al.*, 2015). The relevance of soil biodiversity for crop production is discussed in chapter 11.

2.8 References

Amann, M., Klimont, Z. & Winiwarter, W. 2012. Emissions from agriculture and their control potentials.

Artz, R., Anastasoiu, D., Arrouays, D., Bastos, A., Benedetti, A. & Pagliai, M. 2010. *European atlas of soil biodiversity,* Publication Office of the European Union.

Bellarby, J., Tirado, R., Leip, A., Weiss, F., Lesschen, J. P. & Smith, P. 2013. Livestock greenhouse gas emissions and mitigation potential in Europe. *Global change biology,* 19, 3-18.

Eagle, A. J. & Olander, L. P. 2012. Greenhouse gas mitigation with agricultural land management activities in the United States-a side-by-side comparison of biophysical potential. In: *Advances in Agronomy.* pp. 79-179.

EC 2008. Comiision Regulation (EC) No 1242/2008 of 8 December 2008 establishing a Community typology for agricultural holdings. *Official Journal of the European Union,* L 335, 3-24.

EC 2013. *Report from the Commision to the Council and the European Parlement, on the implementation of Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources based on Member State reports for the period 2008–2011 COM(2013) 683 final. {SWD(2013) 405 final}* Brussels.

EEA 2015. Agriculture: nitrogen balance https://www.eea.europa.eu/data-and-maps/indicators/agriculture-nitrogen-balance/agriculture-nitrogen-balance-assessment-published. In. (ed E. E. A. (EEA).

EU-FADN 2016. webiste: http://ec.europa.eu/agriculture/rica/ (last accessed june 2, 2016).

EU 2008. *Regulation (EC) No. 1166/2008 of the European Parliament and of the Council on farm structure surveys and the survey on agricultural production methods and repealing Council Regulation (EEC) No. 571/88.*

EU 2012. Agri-environmental indicator - landscape state and diversity http://ec.europa.eu/eurostat/statistics-explained/index.php/Agri-environmental_indicator_-_landscape_state_and_diversity. In.

Eurostat 2013a. Agri-environmental indicator - animal housing http://ec.europa.eu/eurostat/statistics-explained/index.php?title=Archive:Agri-environmental_indicator_-_animal_housing&oldid=153117#Animal_housing. In.

Eurostat 2013b. Agri-environmental indicator - manure storage http://ec.europa.eu/eurostat/statistics-explained/index.php/Agri-environmental_indicator_-_manure_storage. In.

Eurostat 2015. *Eurostat Handbook for Annual Crop Statistics (Regulation 543/2009 & Gentlemen's/ESS agreements) (Revision 2015 – Presented in the WPM of the 22-23 October 2014, finalised in May 2015)*.

Eurostat 2016a. Agri-environmental indicator - irrigation. http://ec.europa.eu/eurostat/statistics-explained/index.php/Agri-environmental_indicator_-_irrigation. Last accessed June 3, 2016).

Eurostat 2016b. *Agri-environmental indicator -tillage practices. http://ec.europa.eu/eurostat/statistics-explained/index.php/Agri-environmental_indicator_-_tillage_practices (last accessed on june 3, 2016).*

EUROSTAT 2016c. *Survey on agricultural production methods http://ec.europa.eu/eurostat/statistics-explained/index.php/Survey_on_agricultural_production_methods (last accessed on June 2 2016).*

Foged, H., Flotats Ripoll, X., Bonmatí Blasi, A., Palatsi Civit, J., Magrí Aloy, A. & Schelde, K. M. 2012. *Inventory of manure processing activities in Europe.Technical Report No. I concerning "Manure Processing Activities in Europe" to the European Commission, Directorate-General Environment. 138 pp*.

García-Ruiz, J. M. 2010. The effects of land uses on soil erosion in Spain: A review. *Catena,* 81, 1-11.

Gobin, A., Campling, P., Janssen, L., Desmet, N., van Delden, H., Hurkens, J., Lavelle, P. & Berman, S. 2011. Soil organic matter management across the EU—best practices, constraints and trade-offs. *Final report for the European Commission's DG Environment*.

Grashof-Bokdam, C. J. & van Langevelde, F. 2005. Green veining: landscape determinants of biodiversity in European agricultural landscapes. *Landscape Ecology,* 20, 417-439.

Herrero, M., Henderson, B., Havlík, P., Thornton, P. K., Conant, R. T., Smith, P., Wirsenius, S., Hristov, A. N., Gerber, P. & Gill, M. 2016. Greenhouse gas mitigation potentials in the livestock sector. *Nature Climate Change,* 6, 452-461.

Jeanneret, P., Begg, G., Gosme, M., Alomar, O., Reubens, B., Baudry, J., Guerin, O., Flamm, C. & Wäckers, F. 2016. Landscape features to improve pest control in agriculture. *Solutions,* 7, 48-57.

Kleijn, D., Kohler, F., Báldi, A., Batáry, P., Concepción, E., Clough, Y., Díaz, M., Gabriel, D., Holzschuh, A. & Knop, E. 2009. On the relationship between farmland biodiversity and land-use intensity in Europe. *Proceedings of the Royal Society of London B: Biological Sciences,* 276, 903-909.

Leteinturier, B., Herman, J. L., Longueville, F. d., Quintin, L. & Oger, R. 2006. Adaptation of a crop sequence indicator based on a land parcel management system. *Agriculture, Ecosystems & Environment,* 112, 324-334.

Lorenz, M., Fürst, C. & Thiel, E. 2013. A methodological approach for deriving regional crop rotations as basis for the assessment of the impact of agricultural strategies using soil erosion as example. *Journal of Environmental Management,* 127, S37-S47.

Loyon, L., Burton, C. H., Misselbrook, T., Webb, J., Philippe, F. X., Aguilar, M., Doreau, M., Hassouna, M., Veldkamp, T., Dourmad, J. Y., Bonmati, A., Grimm, E. & Sommer, S. G. 2016. Best available technology for European livestock farms: Availability, effectiveness and uptake. *Journal of Environmental Management,* 166, 1-11.

Maiorano, L., Amori, G., Montemaggiori, A., Rondinini, C., Santini, L., Saura, S. & Boitani, L. 2015. On how much biodiversity is covered in Europe by national protected areas and by the Natura 2000 network: insights from terrestrial vertebrates. *Conservation Biology,* 29, 986-995.

Möller, K. & Müller, T. 2012. Effects of anaerobic digestion on digestate nutrient availability and crop growth: A review. *Engineering in Life Sciences,* 12, 242-257.

Mudgal, S., Lavelle, P., Cachia, F., Somogyi, D., Majewski, E., Fontaine, L., Bechini, L. & Debaeke, P. 2010. Environmental impacts of different crop rotations in the European Union. *European Commission, Brussels*.

Pachepsky, Y., Shelton, D. R., McLain, J. E. T., Patel, J. & Mandrell, R. E. 2011. Irrigation Waters as a Source of Pathogenic Microorganisms in Produce. A Review. In: *Advances in Agronomy.* pp. 73-138.

Panagos, P., Borrelli, P., Poesen, J., Ballabio, C., Lugato, E., Meusburger, K., Montanarella, L. & Alewell, C. 2015. The new assessment of soil loss by water erosion in Europe. *Environmental Science & Policy,* 54, 438-447.

Petit, S., Firbank, L., Wyatt, B. & Howard, D. 2001. MIRABEL: models for integrated review and assessment of biodiversity in European landscapes. *AMBIO: A Journal of the Human Environment,* 30, 81-88.

Rands, S. A. & Whitney, H. M. 2011. Field margins, foraging distances and their impacts on nesting pollinator success. *PLoS ONE,* 6, e25971.

Schönhart, M., Schmid, E. & Schneider, U. A. 2011. CropRota – A crop rotation model to support integrated land use assessments. *European Journal of Agronomy,* 34, 263-277.

Söderström, B., Hedlund, K., Jackson, L. E., Kätterer, T., Lugato, E., Thomsen, I. K. & Bracht Jørgensen, H. 2014. What are the effects of agricultural management on soil organic carbon (SOC) stocks? *Environmental Evidence,* 3, 2.

Townsend, T., Ramsden, S. & Wilson, P. 2016. How do we cultivate in England? Tillage practices in crop production systems. *Soil Use and Management,* 32, 106-117.

Tschumi, M., Albrecht, M., Entling, M. H. & Jacot, K. 2015. High effectiveness of tailored flower strips in reducing pests and crop plant damage. In: *Proc. R. Soc. B.* The Royal Society, pp. 20151369.

Tsiafouli, M. A., Thébault, E., Sgardelis, S. P., Ruiter, P. C., Putten, W. H., Birkhofer, K., Hemerik, L., Vries, F. T., Bardgett, R. D. & Brady, M. V. 2015. Intensive agriculture reduces soil biodiversity across Europe. *Global change biology,* 21, 973-985.

van den Pol-van Dasselaar, A., Aarts, H. F. M., De Vliegher, A., Elgersma, A., Reheul, D., Reijneveld, J. A., Verloop, J., Hopkins, A. & European Grassland, F. 2015. *Grassland and forages in high output dairy farming systems : proceedings of the 18th Symposium of the European Grassland Federation Wageningen, the Netherlands 15-17 June 2015,* Wageningen Academic Publishers, Wageningen.

Xiao, Y., Mignolet, C., Mari, J.-F. & Benoît, M. 2014. Modeling the spatial distribution of crop sequences at a large regional scale using land-cover survey data: A case from France. *Computers and Electronics in Agriculture,* 102, 51-63.

3 Overview of driving forces of farm and cropping systems

R. Rietra

Farming systems and cropping systems change over time, due to the influence of external and internal driving forces. Common trends during the past 50 years have been specialization, intensification and up-scaling. Specialization generally refers to a decrease in crop types within cropping systems or to the break-up of mixed systems into separate cropping systems and animal production systems. Intensification refers to the increased output per unit of labour, or the increased output per unit of agricultural land.

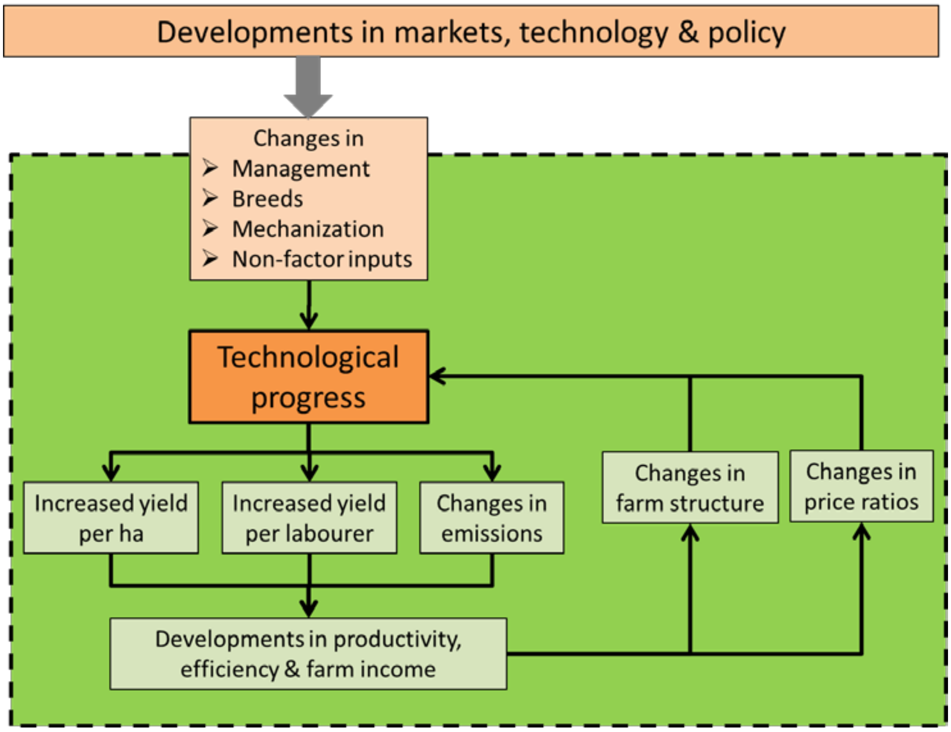


Figure 3.1. *Concept of intensification of agricultural land use. External driving forces are on top. Arrows represent influences and/or incentives; boxes represent processes or results (after Oenema et al., 2014).*

Intensification is a result of technological progress, which is fuelled by developments in markets, technology, and/or policy (Figure 3.1). These developments provide tools for technological progress, including improvements in knowledge, management, mechanization and crop varieties/breeds. Commonly, there is also a change in inputs like fertilizers, herbicides, pesticides, contractor assistance, etc. Technological progress leads to changes in the utilization

of land and labour, which subsequently leads to higher yields per ha and per unit labour, but also to changes in various emissions. The resulting changes in productivity, efficiency and farm income may subsequently lead to changes in farm structure and in the price ratio of outputs and inputs, which may provide new impulses to intensification. Hence, intensification of agricultural productivity involves a chain of processes.

In their 'History of World Agriculture', Mazoyer and Roudart (2006) argue that in a globalizing world (i) modern farms in the western world compete on the world market with small subsistence farms elsewhere, (ii) the productivity per ha and per unit labour increases due to technical progress, but much more in the western world than in the developing world, (iii) prices for agricultural commodities decrease due to technical progress and increased competition, (iv) cost of living increase due to higher standards and inflation, and (v) farmers with low productivity drop out, while on the other side of the spectrum, new, higher productive farms develop further. These lines of thoughts are visualized in Figure 3.2; it basically conveys the message that intensification, up-scaling and increasing labour productivity is the only way to stay in production in a globalizing world. Of course, this is a too simple statement, as there is also a third axis not shown in Figure 3.2, the axis of creating 'added value' and additional income sources. Production and marketing of 'farmer-made cheese', landscape maintenance, tourist housing, and care for less-favoured and disabled people may provide additional income sources for the farmer, especially in rich and densely populated countries.

It is commonly accepted that changes in technology (e.g. new machines, ICT, breeding, food storages), markets (e.g., changes in regional/global balances of food demand and supply, consumer preferences, marketing strategies of retail), governmental policies (e.g. Common Agricultural Policy (CAP), agri-environmental policies, WTO-agreements), education and in culture together contribute to changes in farming systems and cropping systems (Mudgal et al., 2010). Especially during the second half of the 20th century, we have witnessed dramatic changes in agricultural systems. Farm size has steadily increased, and most farms have become more specialized (i.e., focussing on a smaller number of crops and/or animals), while the output of agricultural produce has strongly increased per unit of agricultural land and per labourer (intensification). The total workforce in primary agriculture has strongly decreased (Figure 3.2), but the number of workers in the food processing industry and marketing has increased (e.g., Mazoyer and Roudart, 2006). Climate change and extreme weather conditions may also lead to changes in cropping systems.

Figure 3.2. *Comparison of productivity per worker for various farming systems in the world. Subsistence farms and small farms are situated in the lower left corner, highly mechanized large farms in the upper right corner. Over time, the productivity per worker expressed in constant currency drops down, due to fall in the prices of agricultural products, visualized by a change from green-coloured to yellow-coloured farming systems. At the bottom, farms are in decline, because the cost of living goes up from R to R′ and R′′, i.e., the point of marginalization moves upward (after Mazoyer and Roudart, 2006).*

The role of food processing companies and retail become increasingly important (UNCTAD, 2009). Through mergers and buy-overs, food processing companies and retail have strongly increased in size and market power. Food prices are often determined by a small number of international corporations and large supermarkets. Food retailers have an increasing say in agricultural value chains, including the growth of bio-energy crops, outdoor grazing by dairy cows, and about limiting the carbon foot print of food products (e.g. Cool Farm Alliance, https://coolfarmtool.org/). In 2012, European retailers had 514 environmental commitments, varying from energy use, distribution, communication, certification, carbon food print, waste management and compliance of palm oil production (EC, 2012; Chkanikova & Mont, 2015) (EC, 2016).

Various certification schemes have been developed, such as the certification on organic farming. EU regulation on organic farming (EC No. 834/2007) support production, processing and control and labelling of organic food. GlobalG.A.P. is an international cooperation between supermarkets with a certification scheme which includes good agricultural practices (GAP) and includes soil and water management aspects (http://www.globalgap.org/uk_en/). A new

development is that various retailers and agricultural associations formed an European association (EISA, 2016) to promote sustainable farming systems (integrated farming). This EISA has additional measures compared to Good Agricultural Practices (GAP). To be certified, areas at risk of soil degradation must be documented, and long term crop rotation plans, crop residues management and soil management have to be defined and implemented.

The support schemes under the Common Agricultural Policies (EC, 2003) influence the income of farms and via its cross compliance regulations play a role in the implementation of agro-environmental regulations and best soil management practices. The CAP is used for three types of support: (i) income support and assistance for complying with sustainable agricultural practices (70% of the CAP budget), (ii) rural development measures (20% of the budget), and (iii) market-support measures, used when for example adverse weather conditions or political conflicts destabilise markets (10% of the budget) (EC, 2013). The proportion of direct payments to total agricultural income per farm differ with farm type and per member state, but is decreasing. On average, direct supports contributed 9140 € per farm in 2012. Direct payments represented a substantial part of the income in grazing livestock and mixed and field crop farms, and only a very limited part in granivores, wine and horticulture holdings (Table 3.1). The proportion varied among Member states; it was nearly 20% of the total receipts in Ireland, Greece and Finland, but only 3.5% in the Netherlands (EC, 2015).

Table 3.1. *Proportion of direct payments in total receipts by type of farming in 2012 (EC, 2015).*

#	Farmtype	%
1	Grazing livestock	18.3
2	Field crops	15.7
3	Mixed (crops and livestock)	13
4	Milk	10
5	Other permanent crops	12
6	Granivores	3
7	Wine	3.3
8	Horticulture	1.4

The EU targets for renewable energy (Directive 2009/28/EC) have also an influence on cropping systems, but with large differences between countries. The Renewable Energy Act (2000) and the NAWARO bonus (2004) have strongly stimulated the cultivation of energy crops in Germany; the share of energy crops was 13% of the agricultural used land in 2014 (FNR, 2015). In contrast, only 1 to 2 % of all agriculture land in EU-27 was devoted to the production of energy and biomass crops in 2014. The growth of maize as energy crops has been implicated for increasing water erosion due to the low vegetative soil cover after the seeding of the maize, the complete harvest of all biomass, and the lengthy period of bare soil between harvest and seeding in the new season (Vogel *et al.*, 2016).

Climate change has also effect on cropping systems. The impacts will differ across regions depending on direct effects of climate conditions and indirect effect caused by pests and pathogens (Barros *et al.*, 2015). In the north part of Europe agriculture may benefit from an extension of the growing season (Peltonen-Sainio *et al.*, 2009) while in the southern part the productivity will likely decline as the climate gets drier and warmer (Supit *et al.*, 2012). Climate change will increase irrigation needs which are constrained by risk on erosion (Kovats et al., 2014).

Farmers themselves may also contribute to changes in crop production due to the effects of education, personal preferences, and cultural changes (e.g. the wish to have more free time) (Porceddu & Rabbinge, 1997). There are large differences between countries in EU-28 in the education and training of farmers. Large farmers have in general higher education than small farmers (Eurostat, 2016d).

3.1 References

Barros, V., Field, C., Dokke, D., Mastrandrea, M., Mach, K., Bilir, T., Chatterjee, M., Ebi, K., Estrada, Y. & Genova, R. 2015. Climate change 2014: impacts, adaptation, and vulnerability. Part B: regional aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.

Chkanikova, O. & Mont, O. 2015. Corporate supply chain responsibility: drivers and barriers for sustainable food retailing. Corporate Social Responsibility and Environmental Management, 22, 65-82.

EC 2003. Council Regulation (EC) No 1782/2003 of 29 September 2003 establishing common rules for direct support schemes under the common agricultural policy and establishing certain support schemes for farmers and amending Regulations (EEC) No 2019/93, (EC) No 1452/2001, (EC) No 1453/2001, (EC) No 1454/2001, (EC) 1868/94, (EC) No 1251/1999, (EC) No 1254/1999, (EC) No 1673/2000, (EEC) No 2358/71 and (EC) No 2529/2001.

EC 2012. Monitoring of REAP commitments and technical support for Retail Forum issue papers http://ec.europa.eu/environment/industry/retail/pdf/reap_monitoring_report.pdf.

EC 2013. Memo. The common agricultural policy (CAP) and agriculture in Europe – Frequently asked questions. http://europa.eu/rapid/press-release_MEMO-13-631_en.htm.

EC 2015. EU Farm Economics Overview based on 2012 FADN data.

EC 2016. website http://ec.europa.eu/environment/archives/industry/retail/reap_2014/index_en.html (last accessed June 13, 2016).

EISA 2016. website http://sustainable-agriculture.org/about-eisa/ (last accessed June 13, 2016).

Eurostat 2016. Agriculture, forestry and fishery statistics 2015 edition.

FNR 2015. Basisdaten Bioenergie Deutschland. Fachagentur Nachwachsende Rohstoffe, Gülzow, 48.

Kovats, R. S., Valentini, R., Bouwer, L. M., Georgopoulou, E., Jacob, D., Martin, E., Rounsevell, M. & Soussana, J.-F. 2014. 2014: Europe. In: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Barros, V.R., C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White(eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1267-132.

Mazoyer and Roudart, 2006. A history of world agriculture from the neolithic age to the current crisis, Earthscan, London, United Kingdom.

Mudgal, S., Lavelle, P., Cachia, F., Somogyi, D., Majewski, E., Fontaine, L., Bechini, L. & Debaeke, P. 2010. Environmental impacts of different crop rotations in the European Union. European Commission, Brussels.
Oenema, O. , Klein, C. de, Alfaro, M. (2014) Intensification of grassland and forage use: driving forces and constraints. Crop and Pasture Science 65, 524 – 537.
Peltonen-Sainio, P., Jauhiainen, L., Hakala, K. & Ojanen, H. 2009. Climate change and prolongation of growing season: Changes in regional potential for field crop production in Finland. Agricultural and Food Science, 18, 171-190.
Porceddu, E. & Rabbinge, R. 1997. Role of research and education in the development of agriculture in Europe. European Journal of Agronomy, 7, 1-13.
Supit, I., van Diepen, C. A., de Wit, A. J. W., Wolf, J., Kabat, P., Baruth, B. & Ludwig, F. 2012. Assessing climate change effects on European crop yields using the Crop Growth Monitoring System and a weather generator. Agricultural and Forest Meteorology, 164, 96-111.
Vogel, E., Deumlich, D. & Kaupenjohann, M. 2016. Bioenergy maize and soil erosion — Risk assessment and erosion control concepts. Geoderma, 261, 80-92.

4 Overview of soil threats in Europe

M. Heinen

4.1 Introduction

The purpose of this chapter is to provide a brief overview of the soil threats in Europe, and thereby to emphasize indirectly the necessity of 'soil improving cropping systems'. Soils in Europe are relatively productive, but they are under threat. These threats are diverse and differ greatly between regions, farms and cropping systems.

"*With no soil there would be no life*". This statement introduces the soil's importance to people (Lindbo et al., 2012). Soils provide a base to build houses, to give space to feed animals, and to allow plants and trees to grow either for feeding, supplying raw materials for fabrics (e.g., cotton) or industrial use (e.g., wood, biofuel, bricks), and for recreation. Plants need soil as a base, and as a source for water, oxygen and nutrients.

Soils perform several essential functions for ecosystems and society, including (i) production of food, feed, fuel and fibre, (ii) regulating and purifying water, (iii) carbon regulation & sequestration, (iv) nutrient retention and cycling, (v) providing habitats for all sorts of organisms, (vi) providing an archaeological archive, and (vii) providing a building platform and resources of minerals (Landmark, 2016[2]; Sanderson et al., 2002). These functions of soil are under threat, through various possible forms of soil degradation, as discussed briefly below. For each threat, the relationship with soil quality and cropping systems are briefly indicated, using the anlytical DPSIR-framework further discussed in Chapter 5.

4.2 Soil threats

Soil threats affect soil quality, which is defined as "fitness for use" (Larson and Pierce, 1991) or as "the capacity of a soil to function" (Karlen et al., 1997). A more comprehensive definition is "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation" (Karlen et al., 1997).

Recently, Stolte et al. (2015) presented a comprehensive overview of soil threats, as studied in the EU project RECARE[3]. This section briefly summarizes that report, and briefly indicates how the soil threats affect soil quality and cropping systems. Soil sealing (the destruction or covering of soils by buildings, constructions and layers of completely or partly impermeable

[2] http://landmark2020.eu/
[3] http://www.recare-project.eu/

artificial material), is deliberately left out here, as it is not directly related to agriculture and soil-improving cropping systems.

4.2.1 Salinization

Definition – Salinization can be defined as the accumulations of water soluble salts in the soil, causing a deterioration or loss of one or more soil functions. Salt-affected soils can refer to: i) saline soils with elevated salt concentrations, ii) saline-sodic soils with a disturbed monovalent/divalent cation ratio in favour of the monovalent alkali cations (Na^+, K^+), and iii) sodic soils with a chemical composition skewed towards alkalinity (high pH) often caused by a dominance of (bi)carbonate anions in solution. A soil is considered saline if the electrical conductivity of its saturation extract (EC_e) is above 4 dS m^{-1} (Richards, 1954).


www.FAO.org

Occurrence – Tsanis et al. (2015) reported that about 3.8 Mha in the EU is affected by salinization, mainly in southern EU. Salinity due to sea water intrusion occurs along the coasts, including areas in NW-EU.

Effects on soil functions – Salinization negatively impacts on soil structure and lowers soil fertility, biomass production, soil biodiversity and microorganisms' activity (e.g. lower respiration, lower mineralization). Saline and sodic soils often have low water infiltration rate, leading to more runoff and erosion.

Drivers and pressures – Soil salinization is primarily subject to climatic drivers and secondary to human and policy drivers. Primary salinization involves the accumulation of salts by natural processes including physical or chemical weathering and transport from parent material, geological deposits or groundwater.

Salinity may also occur due to seawater intrusion as a result of sea level rise, seawater seepage and seawater infiltration into groundwater. Secondary salinization is caused by human interventions such as use of salt-rich irrigation water or other inappropriate irrigation practices, and/or poor drainage conditions (import of salts to the root zone from below).

Key indicators – Soil salinization can be characterized by the salt profile, the Exchangeable Sodium Percentage (ESP, %) together with the Sodium Adsorption Ratio (SAR, $(mol\ m^{-3})^{0.5}$), and a list of potential salt sources. The total ionic concentration of water can be expressed as Electrical Conductivity (EC, dS m^{-1}) or as Total Dissolved Solids (TDS, mg kg^{-1} or ppm).

Effects of cropping systems – Salinization severely limits crop production and effects crop quality. Some crops are more tolerant to salinity than others. Site-specific irrigation and drainage systems and tolerant crop types and varieties may help to enhance crop production.

4.2.2 Erosion

www.timesofmalta.com

Definition – Soil erosion can be described as a three-stage process: i) the detachment of individual soil particles from the soil mass by rain splash, water running over the soil or wind; ii) their subsequent transport by an erosive agent (water, wind); and, iii) their deposition when the erosive agent lacks sufficient energy for further transport (Morgan, 2005 as referred to by Keizer et al., 2015).

Occurrence – The extent and amount of soil erosion is not well known, because of lack of data and different approaches used across EU (Keizer et al., 2015). Erosion rates in EU are on average modest, compared to some other areas in the world: 1.2 ton ha^{-1} yr^{-1} (3.6 ton ha^{-1} yr^{-1} on arable land); about 105 Mha (or 17%) is subject to some degree of erosion. Severe erosion occurs incidentally with values exceeding 10 ton ha^{-1} yr^{-1}. Wind erosion is reported to occur in, e.g., northern Germany, eastern Netherlands, eastern England, and in the Iberian Peninsula: estimates range from 10-42 Mha, with rates of 0.1-2 ton ha^{-1} yr^{-1} and extremes of >10 ton ha^{-1} yr^{-1} (Borrelli et al., 2015).

Effects on soil functions – erosion affects food and biomass production: direct: removal of seeds, damage to plants; indirect: reduced rooting space, reduced availability of water and nutrients. Erosion negatively affects (amongst others): the soil's capacity for storage, filtering, buffering and transformation, and the soil's function as a habitat.

Drivers and pressures –Soil erosion depends on geomorphology (slope), soil type, vegetation cover and litter (mulch) cover, climate (rainfall, wind, freezing-thawing), human activities (land management, soil conservation techniques), and socio-economic and policy drivers (may induce changes in land use and land management).

Key indicators – Two indicators are commonly used: the area affected by soil erosion (km^2, or % of total area), and the magnitude of soil erosion or sediment (tons ha^{-1} yr^{-1}). These can be either measured or modelled (or a combination of both). For wind erosion several proxy indicators have been proposed as well (soil resistance, Ohm), surface roughness (%), wind velocity (km hr^{-1}), soil moisture content (%), and soil cover (ha or %)).

Effects of cropping systems – Erosion effects crop production negatively, due to the washing away of soil and plants in case of water erosion, and due to abrasion of seedlings during wind erosion. Subsequent sedimentation may also impact crop yield. Conversely, the choice of cropping systems and agro-techniques can greatly minimize erosion. Permanent cropping systems are usually superior in strongly reducing erosion. Contour ridging, terracing, cover crops, mulching, minimum tillage are also effective to reduce erosion.

4.2.3 Compaction

Definition – Soil compaction is defined as the densification and distortion of soil by which total porosity and air-filled porosity are reduced, causing deterioration or loss of one or more soil functions (van den Akker, 2008; Schjønning et al., 2015). Compaction may be induced by natural factors, including trampling of animals, and by heavy agricultural machinery.


www.omafra.gov.on.ca

Occurrence – Quantification of soil properties, typically for the subsoil, is laborious and costly. Thus there are not much quantitative data available. Schjønning et al. (2015) estimated the relative normalized density (RND) from the SPADE8 database and concluded that about 29% of the subsoils (excluding organic soils) have critically high RND values > 1.

Effects on soil functions – Soil compaction affects the soil pore system, and therefore, most of the soil functions, including i) food and other biomass production, ii) storage, filtering, buffering and transformation, and iii) filtering of contaminants. Soil compaction influences soil aeration, soil water movement, root penetration, and crop growth. Compaction is often persistent, especially in the subsoil. It may be overcome (often in top soil) by drying-wetting processes, freeze-thaw processes, the action of soil biota, and by soil tillage.

Drivers and pressures – The main drivers for compaction are via (in)direct human activities (often economically driven): outdoor livestock, and using (too) heavy machinery for land management. The impact can be reduced by spreading the weight of machinery over a larger area by wider tires with lower pressure or by rubber tracks. Climate may also impact on compaction because the ability of the soil to withstand mechanical stresses decreases with an increase in soil water content.

Key indicators – In the RECARE project the following three indicators were listed: i) Relative Normalized Density (RND, dimensionless or %; defined as the actual dry bulk density divided by a critical bulk density, the latter being a function of the clay content), ii) air-filled porosity (%), and iii) penetration resistance (MPa).

Effects of cropping systems – Soil compaction effects crop yields negatively, because crop roots are unable to explore compacted soils for water and nutrients effectively. Hence, crops grown in compacted soils often suffer from water and/or nutrient stress. The morphology of root crops (e.g. carrots, sugar beet) may also be negatively affected. Conversely, deep rooting crops like cereals, alfalfa, some cabbages and trees may alleviate compacted subsoil. Freeze-thawing in the topsoil, and and especially wetting-drying cycles in shrinking soils are also beneficial. Deep soil cultivation may be practiced too to alleviate soil compaction.

4.2.4 Contamination or pollution


blogs.egu.eu

Definition – Soil contamination is the occurrence of contaminants (any physical, chemical or biological agent) in soil above a certain level causing deterioration (irreversible or not) or loss of one or more soil functions. Soil pollution is often referred to as the activity that causes soil contamination. More than 700 pollutants have been documented in Europe. Prominent classes are: pharmaceuticals, pesticides, heavy metals, disinfection by-products, and wood preservation and industrial chemicals.

Occurrence –both point source and diffuse contamination or pollution can be distingsuied (e.g., Anaya-Romero et al., 2015). From the EIONET-SOIL analysis 2.5 million point polluted sites were identified with 11.7 million potentially polluted sites. The main sources are municipal and treatment wastes and commercial/industrial activities, with mineral oil and heavy metals as dominant pollutants. Diffuse pollution occurs with heavy metals, emerging pollutants, and agrochemicals. No data exist on the pesticide pollution in EU soils.

Effects on soil functions – Soil contamination affects the following soil functions: i) biomass production, and especially food crop quality and safety, ii) storing, filtering and transforming nutrients, substances and water, and carbon, iii) biodiversity pool: habitats, species and genes, and iv) physical and cultural environment for humans and human activities. Soil contaminants may limit the biodegradation of organic matter and may cause nutrient imbalances and deficiency.

Drivers and pressures – The main drivers of soil contamination are of anthropogenic character, and they originate from activities in industry, transport, waste management and agriculture. Several national and European regulations have been put in practice aimed at reducing the (negative) pressures of urban and industrial development.

Key indicators – Because of the diversity of pollutants, there is diversity of indicators. The top 3 indicators advocated by the ENVASSO project[4] are (Huber et al., 2008): i) heavy metal contents in soils, ii) critical load exceedance by sulphur and nitrogen (%), and iii) progress in management of contaminated sites (%). Other possible indicators are: concentration of persistent organic pollutants, topsoil pH, bioavailability of pollutants.

Effects of cropping systems – Pollution affects crop quality (and health) more than crop yield. However, serious pollution affects also crop yields. Specific soil amendments and liming alleviate the effects of pollution. Some crops and some varieties are more sensitive to pollution than others. Special crops may be grown to withdraw some pollutants from soil, through the process called phytoremediation.

[4] http://esdac.jrc.ec.europa.eu/projects/envasso

4.2.5 Decline in organic matter


www.deeproot.com

Definition – Decline of soil organic matter (SOM) is defined as a loss of organic matter mass in soils over time. The loss occurs mostly from the top soils, and may lead to a deterioration of soil structure and to a loss of soil functions. Soils containing ≥20-35% (by weight) of organic matter are classified as organic or peat soils; through drainage and net mineralisation organic soils may lose large amounts of organic matter, which shows up as subsidence. Loss of soil organic matter is associated with net CO_2 emission.

Occurrence – The total area of peat soils in Europe is 0.32 million km^2 and declining. About 50% of the peat area is in Norway, Finland, Sweden, United Kingdom; the remainder mainly in Ireland, the Netherlands, Germany, Poland and the Baltic states. For mineral soils there is uncertainty about SOM stocks and trends. Soils in Mediterranean countries have relatively low organic matter contents. Scandinavian countries have relatively high SOM level, due to the lower temperature and the moist climate. There is some evidence that climate change contributes to a decline in SOM.

Effects on soil functions – Organic matter is tied to many soil functions. Direct effects of SOM decline are i) a loss of nutrients and cation exchange capacity, ii) a loss of soil structure, iii) a reduction in water and nutrient use efficiency, iv) a reduction in the biological activity, and v) the reduction of the available water capacity (particularly in sandy soils).

Drivers and pressures – Drivers affecting the SOM content are: i) natural (climate, topography, soil type and properties, land cover/vegetation type), ii) anthropogenic activities (land use change, land management and soil cultivation, manuring, drainage).

Key indicators – For peat soils the key indicator is the stock of peat (Mt); as proxy indicators the water table depth (m), soil moisture content (%), soil temperature (^oC) and vegetation type (species) can be considered. For mineral soils the total carbon stock to 100 cm depth (t ha^{-1}), the clay: SOC ratio, the topsoil organic carbon content (% or g kg^{-1}), and the topsoil organic carbon stock (t ha^{-1}).

Effects of cropping systems – Choice of cropping system (crop type, crop rotation, soil cultivation, manuring, drainage) greatly influence the build-up or decline of organic matter in soil. Permanent cropping systems, perennial cropping systems, cereals, minimum tillage, manuring, green manures, are known to build-up organic matter. Intensive soil cultivation, growing root crops, and bare fallows are known to decrease SOM levels. Conversion of grassland to arable land is associated with a decrease in SOM; conversion of arable land to grassland into an increase in SOM levels.

4.2.6 Decline in soil biodiversity


esdac.jrc.ec.europa.eu

Definition – Soil biodiversity is generally defined as the variability of living organisms in soil and the ecological complexes of which they are part. Biodiversity includes: i) ecosystem diversity, ii) species diversity, iii) genetic diversity, iv) phenotypic diversity, and v) functional diversity. The threat 'decline in soil biodiversity' can be defined as the reduction of forms of life, living in soils (both in terms of quantity and variety) and of related functions.

Occurrence – Soil biodiversity does not decline independent of other factors and is usually related to some other deterioration in soil quality, aboveground biodiversity, and/or other threats. Before a decline in soil biodiversity can be well assessed, one should know the (current) state of soil biodiversity; a description of this can be found in the European Atlas of Soil Biodiversity (Tibbett, 2015). Soil biodiversity can be linked to other soil threats as soil sealing, erosion, organic matter depletion, salinization, contamination, and compaction.

Effects on soil functions – Activities of the soil biota are essential to most of the soil functions. Soil biota activities are essential to provide most of the ecosystem services that are considered typical of the wider landscape (Tibbett, 2015). These stretch beyond supporting food and fibre production, controlling erosion and attenuating pollution.

Drivers and pressures – The major pressures on soil that can cause a decline in soil biodiversity are: i) human activities concerning local land management, ii) socio-political factors, and iii) climate change effects. For land management aspects on organic matter management and effects of agricultural intensification (use of fertilizers, pesticides, and herbicides) are important. Climate change (drought, temperature) may not directly affect biodiversity (soil organisms are rather tolerant), but it has an indirect effect via interaction with other soil threats.

Key indicators – Measurement of biodiversity is challenging (Tibbett, 2015). Easy-to-use indicators include: i) earthworms diversity and fresh biomass (number m^{-2}, g fresh weight m^{-2}), ii) Collembola diversity (number m^{-2}, g fresh weight m^{-2}), and iii) microbial respiration (g CO_2 kg^{-1} soil). Tibbett (2015) mentioned two biodiversity indices: Simpson's Index and Shannon-Weaver Index. Both these indices are related to the relative abundance of each species present in a sample.

Effects of cropping systems – Soil biodiversity is related to soil borne diseases, which affects crop yield and quality of specific crops. Crop rotations are key to minimize soil borne diseases. Soil biodiversity may minimize also the need for pesticides. There is evidence that belowground biodiversity is positively related to aboveground biodiversity and vice versa, indicating that multi-species crops (swards) and intercropping stimulate the development of soil biodiversity.

4.2.7 Landslides

www.kgs.ku.edu

Definition – A landslide is defined as the movement of a mass of rock, debris, artificial fill or earth down a slope, under the force of gravity, causing a deterioration or loss of one or more soil functions. Landslides are classified on the type of movement (fall, topple, slide, lateral spread, and flow) and the type of material involved such as rock or soil. A gradual transition from landslides to floods (see Section 4.2.8) and vice versa may occur in areas with high soil erosion and local flooding potential.

Occurrence – Landslides dominantly occur in mountainous regions and on slopes (Szolgay et al., 2015). High to very high susceptibility areas include the Pyrenees, the Alps, and the mountainous areas in the south-eastern EU member states. Moderate to high susceptibility occurs, amongst others, in north and west Britain and Norway.

Effects on soil functions – Landslides affect stability and functionality of (man-made) structures and sometimes completely destroys these. The actual movement of soil mass can have negative effects on food production, biological habitats, environment interaction, physical and cultural heritages and sources of raw material. On the other hand, landslides can lead to a rejuvenation of soils favouring the development of new biological and ecological systems and the restoration of soil functions in a short time period (< 5 years).

Drivers and pressures – Climate and climate change control precipitation and snowmelt and these form the most important external drivers for landslides. Major socio-economic drivers for landslides are human-induced changes in land use (including infrastructure) and drainage. At EU level policies exist addressing risks of landslides, including the water framework directive and the EU floods directive.

Key indicators – According to ENVASSO[4] the main set of indicators are: i) occurrence of landslide activity (ha, km^2), ii) volume or mass of displaced material (m^3, km^3, ton), and iii) landslide hazard assessment. Other useful indicators include soil depth (to estimate volume of water that can be stored), slope angle (determining the forces acting on the soil), and the proximity to faults (determining diffuse weakness and earthquake activity).

Effects of cropping systems – Landslides have a dramatic effect on crop production and yield. Permanent cropping and especially trees can minimize the risk of landslides. Proper drainage is also importance. High-risk areas should not be used for agricultural purposes, but for forestry and nature conservation.

4.2.8 Flooding and water logging


www.123rf.com

Definition – Flooding is defined as the submergence of soil/land by water, including the submergence over the soil surface by a watercourse or water body, exceeding its embankment and/or by the accumulation of surface or subsurface drainage water from surrounding areas (Szolgay et al., 2015). Water logging is the process where the soil becomes saturated, either due to flooding or by water table reaching the soil surface.

Occurrence – Flooding can occur anywhere in Europe. According to a JRC study the flood damage potential shows hot spots in Britain, NW Europe (typically the Netherlands), the Po valley (Italy) and several East-EU countries (Barredo et al., 2008).

Effects on soil functions – Flooding has a direct effect on soil health (incl. soil structure, water holding capacity, soil fertility and nutrient availability, etc.), erosion, mudflows, deposition of sediments and debris, soil crusting, nutrient leaching, changes in microbial and fungi populations, changes in soil chemical properties, deterioration of soil aggregation, and temporary water logging (Szolgay et al., 2015). The direct impact is on food and biomass production, either through soil erosion, leaching of nutrients, or by waterlogging of the root zone. Floods also have impact on the soil as a platform for man-made structures (buildings, roads), and they can cause damage to cultural heritage.

Drivers and pressures – Climate and climate change control precipitation and snowmelt and these form the most important external drivers for flooding. Major socio-economic drivers for flooding are human-induced changes in land use and drainage.

Key indicators – Szolgay et al. (2015) listed a range of possible indicators, including: i) seasonality, magnitude and frequency of precipitation, rainfall intensity; ii) water level and duration; iii) extent of inundated area (ha); iv) flood frequency (occurrences per year). These indicators can be further elaborated at the plot, farm, and/or at the catchment scale.

Effects of cropping systems – Flooding has a devastating effect on crop yield and quality, especially during periods of relatively high temperature. Some crops are more sensitive to flooding than other crops. Proper drainage systems and growth of crops that tolerate some temporally flooding is the best strategy to minimize the impacts of flooding.

4.2.9 Desertification


desertification.wordpress.com

Definition –According to the United Nations Convention to Combat Desertification (UNCCD), desertification is defined as land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities. Desertification is often the result of soil erosion, loss of soil fertility, and long-term loss of natural or desirable vegetation.

Occurrence – In the DISMED[5] project the sensitivity to desertification was mapped based on soil quality and vegetation parameters. About 8% (approximately 14 Mha) of the territory in southern, central and eastern Europe shows high sensitivity to desertification. Another 26 Mha was mapped as moderately sensitive.

Effects on soil functions – Degraded soils lose their capacity to capture and store water, nutrients and carbon, and to support microbiological processes. Desertification has negative effects on all soil functions. A decline in desertification has strong positive effects on: i) food and other biomass production, ii), environmental interaction: storage, filtering, buffering and transformation (including carbon pool) and iii) biological habitat and gene pool.

Drivers and pressures – Desertification is influenced both by bio-physical and socio-economic drivers. The influence of socio-economic drivers is via land use and land management. Climate change may lead to the expansion of the area that is susceptible to desertification: e.g., top soil may become drier, increasing chance of wildfires. Rural depopulation causes land to be abandoned which is then no longer sustainably maintained and thus become susceptible to desertification. Improper land management (e.g. overgrazing) including overuse of available resources is another form of pressures on the land. Policies may directly and indirectly affect desertification through their effects on land use and land management, and the intensification of agriculture.

Key indicators – There are various possible indicators; Kirkby et al. (2015) listed 148 candidate indicators. The ENVASSO[4] project proposed three indicators: i) land area at risk of desertification (ha), ii) land area burnt by forest fires (ha), and iii) soil organic carbon content in desertified areas (%, g kg^{-1}).

Effects of cropping systems – Evidently, desertification has devastating effects on the crop production potential. A limited number of perennial, permanent crop types are possible without inputs (e.g., irrigation, fertilizers). Management is a key factor here; overgrazing and intensive cropping must be prevented.

[5] https://www.eea.europa.eu/data-and-maps/data-providers-and-partners/desertification-information-system-for-the-mediterranean

4.2.10 Acidification

Definition – Soil acidification is a process where the soils' acid neutralizing capacity decreases over time, followed by a drop in pH (the log_{10} of the H^+ concentration in soil solution). This process is accelerated by atmospheric deposition (acid rain), crop harvest, and the use of acidifying fertilisers.

sustainableagriculture.perth regionnrm.com

Soil acidification was not considered in previous EU projects such as RECARE and ENVASSO. Here we made use of the concise information provided by the Australian Government of Queensland[6], SoilQuality.org[7], EU Eurostat[8] and JRC[9].

Occurrence – According to a JRC survey 16.7% of the EU-27 territory has pH values (pH-$CaCl_2$) lower than 4.2 and 1.9 % of the area have values of pH > 8. Low pH values are found in the granitic areas of Portugal and north of Spain, in the Vosges mountains, in the Pyrenees, and in the shallow soils from Scandinavia, mainly developed on acidic parent materials.

Effects on soil functions – Soil pH affects the soil's physical, chemical, and biological properties and processes, as well as plant growth. Acidic soils may have some or all of the following problems: i) reduction biological activity and nutrient recycling, ii) reduction in phosphate availability, iii) nutrient deficiencies (e.g., Ca, Mg, Mo), iv) reduction of use of subsoil water by roots, v) release of toxic substances (e.g., Al, Mn), and vi) increase of contaminant uptake by roots (e.g., Cd).

Drivers and pressures – Factors that contribute to soil acidification are mainly human driven: i) the application of high levels of ammonium-based N fertilisers, ii) leaching (resulting from climatic factors) of cations and nitrates (often originally applied as ammonium-based fertilisers), iii) harvesting plant materials, especially leguminous crops, and iv) deposition of acidifying substances (sulphur dioxide, nitrogen oxides and ammonia).

Key indicators – The main key indicator is the soil pH, which can be measured either in a soil:water extract (pH-H_2O) or in a KCl or $CaCl_2$ soil extract (pH-KCl; pH-$CaCl_2$). Soil pH values range from 0 to 14, with pH = 7 being neutral, pH < 7 being acidic, and pH > 7 being alkaline (or basic). Optimal pH values for proper crop growth lies in the range 5.5-7.

Effects of cropping systems – Acidification negatively affects crop yield and quality. Some crops are more sensitive to acidity than other crops. To minimise soil acidification the following can be done: i) use less acidifying farming practices, ii) apply lime, stone meal, manures, and iii) decrease the deposition of N and S compounds through policy measures.

---

6 http://www.qld.gov.au/environment/land/soil/soil-health/acidification/
7 http://soilquality.org/
8 http://ec.europa.eu/agriculture/envir/report/en/acid_en/report.htm
9 http://eusoils.jrc.ec.europa.eu/library/Data/PH/Documents/pH_Pub.pdf

4.3 References

Anaya-Romero, M., T. Marañón, F. Cabrera, E. Madejón, P. Madejón, J.M. Murillo, N.-O. Vrinceanu, G. Siebielec, and V. Geissen. 2015. Soil contamination. *In*: Stolte, J., M. Tesfai, and L. Oygarden (eds.). 2015. Soil in Europe – Threats, functions and ecosystem services, pp. 109-124.

Barredo, J.I., P. Salamon, L. Feyen, R. Dankers, K. Bódis, and A. De Roo. 2008. Flood damage potential in Europe. Available at: http://biblio.central.ucv.ro/bib_web/bib_pdf/EU_books/0020.pdf (last access: 26-09-2017).

Borrelli, P., P. Panagos, R. Hessel, M. Riksen, and J. Stolte. 2015. Soil erosion by wind. *In*: Stolte, J., M. Tesfai, and L. Oygarden (eds.). 2015. Soil in Europe – Threats, functions and ecosystem services, pp. 25-39.

Huber S., , G. Prokop, D. Arrouays, G. Banko, A. Bispo, R.J.A. Jones, M.G. Kibblewhite, W. Lexer, A. Möller, R.J. Rickson, T. Shishkov, M. Stephens, G. Toth, J.J.H. Van den Akker, G. Varallyay, F.G.A. Verheijen, A.R. Jones (eds). 2008. Environmental assessment of soil for monitoring, Volume I: indicators & criteria. EUR 23490 EN/1, Office for the Official Publications of the European Communities, Luxembourg; available at: http://esdac.jrc.ec.europa.eu/Projects/Envasso/ (last accessed at 26-09-2017)

Karlen, D.L., M.J. Mausbach, J.W. Doran, R.G. Cline, R.F. Harris, and G.E. Schuman. 1997. Soil quality: A concept, definition, and framework for evaluation. Soil Sci. Soc. Am. J. 61: 4-10.

Keizer, J., H. Djuma and V. Prasuhn. 2015. Soil erosion by water. *In*: Stolte, J., M. Tesfai, and L. Oygarden (eds.). 2015. Soil in Europe – Threats, functions and ecosystem services, pp. 10-24.

Kirkby, M.J., R. Hessel, and A. Bruggeman. 2015. Desertification. *In*: Stolte, J., M. Tesfai, and L. Oygarden (eds.). 2015. Soil in Europe – Threats, functions and ecosystem services, pp. 139-151.

Larson, W.E., and F.J. Pierce. 1991. Conservation and enhancement of soil quality. *In*: J. Dumanski, E. Pushparajah, M. Latham, and R. Myers, eds. Evaluation for Sustainable Land Management in the Developing World. Vol. 2: Technical Papers. Proc. Int. Workshop., Chiang Rai, Thailand. 15-21 Sept. 1991. Int. Board for Soil Res. and Management, Bangkok, Thailand. (as referred to at http://soilquality.org/basics/definitions.html; last accessed at 26-09-2017)

Lindbo, D.L., D.A. Kozlowski, and C. Robinson (eds). 2012. Know Soil Know Life. SSSA, Madison, WI, USA.

Morgan, R.P.C. 2005. Soil erosion and conservation, 3rd edition. Blackwell Publishing Ltd., Malden, MA, USA.

Richards, L.A. (ed.). 1954. Diagnosis and improvement of saline and alkali soils. Agricultural Handbook No. 60, USDA, Washington D.C., USA

Sanderson, E.W., M. Jaiteh, M.A. Levy, K.H. Redford, A.V. Wannebo, and G. Woolmer. 2002. The human footprint and the last of the wild. BioScience 52: 891-904.

Schjønning, P., J.J.H. van den Akker, T. Keller, M.H. Greve, M. Lamandé, A. Simojoki, M. Stettler, J. Arvidsson, and H. Breuning-Madsen. 2015. Soil compaction. *In*: Stolte, J., M. Tesfai, and L. Oygarden (eds.). 2015. Soil in Europe – Threats, functions and ecosystem services, pp. 80-93.

Stolte, J., M. Tesfai, and L. Oygarden (eds.). 2015. Soil in Europe – Threats, functions and ecosystem services. Recare Project Report D2.1. Available at: http://www.recare-project.eu/ (last accessed: 26-09-2017). Also appeared as Stolte et al. (eds.), 2016, Soil Threats in Europe. JRC Technical Report JRC 98673, EUR 27607 EN; doi:10.2788/488054 (print); doi:10.2788/828742 (online); available at: https://ec.europa.eu/jrc/en/publication/soil-threats-europe?search (last accessed: 26-09-2017)

Szolkay, J., C. Camerra, E. Skarbøvik, and K. Hlavcova. 2015. Flooding and landslides. *In*: Stolte, J., M. Tesfai, and L. Oygarden (eds.). 2015. Soil in Europe – Threats, functions and ecosystem services, pp. 152-173.

Tibbett, M. 2015. Decline in soil biodiversity. *In*: Stolte, J., M. Tesfai, and L. Oygarden (eds.). 2015. Soil in Europe – Threats, functions and ecosystem services, pp. 174-184..

Tsanis, I.K., I.N. Daliakopoulos, A.G. Koutroulis, G.P. Karatzas, E. Varouchakis, and N. Kourgialas. 2015. Soil salinization. *In*: Stolte, J., M. Tesfai, and L. Oygarden (eds.). 2015. Soil in Europe – Threats, functions and ecosystem services, pp. 125-138.

Van den Akker, J.J.H. 2008. Soil compaction. *In*: Huber, S., Prokop, G., Arrouays, D., Banko, G., Bispo, A., Jones, R.J.A., Kibblewhite, M.G., Lexer, W., Möller, A., Rickson, R.J., Shishkov, T., Stephens, M., Toth, G.

Van den Akker, J.J.H., Varallyay, G., Verheijen, F.G.A., Jones, A.R. (Eds.) Environmental Assessment of Soil for Monitoring: Volume I Indicators & Criteria. EUR 23490 EN/1, Office for the Official Publications of the European Communities, Luxembourg, pp 107-124.

5 An analytical framework for assessing soil-improving cropping systems

O. Oenema and M. Heinen

5.1 Concept of soil-improving cropping systems

Soil-improving cropping systems (SICS) are defined as: "*cropping systems that result in a durable increased ability of the soil to fulfil its functions, including food and biomass production, buffering and filtering capacity, and other ecosystem services*". Cropping systems refer to the crops and crop sequences (rotations) and the management techniques used on a particular field over a period of years. SICS encompass soils/land, crops, inputs, and management (Table 5.1; Figure 5.1). Inputs refer to labour, machines, irrigation, pesticides, fertilizers, manures. The concept of SICS appears at first sight broader than the concept of soil conservation, which strongly focusses on preventing erosion and conserving soil and water (Blanco and Lal, 2008), but seems rather similar to the concept of sustainable soil management recently promoted by FAO (FAO, 2017).

Table 5.1. *Components of cropping systems that can be adjusted so as to create soil improving cropping systems (SICS).*

Nr	Components of cropping systems
A	Crop rotations, including cover crops, etc.
B	• Nutrient management, techniques and inputs
C	• Irrigation management, techniques and inputs
D	• Drainage management and techniques
E	• Tillage management, techniques and inputs
F	• Pest management, techniques and inputs
G	• Weed management, techniques and inputs
H	• Residue management, techniques and inputs
J	• Mechanization management, including planting and harvesting machines
K	• Landscape management techniques and inputs

Management is often called the 'fourth production factor' next to the traditional production factors land, labour and capital. Management encompasses a coherent set of activities, in this case related to the cultivation of crops and land, and the handling and allocation of inputs, to achieve objectives (including agronomic, economic, environmental, social objectives). Management is target oriented; in the case of SICS, management activities are also targeted at improving soil quality and preventing/minimizing soil threats.

Following the law of the optimum, which was formulated more than one hundred years ago (Liebscher 1895; De Wit, 1992), all crop yield influencing factors and soil quality improving factors need to be 'optimal' to make soil improving cropping systems effective, efficient and thereby attractive. Hence the ideal SICS consist of a particular crop rotation and an optimal combination of inputs, techniques and management (Table 5.1), as function of soil type (soil threat), climate, and socio-economic conditions. If there is no optimal combination of crop rotation and inputs, techniques and management, soil quality may be under threat and/or crop yields, farm profitability and sustainability may be suboptimal.

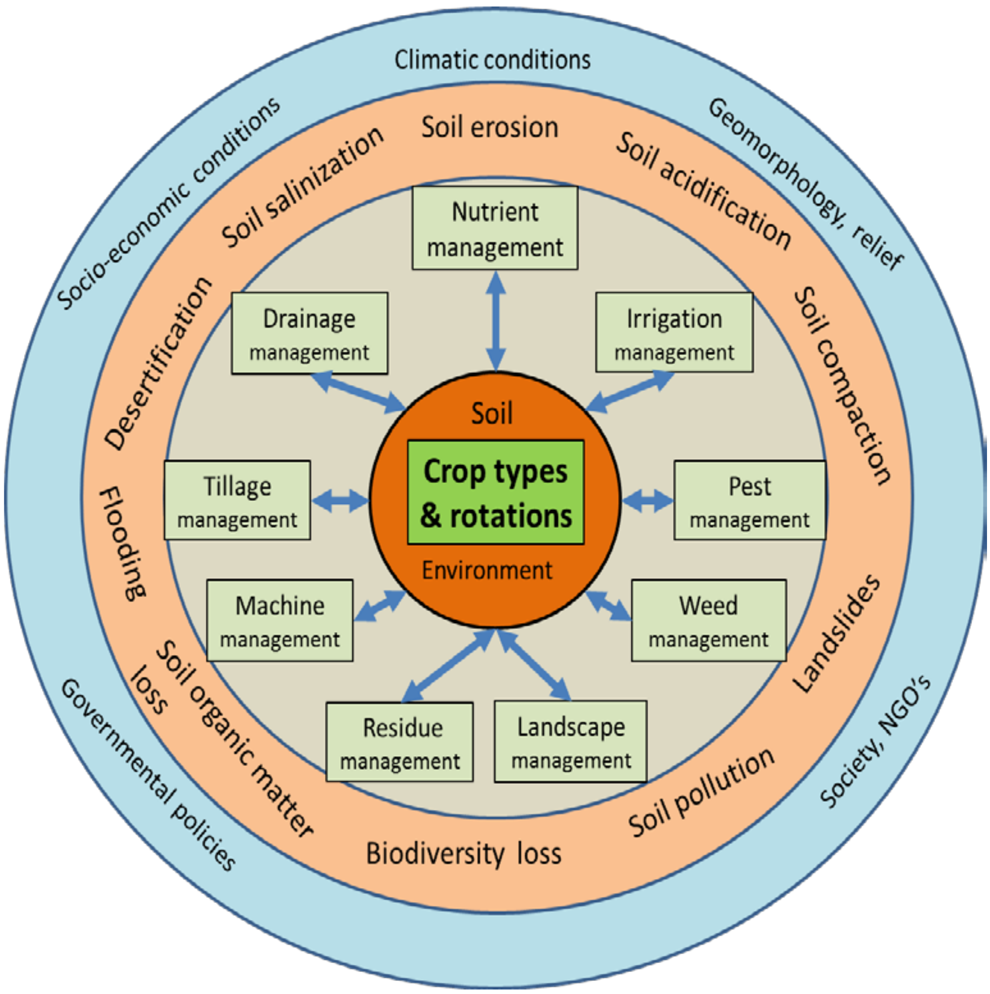


Figure 5.1. *Concept of Soil Improving Cropping Systems (SICS), with crop rotations and the soil environment in the centre and the nine key agro-management techniques (light-green boxes) surrounding and directly affecting soil quality and the sustainability of cropping systems. Soil threats (light-brown circle) are surrounding the SICS, while the external driving forces for the soil threats and SICS are in the outer (light-blue) circle.*

The wider circle of the SICS concept presented in Figure 5.1 encompasses the external driving forces of both soil threats and SICS. Various drivers have been distinguished, including (i) natural (climate, geomorphology, hydrology), (ii) socio-economic conditions (development in markets, including developments in science and technology), (iii) societal opinions and NGO's, and (iv) governmental policies. The last two seem important for SICS. The Common Agricultural Policy (CAP) of the European Union provides several incentives to stimulate the adoption of components of SICS, including crop rotation, permanent cropping systems, biodiverse strips, soil organic matter maintenance, and erosion control (EEA, 2016; Frelih-Larsen et al., 2017; Berge et al., 2017). Further, there are various voluntary measures with compensation for cost incurred and/or income forgone in the Rural Development Program. These EU-governmental policy measures address some main soil threats, including soil organic matter decline, soil biodiversity decline and erosion. The EU fertilizer, pesticide and animal feed Regulations (and many national policies) provide incentives to minimize the inputs of possible contaminant materials into agriculture and thereby safeguard food quality and prevent/minimize soil pollution. There are also strict regional/national regulations in landslide-prone areas aimed at minimizing the risk of landslides. Further, countries with desertification-prone areas and soil degradation problems are under the regime of the United Nations Convention to Combat Desertification (UNCCD) with a legally binding international agreement. However, other soil threats like acidification, compaction, salinization, soil structure deterioration, and soil nutrient imbalances are not addressed specifically, and there are no clear incentives to address/maintain and improve soil quality in general.

The implementation of SICS in practice depends on the decisions of farmers and land managers. Figure 5.2 briefly presents the decision environment of the farmer. Crop rotations and agro-management techniques are selected while considering socio-economic conditions (markets, policy, technology incentives), environmental conditions (soils, climate), and own preferences. In SICS, the decisions about crop rotations and agro-management techniques are also based on (i) preventing soil threats, (ii) alleviating the effects of soil threats, and (iii) enhancing soil quality and functions in general. This requires that the farmer is (a) convinced about the need to do so, (b) is able to do so, and (c) has the information and tools to do so. Hence, the crop rotations and agro-management techniques are also based on the occurrence of soil threats and the need to enhance soil quality and functions.

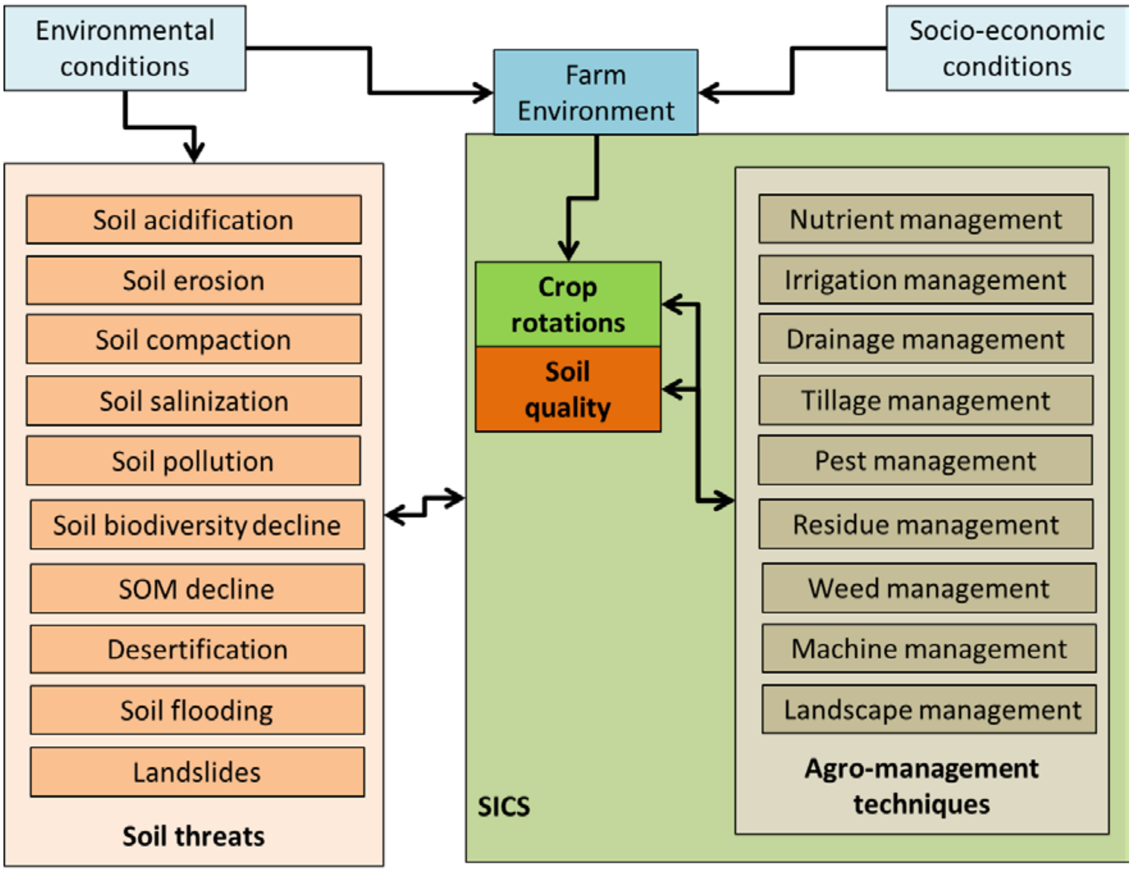


Figure 5.2. *Main driving forces and components of cropping Systems. The farmer selects the crop rotation and then the agro-management techniques while considering socio-economic conditions (markets, policy technology), environmental conditions (soils, climate and emanating soil threats), and own preferences.*

<u>Crop rotations and the 9 agro-management techniques are the tools for deriving optimal SICS.</u> Following the law of the optimum, all growth limiting and reducing factors have to be considered (removed/minimized) for establishing profitable and sustainable cropping systems. The law of the optimum is often implicitly expressed by the term 'integrated' in for example integrated pest management and integrated nutrient management. It is also expressed by the terms 'controlled', 'enhanced' and 'smart'. These terms emphasize that all factors for enhancing soil quality and the profitability and sustainability of cropping systems have to be considered in an harmonious, integrated manner, and that side-effects of the measures have to be considered as well.

In practice, farm income is commonly maximized (through lowering cost and increasing yield/sales) in the optimization of cropping systems, so as to provide sufficient income to farmers, who increasingly have to compete in a globalized market, on the basis of the cost of production (e.g., Mazoyer and Roudart, 2006). This competition and intensification of cropping

systems is one of the causes of soil threats, and at the same time a barrier for implementing SICS, because farmers give priority to farm income to be able to survive in what Mazoyer and Roudart (2006) call 'the global rat race'.

This indicates that greater priority has to be given to SICS; the need for specific crop rotations and specific agro-management techniques must receive greater priority (setting more serious constraints) in the cropping system optimization. The prioritized crop rotations and prioritized agro-management techniques depend on the site-specific conditions. Most promising SICSs consist therefore of particular crop rotations and an 'integrated' combination of inputs and management techniques, which reflect a site-specific prioritization and subsequent optimization process. The prioritization has to precede the optimization process. Hence, the priority crop types, crop rotations and agro-management techniques are the constraints in the optimization process. Alternatively, equal weight is given to farm profitability and soil quality (and/or the priority crop types, crop rotations and agro-management techniques of SICS).

Prioritization and optimization of crop types, crop rotations and agro-management techniques, as function of site-specific socio-economic and environmental conditions is the key to successful SICS. <u>The proof of the SICS concept is in the prioritization of specific crop rotations and specific agro-management techniques, and the subsequent optimization (and ultimately in the testing.</u>

The action of soil improving cropping systems may be brought about through three principles or mechanisms (Wezel et al., 2014), i.e.,

i) changes in input-output ratio's,
ii) substitution, and
iii) redesign.

The first mechanism relates to inputs (in relation to outputs), including water (irrigation, drainage), nutrients, pesticides, energy, etc. Substitution practices refer to the substitution of an input or practice by another input or practice (e.g., labour vs machines vs pesticides). Redesign refers to changes in crop types, crop rotations, farming systems, and/or market orientation (e.g., specialization vs diversification, commodities vs special niche products, conventional vs organic). Here our focus is on mechanisms that can be handled at the farm level.

Box 1. Brief characterization of the 13 environmental zones adopted in this study (after Metzger et al., 2005)

Z1 Alpine North (ALN): Scandinavian mountains; these have been named Alpine north, because they show environmental conditions as the Alps on a higher latitude, but in lower mountains.

Z2	Alpine South (ALS): The high mountains of central and southern Europe that show the environmental conditions of high mountains. Also small Alpine patches are found in mountain areas in Pyrenees and Carpathians.
Z3	Atlantic North (ATN): The area under influence of the Atlantic ocean and the North sea, humid with rather low temperatures in summer and winter, but not extremely cold.
Z4	Atlantic Central (ATC): The area with moderate climate where the average winter temperature does not go far below 0°C and the average summer temperatures are relatively low. This is a main agricultural production zone in EU-27.
Z5	Boreal (BOR): The environmental zone covering the lowlands of Scandinavia
Z6	Continental (CON): The part of Europe with an environment of warm summers and rather cold winters. This is a main agricultural production zone in EU-27.
Z7	Lusitenean (LUS): The southern Atlantic area from western France to Lisbon. Here, summers are rather warm and sometimes dry, while winters are mild and humid. This is a main agricultural production zone in EU-27.
Z8	Mediterranean North (MDN): The Mediterranean north represents the major part of the Mediterranean climate zone with Cork Oak, fruit plantations and Olive groves
Z9	Mediterranean Mountains (MDM): These mountains are influenced by both the Mediterranean and mountain climates.
Z10	Mediterranean South (MDS): This zone represents the typical Mediterranean climate that is shared with northern Africa, short precipitation periods in winter and long hot, dry summers.
Z11	Nemoral (NEM): The zone covering the southern part of Scandinavia, the Baltic states and Belarus. This is a main agricultural production zone in EU-27.
Z12	Pannonian (PAN): This is the most steppic part of Europe, with cold winters and dry hot summers. Most precipitation is found in spring.
Z13	Anatolian (ANA): Represents the steppes of Turkey, a Mediterranean steppic environment.
Z14	OTHER: Regions outside EU (please indicate which of the 13 EU zones fits best)

Soil types, cropping systems and soil threats greatly vary across Europe due to different environmental (e.g. Box 1) and socio-economic conditions. Hence, soil management and SICS are also site and cropping system specific. Further, SICS can be soil threat-specific, i.e., specific in prevented or overcoming a certain soil threat, as well as have a more general soil quality improving mode of action. Both approaches have been implemented in SOILCARE; Chapters 6-15 discuss subsequently soil threat-specific SICS, while Chapter 16 discusses SICS aimed at improving soil quality in general. Soil threat-specific SICS have the advantage of being specific, but may thereby neglect other aspects of soil physical, chemical and/or biological degradation than that of the soil threats, and/or make integration of various soil-threat-specific SICS more complicated. SICS with a more general mode of action may have the potential advantage of greater applicability, but run the risk that specific soil threats are not addressed effectively and efficiently.

5.2 Methodology of assessing SICS

Results of specific components of SICS have been summarized where possible as relative effects, i.e., the ratio of the specific treatment and the reference (control treatment), according to

$$ES = \frac{Y_T - Y_C}{Y_C} = \frac{Y_T}{Y_C} - 1 \qquad (1)$$

where ES is the effect size (dimensionless; or percentage), Y_T is the component observed (e.g. yield), and Y_C is the component of a reference or control treatment. In case a treatment does not result in a (significant) different outcome than the control treatment, then $ES = 0$. For $Y_T > Y_C$ this results in $ES > 0$, and vice-versa.

In meta-analyses studies the means and standard deviations of the effects are often determined based on ln-transformed ratio's (following the protocol of Hedges et al., 1999) as given by

$$L = \ln\left[\frac{Y_T}{Y_C}\right] \qquad (2)$$

Once the ln-transformed average ratio (and standard deviation) are known, it can be back-transformed to obtain the average effect size according to

$$ES_{avg} = \exp\left[L_{avg}\right] - 1 \qquad (3)$$

Similarly the confidence interval for ES can be determined by back-transforming the confidence interval limits for L. In what follows we assume that the reported average ES is significant when the available confidence interval (based on standard deviation) does not include the value zero. Formal meta-analysis studies often are based on the ln-transformed approach, whereas single studies and some reviews mostly consider the effect size or the ratio Y_T/Y_C.

One cannot generalize the interpretation of ES that positive values for ES are always the best. Sometimes $ES > 0$ indicates an improvement, e.g., an increase in yield due to the implementation of a certain SICS. In other cases $ES < 0$ indicates an improvement, e.g., a decrease in leaching due to the implementation of a certain SICS.

A common way to present the outcome of meta-analyses for ES (or L) is by presenting this in so-called forest plots (Figure 5.3). In a forest plot the effect size ES (or L) is plotted on the horizontal axis for different studies (or studied quantities) as listed along the vertical axis. At the left side of the forest plots studies or quantities are listed, in the middle part the average ES is plotted as a symbol together with a confidence interval (e.g. ± standard deviation; or, 95% confidence interval). At the right side sometimes additional information is provided regarding the number of underlying studies. In the middle part a vertical line is drawn that indicates the reference situation, i.e. at $ES = 0$ (or $L = 1$). A certain effect is significant when the available confidence interval (based on standard deviation) does not include the reference value, i.e. does not intersect the vertical line.

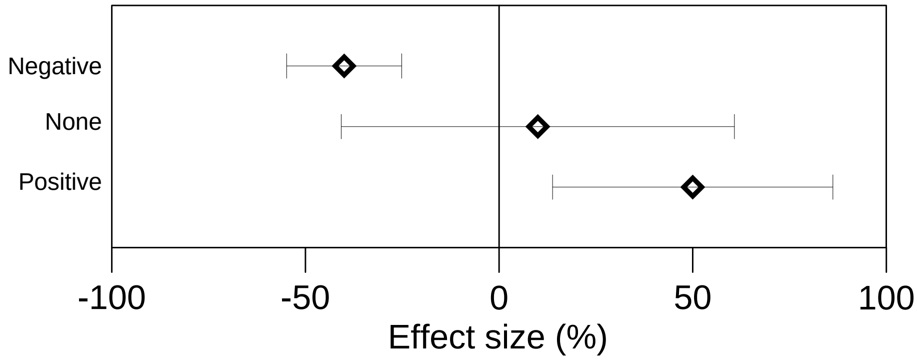


Figure 5.3. *Example of a forest plot with the effect size on the horizontal axis. For each casus the mean value is denoted by a symbol surrounded by a confidence interval (e.g. 95%). The top casus indicates a significant negative effect, the middle cases indicates no effect(the mean value is larger than zero, but tthe value zero falls in the confidence interval), and the lower casus refers to a significant positive effect.*

Literature data on specific effects of crop types, crop rotations and agro-management techniques were not always available. In case, quantitative literature data were missing, but general descriptions were available, expert judgements were made by the authors, using a simple scheme. The reference (control) has been given a score of 0 (zero), a positive effect of the specific treatment in terms of productivity and sustainability has been given the score + or ++, while a negative score has been given the score – or --, using the following key:

0	reference
+	positive effect of 5 to 10% relative to the reference
++	strong positive effect of significantly more than 10% of the reference
-	negative effect of 5 to 10% relative to the reference
- -	strong negative effect of significantly more than 10% of the reference
-/+	unclear effect, but tendency towards a negative effect (up to 5%)
+/-	unclear effect, but tendency towards a positive effect (up to 5%)

5.3 Analytical framework

In order to evaluate whether or not cropping systems can prevent soil threats and remediate soil degradation it must be known what cause-effect relationships (mechanisms) play a role. A cause-effect analytical framework links driving forces, to soil threats and soil quality, and soil-improving cropping systems to driving forces, soil threats and soil quality. We adopt here the classical Drivers-Pressures-States-Impacts-Responses approach (DPSIR; EEA, 2007) (Figure 5.4). The *driving forces* can be either environmental or biophysical (including pedo-climatic zonation and soil type) or external (driven by the market, policy, technology, etc.), the *pressures* here are the soil threats, the *states* are soil quality indicators, whereas the *impacts* relate to the soil functions. *Soil improving cropping systems* (SICS) are seen as a possible response by farmers and land managers to soil threats and soil degradation. The impacts of SICS are considered in terms of changes in soil functioning, but possible other effects (side-effects) should be considered as well, including crop yield, resource use efficiency, farm income, environmental

impacts, and human health effects. Hence, impacts of SICS should be analysed in a broader sense than the 5 or 7 soil functions defined earlier, to make overall judgments about the effectiveness, efficiency and applicability of SICS (Chapters 1, 4).

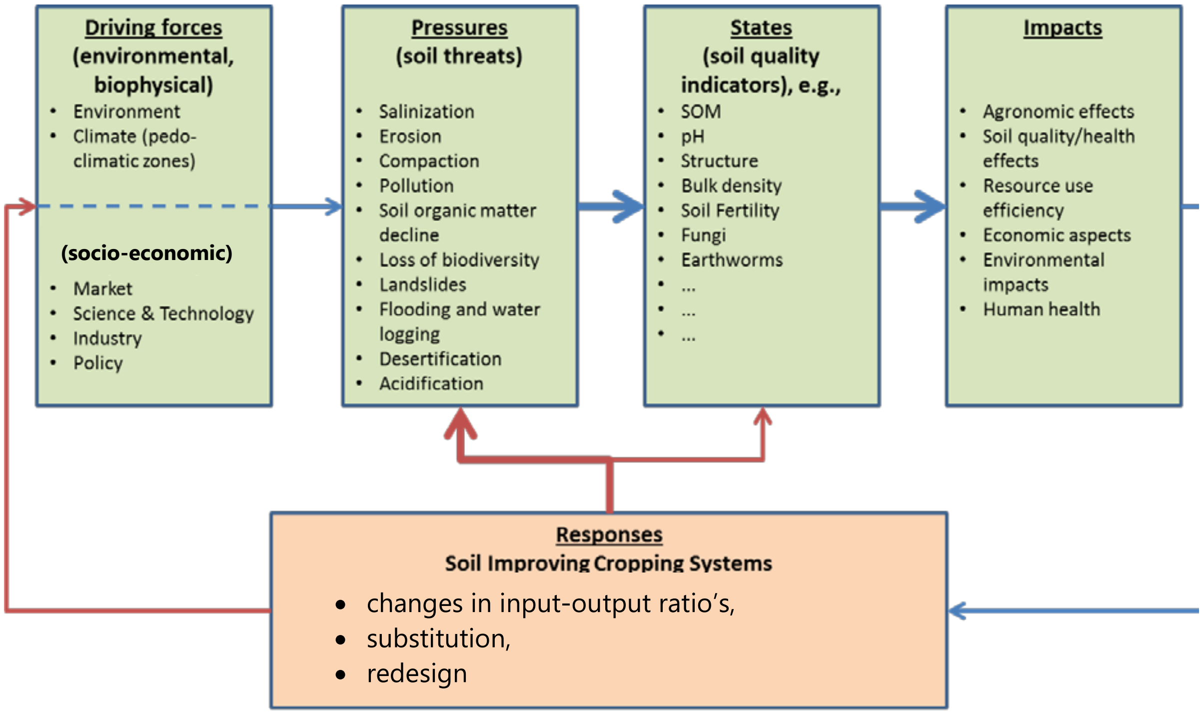


Figure 5.4. *The analytical framework for assessing SICS: the Drivers-Pressures-States-Impacts-Responses (DPSIR) approach.*

In the following chapters (Chapters 6-15) the influence of SICS on the alleviation of specific soil threats and soil functions are discussed. Chapter 16 provides a review on soil-improving cropping systems in general.

The driving forces, pressures, states and impacts of soil threats are briefly introduced in the Introductory sections (Background) of the following Chapters 6 to 15, for each soil threat separately. However, the emphasis in the subsequent chapters is on reviewing the effectiveness and efficiency of SICS, as described in the literature and database sources. A brief summary of the DPSIR-SICS framework applied to soil threats is presented in Chapter 4.

Several databases have been explored, including the database of the World Overview of Conservation Approaches and Technologies (WOCAT, 2017), and the databases of the Survey on Agricultural Production Methods (SAPM), Farm Structure Survey (FSS), Farm Accountancy Data Network (FADN). However, the review focused on soil improving cropping systems, while the original literature references have been used in the list of references for each chapter. Results of a large survey among crop farmers in European countries became only recently available (Hijbeek et al., 2017) and could not be included in the current review.

The description of action (DOA) of SoilCare mentioned also that 'threshold values will be derived, using amongst others work done in the ongoing EU-project iSQAPER project'. However, it turned out that the expected results from the iSQAPER project were not yet available when this review report had to be completed. Also, the concept of 'thresholds' implicitly may suggest that SICS would be needed only when some 'soil quality threshold' has been surpassed. This concept is against the idea of preventing soil threats and the concept of general SICS further discussed in Chapter 16.

5.4 References

Berge, H.F.M. ten, Schroder, J.J., Olesen, J.E. and Giraldez Cervera, J.V. 2017, Research for AGRI Committee – Preserving agricultural soils in the EU, European Parliament, Policy Department for Structural and Cohesion Policies, Brussels.

Blanco H, and R Lal. 2008. Principles of Soil Conservation and Management. Springer. ISBN: 978-1-4020-8708-0, 256 pp.

De Wit CT. 1992. Resource Use Efficiency in Agriculture. Agricultural Systems 40, 125-151.

EEA, 2016. The direct and indirect impacts of EU policies on land. European Environment Agency EEA., no. 8. 2016.

EEA. 2007. The DPSIR framework used by the EEA. European Environment Agency http://ia2dec.pbe.eea.europa.eu/knowledge_base/Frameworks/doc101182 (last access: 10-June-2016).

FAO. 2017. Voluntary Guidelines for Sustainable Soil Management. Food and Agriculture Organization of the United Nations, Rome, Italy, 26 pp.

Frelih-Larsen, A., C. Bowyer, S. Albrecht, C. Keenleyside, M. Kemper, S. Nanni, S. Naumann, R., D. Mottershead, R. Landgrebe, E. Andersen, P. Banfi, S. Bell, I. Brémere, J. Cools, S. Herbert, A. Iles, E. Kampa, M. Kettunen, Z. Lukacova, G. Moreira, Z. Kiresiewa, J. Rouillard, J. Okx, M. Pantzar, K. Paquel, R. Pederson, A. Peepson, F. Pelsy, D. Petrovic, E. Psaila, B. Šarapatka, J. Sobocka, A.-C. Stan, J. Tarpey, R. Vidaurre. 2016. 'Updated Inventory and Assessment of Soil Protection Policy Instruments in EU Member States.' Final Report to DG Environment. Berlin: Ecologic Institute.

Hedges, L. V., J. Gurevitch, and P. S. Curtis. 1999. The meta-analysis of response ratios in experimental ecology. Ecology 80: 1150-1156.

Hijbeek, R. 2017. On the role of soil organic matter for crop production in European arable farming. PhD thesis Wageningen University.

Humphrey, A.S. 2005. SWOT Analysis for Management Consulting. SRI Alumni Association Newsletter, December 2005, pp. 7-8. https://www.sri.com/sites/default/files/brochures/dec-05.pdf (last access: 10-June-2016).

Liebscher, G., (1895). Untersuchungen über die Bestimmung des Düngerbedürfnisses der Ackerböden und Kulturpflanzen. J. Landwirtsch. 43, 49–216.

Mazoyer M, L Roudart. 2006. A history of world agriculture: from the neolithic age to the current crisis. Earthscan, 512 pp.

Metzger, M.J., R.G.H. Bunce, R.H.G. Jongman, C.A. Mücher, and J.W. Watkins. 2005. A climatic stratification of the environment of Europe. Global Ecology and Biogeography 14: 549–563.

Wezel A, M Casagrande, F Celette, JF Vian, A Ferrer, J Peigné. 2014. Agroecological practices for sustainable agriculture. A review. Agronomy for Sustainable Development, 34, 1-20. <10.1007/s13593-013-0180-7>.

WOCAT. 2017. World Overview of Conservation Approaches and Technologies. https://www.wocat.net/

6 Soil-improving cropping systems for soil salinization

J. Cuevas, J.J. Hueso, F. del Moral, I. Tsanis and I. Daliakopoulos

6.1 Background

Salinization refers to the accumulation of water soluble salts in the soil, which causes a deterioration or loss of one or more soil functions (Daliakopoulos et al., 2016; van Beek and Tóth, 2012). Salt-affected soils can be classified as (i) saline soils with elevated salt concentrations, (ii) saline-sodic soils with a disturbed monovalent/divalent cation ratio in favour of the monovalent alkali cations (Na+, K+), and (iii) sodic soils with a chemical composition skewed towards alkalinity (high pH) often caused by a dominance of (bi)carbonate anions in solution. There are two groups of salinization driven by climate and human activities that lead to salinity, namely *primary salinization*, which involves accumulation of salts through natural processes due to the composition of the soils and nearby environment, and *secondary salinization*, a result of human intervention often due to mismanagement of agricultural land, especially ill-planned irrigation and drainage (Daliakopoulos et al., 2016; Apostolakis et al., 2016). Secondary soil salinization is of particular interest due to its effects and feedbacks relevant to agricultural systems.

Soil degradation resulting from salinity and/or sodicity is a major environmental constraint with severe adverse impacts on soil productivity, agricultural sustainability, and food security, particularly in arid and semi-arid regions of the world (Tanji and Wallender, 1990; Suarez, 2001; Pitman and Laüchli, 2004; Qadir et al., 2006a). In several large irrigation schemes, salinity-induced land degradation has increased steadily over the last few decades with concurrent reductions in agricultural productivity and sustainability. Currently, saline and sodic soils occur at least in 100 countries (Qadir et al., 2006a), covering in total 932.2 Mha (Rengasamy, 2006). Outside Europe, soil salinization hotspots include Pakistan, China, United States, India, Argentina, Sudan, and many countries in Central and Western Asia (Aquastat, 2017; Ghassemmi et al., 1995).

Europe has 30.7 Mha of saline and sodic soils, i.e. 3.3% of the global saline and sodic area (Rengasamy, 2006). Soil salinity is a major obstacle in coastal and low-lying areas (Li et al., 2012a; Manjunata et al., 2004; Sparks, 2003), and is a major cause of desertification in the Mediterranean countries. Along the Mediterranean coast, the problem of soil salinity is increasing due to scarcity of precipitation and irrigation with low quality water. Especially in the case of overexploitation of coastal aquifers that are hydraulically connected to the sea, seawater intrusion causes wide-spread soil salinity problems through groundwater irrigation (Daliakopoulos et al., 2016). Thus, saline soils here are present mainly due to human activities (Abu Hammad and Tumeizi, 2012; Domínguez-Beisiegel et al., 2013), and augmented by the extension of irrigation and poor drainage systems, eventually leading to 25% of the irrigated

agricultural land being affected at a significant level by salinization in the Mediterranean region (Geeson et al., 2003; Mateo-Sagasta and Burke, 2011). In addition, primary salinization due to seawater intrusion also affects regions near the coasts of Netherlands, Denmark, Belgium, France and England (Raats, 2014; Trnka et al., 2013; van Weert et al., 2009).

Secondary salinization is caused by an imbalance between rainfall and transpiration in dryland cropping systems (Cocks, 2001), but it is also linked to irrigated land where prevailing low rainfall, high evapotranspiration rates and soil characteristics impede soil leaching (Chesworth, 2008; Maas et al., 1985; Mateo-Sagasta and Burke, 2011). Other factors leading to soil salinization in semiarid regions are the raising of the water table due to filtration from unlined canals and reservoirs, uneven distribution of irrigation water, poor irrigation practices, land clearing, and improper drainage. Poorly drained soils allow for too much evaporation leading to salt residuals on the soil surface (Tsanis et al., 2015).

The introduction of irrigation in arid and semi-arid environments almost inevitably leads to water table rise and often to problems of waterlogging and salinization (Crescimanno and Garrofalo, 2006; Bhutta and Smedema, 2007). When watering schedules are not properly conceived, excess evaporation causes part of the soluble salts applied by irrigation to accumulate at the soil surface. In other cases, excess water due to over-irrigation, unlined canals and reservoirs, or vegetation clearing, in combination with inadequate drainage, filters into the groundwaters, from where the dissolved salts are remobilised to the upper layers of the soil by means of upward water flows during dry periods. The sustainability of irrigated agriculture in these regions could then be under serious threat due to recharge to saline groundwater leading to this secondary salinization. Provided that appropriate agricultural and irrigation strategies are followed, moderately saline water may be used for irrigating some tolerant crops. In some cases, this has been found beneficial to fruit quality and less often to yield (Oron et al., 2002; Pang et al., 2010). Nonetheless, there is a widespread acceptance that irrigation, without a well-engineered drainage to help halting secondary salinization, will be no longer sustainable (Ritzema, 2016). Under the prevalent arid conditions in the Mediterranean area, and their projected intensification and expansion (Huang, 2016; Daliakopoulos et al., 2017), the improvement of the management of irrigation water, crops, and nutrient inputs, and an increasing efficiency in the use of water and fertilizers will become indispensable to conserve and sustain the already fragile agricultural soils. The agricultural competitiveness of Europe is in danger if the salinized lands of coastal areas of the Mediterranean cannot be reclaimed for cultivation in the near future.

6.2 Purpose

The aim of the study reported here is to review literature related to soil-improving cropping systems that prevent, mitigate or remediate the impacts of human interventions on soil salinization processes. We focus on secondary salinization, and on saline and sodic soils. This chapter also reviews the different strategies and agronomic techniques used to ameliorate soil quality regarding salinity.

6.3 Results and Discussion

6.3.1 Concept

From a farmers' point of view, the ideal situation is the availability of irrigation water of good quality at an affordable price. If this objective is not reached and farmers have to deal with a serious problem of salinization, they adopt different cultivation techniques in order to minimize the economic impact on crops. Once the problem of salinization is well established, they either abandon land after several years of exploitation (Darwish et al., 2005; Kitamura et al., 2006) or they try to implement measures to minimize the impact of salinization. In this last group of measures, biotechnology and engineering can be of help.

Approaches to salinity management are often described as involving either recharge or discharge management. Recharge management is typically associated with avoiding or minimizing salinity by reducing net recharge to groundwater (planting tree species with high evapotranspiration demands is a commonly cited example). In contrast, discharge management tends to involve adaptive strategies, such as planting crops that are tolerant to saline soils, engineering solutions to reduce salt entering streams, and remediation of saline soils (Finlayson et al., 2010).

Four strategies have been proposed by Qadir et al. (2006b), each one composed of different measures, as a remedy to halt secondary salinization of agricultural lands. The first strategy combines different agronomic techniques and aims at minimizing the negative impact of salinity on yield and fruit quality of the main crop. The second strategy pursues the use of irrigation water of better quality and the prevention of further salinization by an improvement of drainage. The third strategy is the planting of halophytes plants in order to extract salts from the aquafers. Finally, Qadir et al. (2006b) proposed the mechanical removing of salts from the soils.

The best solution to dealing with the twin menace of salinity and saturation is the drainage of a net flux of salt away from the root zone, and to control the water table height. Applying adequate water of reasonable quality as leaching fraction along with installing adequate drainage network are then proposed as the most sustainable and affordable solutions to prevent salinization when sources of water of good quality are available. The use of desalinized water is an effective solution, but the costs are often too high for many farmers. Mixing water of different sources (desalinized and well water) could then be a partial solution for moderately tolerant crops.

Once the problem of salinization is well-established, farmers may adopt different management decisions to minimise the impact of salinization. Among them, we may select the management of irrigation and fertilisation as the most crucial techniques for halting further degradation of soils. Selection of crops tolerant to moderate levels of salts can be considered as far as the cash provided by their cultivation does not lessen their acceptance by farmers.

Biosaline agriculture seems, so far, the last resort when the soil accumulates in the root zone large amounts of salts. Although there is a growing interest in developing crops able to tolerate high levels of salinity, very few of these crops can be considered cash crops of high interest for growers. As we show below, in the near future, we may expect some help from biotechnology for reclaiming salinized lands. Screening germplasm of halophytes plants could be of interest for recovering lands heavily affected by the problem of salinization. The use of tolerant rootstocks and varieties is thought to increase in the years to come. Transgenic crops have also been proposed as a solution to deal with salinized soils. Significant progress is expected from classical breeding too.

Measures proposed for halting salinization of agricultural lands should consider farmers' willingness to adopt them. Stakeholder-inclusive decisions in fighting soil salinity is gaining consideration (Panagea et al., 2016). We firmly believe that the salinization process can be halted with effective policy decisions through stakeholder engagement. Policy could play a key role in preventing and remediating salinization of soils (Bai et al., 2015).

In summary, three groups of SICS remedial measures can be distinguished: (i) Preventing or halting secondary salinization, (ii) Dealing with salinization, and (iii) Reversing salinization. These SICS are further discussed below.

6.3.2 Preventing or halting secondary salinization

The basic concept to prevent secondary salinization is to maintain the overall salt balance in an irrigated area by draining off the excessive salt, which is equivalent to the amount of salt inflow introduced to the area with irrigation water. For keeping such a balance, effective and sustainable procedures for draining salts applied by irrigation are required, while efficient methods are equally needed for diminishing the amount of fertilizer salts applied avoiding their upward flow to the upper soil horizons.

Leaching

Leaching refers to the practice of applying an extra amount of water of reasonable quality and beyond crop requirements to prevent salts from building up in the soil. Leaching is considered one of the main basic management tool for controlling salinity (Crescimano and Garofalo, 2006). Leaching can contribute to reduce soil water salinity by discharging salts from the upper horizons to the lower soil layers. The strategy is to keep the salts in solution and flush them below the root zone. The amount of water needed is referred to as the leaching requirement or the leaching fraction. However, excess irrigation also increases the leaching of nutrients and other agrochemicals (Gabriel et al., 2014), and often reduces the water quality of the receiving water bodies (Wichelns and Oster, 2006; Castanheira and Serralheiro, 2010). In addition, it reduces water and nutrient use efficiency (Díez et al., 2000), and contribute to groundwater contamination.

Farmers can mitigate the effect of salinity on profit margins managing the level of salt in the soil through leaching (Young, 2005). Under limited water supply conditions, the farmer needs to decide between fully allocating the available volumes of irrigation water in order to plant the maximum area, with reduced crop yield per hectare, or reducing the crop area and thereby releasing some water for leaching purposes, which will increase the crop yield per unit area. Modelling results suggest that it is more profitable to leach excess salt from the soil once salinity levels exceed the crop's salinity threshold (Matthews et al., 2010).

Seasonal analyses of groundwater salinity have revealed that the highest electrical conductivity value is observed in summer (Abliz et al., 2016), when rainfall inputs are lower and evapotranspiration increases. In relevance to cropping systems, this could mean that earlier summer crops and/or picking fruits from only the first few nodes and then stopping cultivation, or winter cultivations (open air or under plastic) may be beneficial. The first conclusion is also reached by Daliakopoulos et al. (under review) for a tomato (*Solanum lycopersicum*) crop.

Soil salinity can be diminished by pre-season salt leaching using high amounts of water. Forktutsa et al. (2009) observed that leaching did not efficiently remove salts from the 2 m profile, at two out of three experimental plots. Instead, salts were only shifted from the upper (0-0.8 m) to the lower (0.8-2 m) soil layer. Even worse, in their experiments, strong groundwater contribution to evapotranspiration triggered secondary re-salinization of topsoil during the next cropping season. Consequently, salt amounts in the top 0.8 m of soil increased from 9 to 22 t/ha in a field with loamy texture, and from 4 to 12 t/ha in a field with sandy texture. Simulations confirmed that present leaching practices are hardly effective, and that complementary techniques have to be put in practice.

The use of rain water to leach salts from soil has been proposed as a win-win strategy to control soil salinization in protected cultivation as well as in open field. Ashraf and Saeed (2006) describe the use the monsoon rain water to leach the excess of soil salts after a maize (*Zea mays*)-wheat (*Triticum aestivus*)-dhanicha (*Sesbania aculeate*) crop rotation. Panagea et al. (2016) surveyed protected cultivation farmers' opinion to know which remedial measures for salinization are more easily to be adopted. Rainwater harvesting from greenhouse roofs was clearly the best strategy for farmers, because of their willingness to adopt this measure. Harvested water can be used for irrigation purposes either on its own or mixed with water of poorer quality, or employed for leaching. When at least two qualities of water exist for irrigation, cyclic irrigation, with intermittent leaching fractions, can be employed to keep salinity under levels not affecting crop productivity. Crescimanno and Garofalo (2006) indicated that this strategy can be more effective at leaching salts than continuous leaching (imposing a leaching fraction at each irrigation event).

Drainage

Where leaching is practiced, drainage must be enhanced to carry away the excess (salty) drainage water and therefore preventing or reducing the upward flow of salt. For leaching to

be more effective, an adequate drainage network is needed. Regardless the necessary environmental consideration and of those regarding the water availability, improving the efficiency of the drainage system to lower the groundwater table would be more effective than leaching. Kobt et al. (2000) considered, therefore, that governmental efforts should focus on facilitating the improvement of field-drainage conditions through the installation of subsurface drainage systems.

Since the drainage and salinity control of an irrigated area by only an open drain system poses limitations, installation of subsurface tile drainage is recommended for supplementing the open drain system. In this regard, a tile drain system can enhance the drainage efficiency and effectively remove accumulated salts from the root zone. Kitamura et al. (2006) suggested to install subsurface tile drainage for the management of drainage outfall and thus minimize environmental degradation caused by saline drainage water in downstream area. In case a tile drainage system is installed in an area with high salt accumulation, the danger of deterioration of the water quality downstream is a possibility due to the outflow of increased amounts of accumulated salts from the area. Kitamura et al. (2006) recommended that each irrigated area should be well equipped with a special pond at the end of the drainage system to control the quantity and quality of drainage water to be drained off to the downstream. It is desirable to develop a design and management technique of an evaporation pond for better effluent management at the outfall of each irrigation block. This measure allows to monitor groundwater quality evolution. Konukcu et al. (2006) proposed "*dry drainage*" as a partial solution for groundwater salinization, i.e., dedicating part of the land as a sink for the excess groundwater. The reasoning behind this strategy is that, if inflow (rainwater excess, field application losses, watercourse and/or canal seepage losses) within a given area balances outflow (supply to crops from water table, evaporation from uncropped areas, artificial and/or natural drainage sinks), then the water table will be stable. If the uncropped area is large enough and evaporation from this area is fast enough, then the necessary balance can be achieved without artificial tile drainage. This is the concept of "*dry drainage*". It means that part of the available land is set-aside as a sink for excess groundwater and for the salt transported with it. Given that under current uncertainties in water supply many farmers partly rely on groundwater to avoid crop failure, improving the drainage system without improving irrigation scheduling seems not an advisable strategy. A severe reduction in crop growth and yield would be the likely consequence (Forkutsa et al., 2009).

"Bio-drainage" involves growing certain types of plants that draw their main water demand directly from the canal seepage water or the capillary fringe immediately above it (Heuperman, 1999; Bhutta and Chaudhry, 2000). This consumption may help maintaining the groundwater table at a safe level (and thereby prevents the saturation of the top 2 m of soil). The success of *bio-drainage* depends on the soil texture (Vlek et al., 2002). The growing of poplars (*Populus spp.*) and tamarisks (*Tamarix gallica*) is applicable here (Bhutta and Chaudhry, 2000; IPTRID,

2002). However, some authors question the efficiency of bio-drainage due to its limited effects (Morris and Collopy, 1999; Slavich et al., 1999).

Preventing upward flow of salt. Lowering water tables.

The prevention of upward flow of salt is also achieved by lowering water tables. Percolation control methods, such as puddling and subsoil compaction (Yamazaki, 1976; Sharma and Bhagat, 1993), have been considered as measures to prevent the upward flux of dissolved salts.

Lowering water tables may also bring about some decreases in surface soil salinity. Different strategies can be put in practice with this goal in mind. Re-vegetation with deep-rooted salt-tolerant species can produce a partial and short-term rehabilitation of salt-affected land. The problem is that, after the initial decrease in the water table, the hydraulic gradient towards the root zone increases and thus does the intrusion of groundwater. Wet (sub)soils impair the efficiency of roots to salt exclusion, so that there is an increased transport of sodium and chloride to the shoots, which damages leaves. Eventually, salt concentrations in root zone reach levels that substantially decrease the availability of water to the plants. Consequently, there is a reduction in LAI, a decrease in transpiration and the water table begins to rise back, often to their initial levels. One undesirable consequence of the use of groundwater by perennial plants is a long-term accumulation of salt in the root zone. Fully understanding this process is fundamental to the development of sustainable agricultural systems and the management of saline land (Barret-Lennard, 2002).

In this regard, Australia has an extensive secondary salinity problem caused by the replacement of native vegetation composed of deep-rooted perennials with shallow rooted annual species that causes a consequent rise in water tables and brings salt stored deep in the profile to the soil surface. The replacement of the native perennial evergreen vegetation by winter–spring active, annual crops and pastures, and by inactive fallows, changes the hydrological balance increasing the drainage through the soil profile. Salt is then leached more deeply in the profile and water tables gradually rise in the lower parts of the topography bringing salt to the surface of those low areas. The initial effects of this redistribution of salt in the soil can be seen in the changes in the composition of species in pastures and in the poor growth of crops. The final effect is bare, erosion-prone land, and salinized streams (Connor, 2004). This can be seen also in the pampas of Argentina.

One solution to salinity lies in the reintegration of trees and other perennial species back into the agricultural systems with the aim of returning hydrological function to a condition that mimics that of the original landscape (Barret-Lennard, 2002). Re-vegetation can also reduce in some extent soil salinity by the direct uptake of salt if the proper species is chosen.

The effects of cropping systems on the progression or reversion of salinization of agricultural land are summarized in at the end of this chapter (Figure 6.3).

6.3.3 Dealing with salinization

Given the role of over-optimal irrigation in aggravating the problem of salinization, it is necessary to correct farmers' attitude toward the value of water. Irrigation water is still cheap in many countries, which does not encourage water conservation by farmers (Anderson, 1997; Zhou et al., 2015). Some progress has been made in water pricing, especially in some arid and semi-arid areas where water is a very scarce and a valuable resource (Alcon et al., 2014).

A second approach to deal with already established secondary salinization of agricultural lands is to improve field management including irrigation, fertilization, tillage, and crop selection. Minimizing the impact of salinization on the crop yield and fruit quality often lies in the hand of the manager. Therefore, upgrading farmers' skills to deal with salinization is required.

Water management

Water management and drainage are key to deal with salinization. Even the use of water with moderated salt content can lead to an increase of the salt content of the soil under semi-arid conditions, especially on soils with low saturated hydraulic conductivity (Mendes and Carvalho, 2009). Therefore, more efficient irrigation through a modification of watering schedules and a decrease of watering consumption must be accomplished to halt the process of soil degradation. Different procedures linked to the system and timing and dose of irrigation are available.

Flood irrigation is a common cause of water table rising, especially in clay and loamy soils, where deep drainage is impeded, and salts present in groundwater ascend to the upper soil layers. Under conditions of aridity, these soils might drive a process of salinization even using water of good quality for irrigation. Hence, flood irrigation should be forbidden in these cases.

Microirrigation has several advantages when the irrigation water is saline. Except for low growing crops irrigated with microsprinklers or sprayers, microirrigation avoids wetting of the leaves with saline water that causes damages in the green tissues. Because microirrigation is normally applied frequently, there is also a continuous leaching of the soil volume from which the plant extracts the water. Leaching can also be provided intermittently, between growing seasons, and by seasonal rainfall when soil salinity in the root zone is maintained below detrimental levels (Hoffman and Shannon, 2007). Drip irrigation might increase the risk of salinization of upper soil horizons (Marchand and El Hadi, 2002), but prevent salt leaching to groundwater when compared to flood irrigation. However, Hanson et al. (2008) demonstrated that the wetting pattern around emitters results in higher leaching fraction and lower salinity levels than in other irrigation systems for a given amount of applied water. Drip irrigation commonly uses lower irrigation volumes and allows heavier yields, reaching then higher water efficiency. Subsurface drip irrigation has been proposed as additional measure since reduces evaporation from the soil (Hanson et al., 2008), and distributes moisture distribution better adjusted to the root pattern in comparison to conventional drip irrigation (Oron et al., 2002).

More efficient irrigation schemes may nevertheless be tuned on the limit of deficit irrigation (see below) and plant salt tolerance, because lower leaching may cause increase of soil salinity.

Deficit irrigation (DI) consists in the application of water below full crop water requirements, so that a mild crop water stress is allowed with bearable effects on yield. The three most common deficit irrigation strategies are: (1) regulated deficit irrigation (RDI), where water deficit is applied at certain developmental stages, (2) partial root-zone drying (PRD), where alternatively half of the root system is fully wetted while the other half is allowed to dry, and (3) sustained deficit irrigation (SDI), where water deficit is uniformly distributed over the whole crop cycle.

Cuevas et al. (2007) have demonstrated that even the imposition of severe levels of water stress in selected phenological stages (RDI) can bring economic benefits to loquat (*Eriobotrya japonica*; a subtropical fruit tree sensitive to salinization) producers not only by reducing irrigation costs, but also by increasing the value of the crop. The application of postharvest regulated deficit irrigation for more than 10 years in the same loquat plots confirm the sustainability of such irrigation strategy (Hueso and Cuevas, 2010). Limited water resources in semi-arid areas of the Mediterranean coast suggest the adoption of this strategy. In this regard, fruit crops, among them loquat and table grape (*Vitis vinifera*), are frequently subjected to regulated deficit irrigation with water of poor quality allowing higher profits if appropriate selected phenological phases are targeted (Hueso and Cuevas, 2008; Pinillos et al., 2016). It is important to consider, however, that DI with saline water may impose additional stress on the plants.

Deficit irrigation strategies save water, but also have the potential to improve soil salinity management by a better control of rising water tables and by a reduction in the import of salts by irrigation water. Nonetheless, deficit irrigation does not provide the same degree or leaching than full irrigated conditions so, it may enhance soil salinization when agricultural plots are irrigated with low-water quality (Aragüés et al., 2014b). Although deficit irrigation in combination with drip irrigation technologies could leach salts away from the root domain in a very efficient way (Hanson et al., 2008), a potential risk for some types of deficit irrigation management emerges during the periods when irrigation is interrupted, because the leaching fraction could be insufficient to displace the salts from the active root zone of the crops (Aragüés et al., 2014a). Increasing irrigation efficiency would help sustaining the present crop production levels while reducing future leaching demands (Forkutsa et al., 2009).

Given the central role irrigation plays for causing secondary salinization, a more drastic alternative is the conversion of irrigated land to rain-fed production systems. This conversion reduces yield and fruit quality (size especially), and limits farmers profits, so it is likely to be adopted only when other strategies for fighting salinization have failed. It often requires changing crops and selecting drought-tolerant ones. Once this decision is adopted, measures

can be taken to maximize the effectiveness of precipitation through adopting tillage practices to reduce water losses and crop demands.

English et al. (2002) argue that irrigation based on economic efficiency principles will be the new paradigm that will govern irrigation management in the light of the limited water supplies that are threatened by deteriorating water quality. Economic efficiency requires the decision-maker to explicitly consider costs, revenues and the opportunity cost of water in his decisions.

Nutrient management

One obvious approach to deal with salinization is limiting the amount of fertilizer salts applied. This implies correct fertilization programs, with fertilisers and doses correctly chosen and adjusted. Many authors have emphasized the importance of reducing fertiliser applications to prevent further salinization of groundwaters. Not only the diminution of the amounts of fertilisers applied may contribute to halting the process of secondary salinization, but also does the implementation of fractional fertilization programs (González-Vázquez et al., 2005). Combining appropriate management of water and nutrient input, through fertigation, provides the key for sustainable agriculture in dry areas. Judicious fertigation allows water saving, reduction of pollution hazards caused by fertilizers, reduced production cost and higher net return for the farmer. The higher use efficiency of water and fertilizer makes economic and environmental sense. Properly choosing the type of fertilizers and managing the nutrient placement in the root zone with drip-fertigation can substantially reduce the risk of salt leaching to groundwater. Improving the management of water and nutrients inputs and increasing their use efficiency is a necessary step to conserve the limited natural resources in arid and semi-arid areas (Darwish et al., 2005).

The choice of fertilizers is another key factor for limiting salinization. The results of many experiments located in areas prone to secondary salinization clearly show the detrimental effect of potassium chloride on yield through an accumulation of salts in the soil, while potassium sulphate has less impact on soils (Marchand and El Hadi, 2002).

Soil management

Soil management includes practices adopted to reduce the amount of irrigation required for soil water conservation and use for crops and weeds. It includes tillage, mulching, and direct drilling.

Different studies have revealed that reducing soil evaporation by a surface residue layer would notably decrease secondary soil salinization (Forkutsa et al., 2009). In this regard, straw mulching is a promising option for farmers to control soil salinity, as it decreased soil water evaporation, and regulated soil water and salt movement (Tian and Lei, 1994; Pang and Xu, 1998; Pang, 1999; Li and Zhang, 1999; Li et al., 2000; Huang et al., 2001; Deng et al., 2003; Qiao et al., 2006). Pang et al. (2010) found a significant decrease in salt content within 0–20, 20–40

and 0–100 cm soil depths when straw mulching was implemented. Straw mulching decreased the salt content of the surface soil through regulating salt vertical distribution, which could reduce the degree of salt damage to crops, enhance crop yields and reduce the risk of soil salinization and erosion.

Hira and Thind (1987) proposed a plantation technique for Eucalyptus trees in salt-affected, non-irrigated plots, using mound and ridge covered with 1 m polythene sheet of 100 µm thickness applied on the ridge and covered with 5-6 cm of soil layer (Figure 6.1). Tree plantation was done on 15 cm diameter hole made in the centre of the sheet in a way that rain water can move radially and deep into the soil profile. This technique improved the establishment and growth of eucalyptus in comparison with flat plantations and mound and ridge not covered, and reduced the soil electrical conductivity by about 50%.

The combination of direct drilling and mulching represents a better way to improve infiltration and reduce evaporation from the soil, and to diminish salt accumulation during summer and improve leaching during winter (Figure 6.2). Direct drilling together with cover crops can be a useful combination in semi-arid conditions, in soils with low values of hydraulic conductivity, even when using water with moderated electrical conductivity (Mendes and Carvalho, 2009).

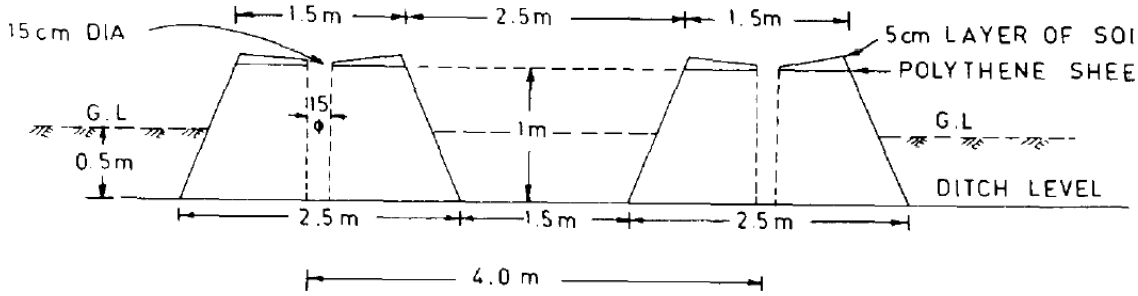


Figure 6.1. *Diagram showing the construction of ridge and hole and application of polythene sheet (Hira and Thind, 1987).*

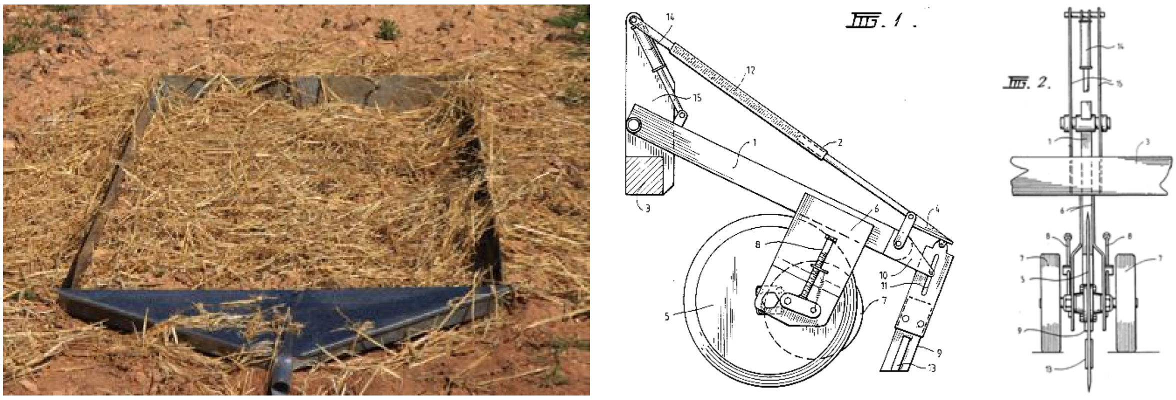


Figure 6.2. *Left: Detail of straw mulch application on an experimental erosion plot, Canals, Valencia (Photo: A. Cerdà). Right: Direct drill stump jump seeder patent EP0506661B1 filed in 1990.*

Crop rotation.

In the evolution of cropping systems seeking to halt salinization, a significant change has been made towards crop diversification and the replacement of long fallow by crop rotation. One option in this context is to include summer active perennials in the cropping systems. The major herbaceous option is *Medicago sativa* (lucerne) that fits well into a wheat–sheep system, providing valuable summer feed as well as additional transpiration (Connor, 2004).

It is well-established that cropping systems that rely on long fallowing for soil moisture conservation are sub-optimal because it favours the raising of water tables. Greiner (1997) recommended to increase cropping frequency as it raises farm income and reduces recharge to groundwater. Unless trees have commercial value, tree planting is neither a favoured option. Sustaining the productivity of the Liverpool Plains in Australia is an issue of reducing recharge to the groundwater system by changing land-use practices (Greiner, 1997). The first approach applies the concept of growing deep-rooted crop (crop consuming water from soil and shallow water table) over an area equivalent to the recharge of a unit area of rice in order to maintain the total water balance. The second approach involves achieving a whole farm water balance for average and for wet climatic conditions to find out optimum cropping pattern to minimise recharge from rice based system. Results from the second approach revealed that it is possible to control net recharge under above average rainfall using a suitable whole farm cropping mix (Khan et al., 2007).

The need to use more perennials in the different cropping systems has been identified. Several genera are likely to be of value in this respect, although few will be as widely adapted as lucerne. Cocks (2001) suggests targeting plant genera growing in dry Mediterranean areas. These may include perennial species of the family Fabaceae such as Astragalus, Hedysarum, Lotus, Onobrychis, Psoralea, and Trifolium. These plants have to match water-using and

nitrogen-fixing capacities of lucerne. Farming systems that can make full use of the new germplasm will be required too (Cocks, 2001).

Gabriel et al. (2014) analysed the impact of replacing long fallow by barley (*Hordeum vulgare*) and vetch (*Vicia villosa*) cover crops, on water, nitrogen, and salinity dynamics of a maize cropping system. The results obtained in field experiments and in simulation models show that the replacement of fallow with cover crops can be effective in reducing nitrate leaching, without increasing soil salinity or, even better, reducing top layer salinity. Moreover, the reduction of net salt loss observed in the plots with cover crops, compared with fallow, allows to limit irrigation volumes for salt leaching, as well as reducing the risks of deep water contamination by nitrates. Continuous cropping systems, incorporating legumes into the rotation, could make thus a significant contribution to restoration of salt and water balances and prevent or reverse salinization.

Weaver et al. (2013) quantified drainage water quality in the subsoil of sodic and non-sodic Vertisols under selected crop rotations, viz. continuous cotton (*Gossypium hirsutum*), cotton-dolichos (*Lablab purpureus*) and cotton-wheat. Their result show that the salinity of the drainage water were many times higher than those of irrigation water. Salinization of shallow groundwaters under irrigated cotton in Vertisol soils is, therefore, a clear possibility. Salinization of the root zone may occur in cotton-based rotations that result in poor subsoil structure and, thus, limited drainage even when irrigated with water of reasonable quality. To underline the importance of rotations, Cao et al. (2004) emphasize that soil salt content more than doubled where a paddy rice (*Oryza sativa*)–wheat (or oilseed rape; *Brassica napus*) cropping system was converted into intensive cultivation of vegetable crops. However, the continuous growing of the same vegetables in the same soil can result in the accumulation of autotoxin, and secondary salinization.

A completely different option is the use of mixed systems composed of herbaceous annuals and woody perennials, preferably fruit trees. This experimentation is interesting because these mixed communities of perennials and annuals with distinct root characteristics and seasonal dynamics, offer a range of competitive and complementary interactions. Deep-rooted, summer active perennials dry the soil to depth providing a large but horizontally discontinuous storage buffer (Lefroy and Stirzaker, 1999).

Salinity tolerant crops and rootstocks

In many areas where irrigation is necessary for crop production, salinization of soil seems unavoidable. Therefore, to guarantee the continuation of crop production in such areas, growing crop species with threshold of yield reduction well above the salinity of the irrigation water is needed. Recent trends and future projections suggest that the need to produce more food and fibre for the expanding population will lead to an increase in the use of salt-prone water and land resources for crop-production systems, and this will be met by using salt-

tolerant crops (Khan et al., 2009; Yensen and Biel, 2006). A distinction is made between plants able to tolerate only low levels of salinity (glycophytes) and those really adapted to saline soils (halophytes). Here we may include the cultivation of tolerant crops for reclaiming salinized soils, however, most crops are glycophytes and able to withstand only moderate levels of salinity, and only a few can be considered halophytes. Among herbaceous crops, we may cite rye, canola, guar (*Cyamopsis tetragonoloba*), wheat, kenaf (*Hibiscus cannabinus*), barley, and cotton, among vegetables purslane (*Portulaca oleracea*) and artichoke (*Cynara cardunculus*), and among fruit trees, guava (*Psidium guajava*), guayule (*Parthenium argentatum*) and different genera of palms (FAO, 2002). Recent studies in the few coastal subtropical areas of Europe have shown that pomegranates (*Punica granatum*), olive trees (*Olea europaea*), grapes, and mango (*Mangifera indica*) can also be considered as moderately salt tolerant (Chartzoulakis, 2005; Paranychianakis and Chartzoulakis, 2005; Zuazo et al., 2004). Some flexibility or adaptability in salt tolerance may be expected depending on soil properties, types of rhizobacteria, growth stage, and agronomical practices including salt-resistant rootstocks (Daliakopoulos et al., 2017). Therefore, the investigation of soil improving cropping systems and the isolation of salt-tolerant species, salt-tolerant genotypes and symbiotic biological agents are currently in the focus of international research projects to reduce yield losses under saline conditions (Cabot et al., 2014; Koubouris et al., 2015; Roy et al., 2014).

Based on the salt tolerance of plant species, there are emerging examples of plant diversification and management for the optimal utilization of salt-affected soils and saline-sodic waters. The plant species that have shown potential under such environments are divided into five groups: 1) fibre, grain and special crops; 2) forage grass and shrub species; 3) medicinal and aromatic plant species; 4) bio-fuel crops; and 5) fruit trees. An appropriate selection is generally based on the ability of plant species to withstand elevated levels of soil salinity while also providing a marketable product or one that can be used on-farm; however, from an economic perspective, much depends on the local needs (Wang et al., 2015). Whatever existing products are contemplated, viz. timber, cellulose, biomass energy, fruit, essential oils etc., they will have to compete with production elsewhere, commonly grown in more favourable conditions. Connor (2004) addressed an interesting question: "*what opportunities exists to include olive as a component of these cropping systems*"? Comparable mixed-cropping systems are already a feature of Spain and other Mediterranean countries. Like eucalypt, olive is evergreen, summer active, and drought resistant, but unlike eucalypt, olive has the advantage of producing very valuable edible oil. More details can be found in the studies of Qadir and Oster (2004) and Qadir et al. (2008).

A variety of practices, including grafting with tolerant rootstocks, microbial agent application and plant modification, has been used to improve the soil quality and enhance crop growth in protected vegetable production systems (Sun et al., 2014). The utility of rootstocks to combat biotic and abiotic stresses in fruit crop production is well-known since the antiquity, probably starting about the beginning of the first millennium (Mudge et al., 2009). Actually, Plinius the

Elder documented its use in the Old Greece in his Natural History. In Mediterranean fruit crops tolerant to salinity such as olive, pomegranate or fig (*Ficus carica*), this use of rootstocks is still negligible; however, in temperate-zone and subtropical fruit trees, rootstocks tolerant to salt represent an excellent tool for their cultivation in degraded soils. More recent is the interest in using the same approach for vegetable production in salty soils or vegetables irrigated with poor quality water, especially in Cucurbitaceae and Solanaceae. Given the success obtained in fruit crops and the promissory results obtained so far (Colla et al., 2010), a growing use of grafting in vegetable production is largely expected. Independently of the chosen crops, there are varieties better adapted to salinization because their phenology allows them to avoid critical periods. It is thus essential to develop varieties that are phenologically capable of sustaining excess salt throughout its life span and produce high yields (Vinod et al., 2013). The use of microorganisms (Trichoderma harzianum isolate T78 and Pseudomonas stutzeri) also has been proposed as a mean to enhance soil microbiological diversity and mitigate salinity effects on plant growth and soil quality (Bacilio et al., 2016; Daliakopoulos et al., under review; Mbarki et al., 2017).

A possible controversial solution may come from transgenic modified plants. Previous work has suggested the capacity to enhance salt tolerance of staple food crops by inserting transgenes in them. Abebe et al. (2003) showed already that ectopic expression of the mtlD gene from Escherichia coli implicated in the biosynthesis of mannitol improves wheat tolerance to water stress and salinity. This same gene is effective in poplars trees (Hu et al., 2005). It has been shown that plant height and plant weight increased for transgenic potato plants under NaCl and polyethylene glycol stresses compared with the control potato plants when betaine aldehyde dehydrogenase gene from spinach was introduced in them. These results indicate that transgenic plants better tolerate salinity (Zhang et al., 2011).

Less controversial is the use of wild relatives for enhancing salt tolerance in crops and the utilization of their ability by conventional breeding. Colmers et al. (2006) analysed the underlying mechanisms of salt tolerance of several halophytes genera of the tribe Triticeae and possible to hybridize several wild species with durum and bread wheat. Similar approaches have been suggested for wheat and the more salt tolerant barley, searching into the germplasm of *Triticum dicoccoides* and *Hordeum spontaneum*, the progenitors of cultivated wheat and barley (Nevo and Chen, 2010).

The effects of the above-mentioned soil improving crop systems on agronomic, economic, and environmental aspects regarding the salinization of soils are described at the end of this chapter (Figure 6.3).

6.3.4 Reversing salinization. Removing salt from the system

There are tolerant plants capable to live in saline soils. The natural existence of this kind of plants, true halophytes, could be useful for reducing the amount of salts present in the root zone. However, plants cope with the problems of salinity in various ways, some of them avoid

salinity by completing their cycle when salinity is lower (rain periods), some other resist salinity, and a few others tolerate salinity, being useful for reversing salinization. These latter plants accumulate salts in their cells and/or secreting it through specials organs. The idea implies the later disposal of the above ground material and the continuous growing of them in order to reverse salinization levels and reclaim salinized lands.

Biosaline agriculture is a relatively new way of dealing with salinity in agriculture. It develops cropping systems for saline environments, using the capacity of certain plants to grow under saline conditions in combination with the use of saline soil and alternative water-resources. Biosaline agriculture requires improved soil and water management, but most importantly new genetic resources (new genotypes or more salt-tolerant species). High global variation in salinity, availability and ionic composition of saline water and soil conditions renders any single plant unsuitable for all systems.

Biosaline agriculture is becoming a reliable strategy for using saline environments. The first patent for a halophyte crop was issued less than 20 years ago, and at present, crops are being developed by classical breeding, biotechnology, tissue culture, and plant exploration (Yensen and Biel, 2006; Qureshi et al., 2007; Rabhi et al., 2010). There is a special interest on the production of halophytes using saline waters and soils in desert ecosystems and feeding them to livestock (Kafi et al., 2010). Kochia (*Kochia scoparia*) is a salt- and drought-tolerant species, an annual plant of the family Chenopodiaceae, which can be a valuable source of fodder. The results reported by Kafi et al. (2010) suggest that Kochia may be a candidate species for cultivation in areas where salinity cannot be diminished to acceptable limits by leaching or other salinity-management techniques. This plant has a high potential to grow on soils under irrigation with saline water in summer. Kochia can produce considerable dry mass, and up to a 20% reduction in its water requirements has no significant effect on its fodder production.

Azhar et al. (2015) compared base scenario keeping land and crops managements as usual with bioremediation by growing salt tolerant fodder such as Sudan grass (*Sorghum drummondi*) and Berseem clover (*Trifolium alexandrinum*) and with the strategy of optimum land allocation with different favourable crops. The results show that in comparison with a base scenario, bioremediation techniques are helpful in reducing the salt balance of the crop root zone in the long term, while crop allocation was found to be effective as a short-term solution, but was less effective on reducing salt content in the soil. Despite these encouraging experiences, the effects of growth of halophytes on soil salinity are likely to be minimal according to Barret-Lennard (2002).

The last strategy is the mechanical removal of salts from the surface of the soil using appropriate machinery (Qadir et al., 2006b). It consists of surface flushing or mechanical removal of salts from salt crusts at the surface, that could be the last solution where drainage is inadequate, and leaching is restricted by the presence of a shallow water table or highly impermeable profile. This strategy seems rarely affordable.

Table 6.1 provides an overview of all agro-management techniques that were studied in 30 studies. Some studies reported on more than one treatment, so that in total 123 treatments were studied. Studies are grouped based on the type of treatment used and may therefore appear more than once on the list.

Table 6.1. *Tested agro-management techniques on soil salinity in the references studied in this chapter. A: amendment (chemical, other than conditioner); M: mulching; C: conditioner (biological); D: drainage; Fe: fertilization; Fl: flushing; I: irrigation; P: phytoremediation; R: rotation instead of mono cultivation; T: tillage.*

SICS	Details	Reference
A	Gypsum	Qadir et al., 1996
A	Gypsum	Ahmad et al., 2013
A	$FeSO_4.7H_2O$	Mahdy, 2011
A	Gypsum, H_2SO_4, citric acid, and polyvinyl alcohol	Adnan et al., 2014
A	Gypsum and H_2SO_4	Ahmad S. et al., 2013
A+M	Gypsum in combination with various straw types	Ahmad M.J. et al., 2013
C	Combinations of compost, anthracite coal powder, and water treatment residuals	Mahdy, 2011
C	Sewage sludge, epicarp-mesocarp of almonds	Pedreño et al., 1996
C	Combined application of manure and humic acid	Shaaban et al., 2013
C+A	Combined application of gypsum, manure, and humic acid	Shaaban et al., 2013
C+A	Combination of compost, anthracite coal powder, water treatment residuals, and $FeSO_4.7H_2O$	Mahdy, 2011
D	Improved drainage (modelled)	Forkutsa et al., 2009
D	Subsurface drainage	Sharma et al., 2006
D	Different types of subsurface drainage systems	Ritzema et al., 2008
D	Subsurface drainage	Satyanarayana et al., 2003
Fe	Application of Potash fertilizers	Marchand & Abd El Hadi, 2002
Fl	Horizontal surface flushing	Nayak et al., 2008
I	Conjunctive use of saline/non-saline irrigation	Kaur et al., 2007
I	Different types of deficit irrigation	Aragüés et al., 2014a
I	Different types of deficit irrigation	Aragüés et al., 2014b
I	Alternate furrows and bed and furrow	Ashraf & Saeed, 2006
I	Different types of deficit irrigation	Aragüés et al., 2015
M	Residue layer	Forkutsa et al., 2009
M	Palm leaves or plastic	Al-dhuhli et al., 2010
M	Straw mulching at different rates	Pang et al., 2010
M	Polyethylene, pine bark or jute fibers	Aragüés et al., 2015
M	Winter wheat straw	Bezborodov et al., 2010

SICS	Details	Reference
M+I +D	Residue layer, improved drainage and optimized irrigation	Forkutsa et al., 2009
P	Forage cultivation	Qadir et al., 1996
P	Various crops during the fallow period	Ado et al., 2016
P	Forage cultivation	Ahmad et al., 1990
P	Tomato in consociation with halophytes	Zuccarini, 2008
P	Phytodesalinized soil	Rabhi et al., 2010
R	Conversion to continuous vegetable cropping	Cao et al., 2004
R	Various crops during the fallow period	Li et al., 2012b
R	Various crops during the fallow period	Gabriel et al., 2014
T	Zero tillage	Young et al., 2014
T	Intermediate tillage or permanent bed planting	Pulatov et al., 2012
T	Minimum tillage	Pang et al., 2010
T	Direct drilling	Lozano-García et al., 2011

For all 123 treatments the effect of the agro-management techniques treatment was expressed in a change in soil salinity, either expressed in Electrical Conductivity (EC) levels [dS m^{-1}], Sodium Adsorption Ratio (SAR; [(mmoles $L^{-1})^{0.5}$)]), in total salt content (%) or Cl^-/Na^+ ion concentration [meq/100 g]. Figure 6.3 presents the effect size, ES (%), as the change in salinity due to the SICS treatment relative to the reference case: a negative value for ES indicates a decrease (in this case an improvement) of the soil salinity level. Yield is expressed as t ha^{-1}, g $plant^{-1}$ or g $container^{-1}$ depending on experimental design.

Most notable results for Figure 6.3 are:

1. Soil amendments (chemical) typically reduce salinity but have a great variability in yield increase (may also cause a decrease in yield).
2. Mulching, alone or in combination with amendments generally preserve productivity (but have also caused decrease in certain applications) with very satisfactory salinity reducing effects.
3. Biological soil conditioners are increase yield while reducing salinity and may act better in combination with other soil amendments.
4. Drainage increases yield but it’s effect on soil salinity depends on location and procedure.
5. Fertilization increases salinity and yield.
6. Flushing is always advisable.
7. Irrigation management measures are typically aiming at sustaining production while maintaining soil salinity in tolerable levels. Therefore, it’s not really a SICS but rather a last resort for water saving.
8. Phytoremediation increases yield (typically because of the additional biomass generated) but does not have guaranteed effects on soil salinity.
9. Rotation systems are always advisable (rather than mono cultivations).
10. Reduced tillage decreases salinity while increasing yield.

11. Most promising for salinity is a combination of amendments, conditioners and mulching, while performing flushing and maintaining cover crops or some sort of rotation.
12. Most promising for yield is phytoremediation (but this depends on yield requirements) and biological conditioners while maintain cover crops or some sort of rotation.

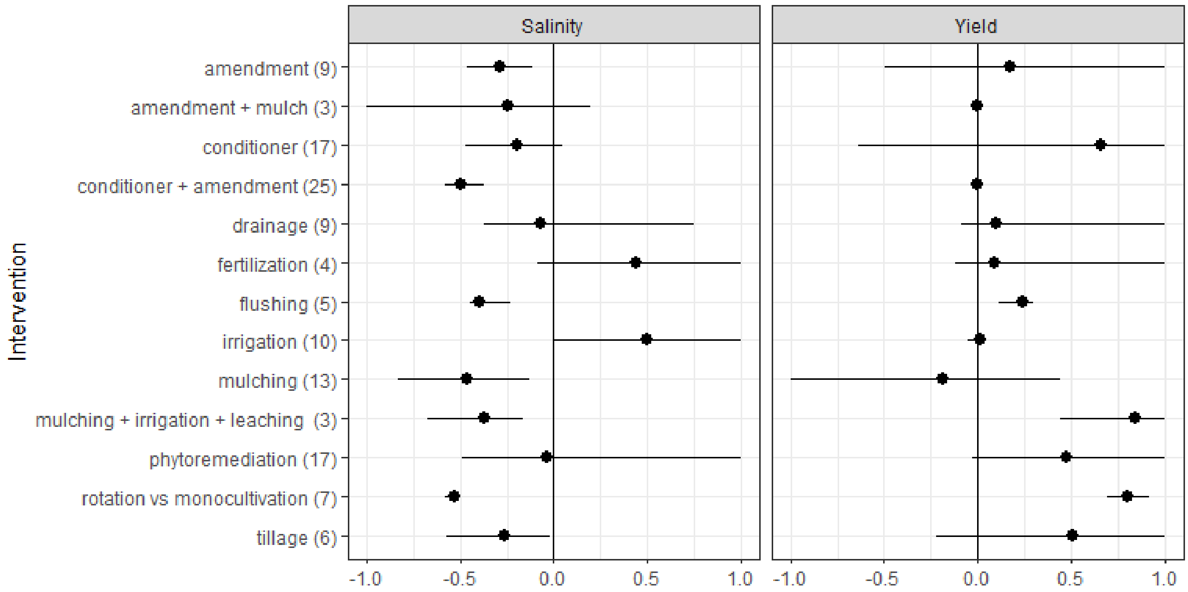


Figure 6.3. *Effect size of different agro-management techniques (focus: soil salinity) for reported data on soil salinity and corresponding reported crop yields from the studies mentioned in Table 6.1.*

6.4 Conclusions

Salinization refers to the accumulation of water soluble salts in soil. It leads to a lower soil fertility, poor soil structure, decreased infiltration, lower crop yields, lower biodiversity and biological activity. It may occur in areas where evapotranspiration is larger than precipitation, in deltas, plains and valleys with salty groundwater intrusion, and/or through the addition of fertilizers and salty irrigation water. The impact of salinization depends on the type and concentration of the salt and soil pH.

Salinization-specific SICS prevent salinization and/or lower the accumulation of unwanted salts and contribute to improving soil structure. Salinization-specific SICS are highly site-specific, and may involve all three mechanisms, i.e., (i) changes in input-output ratio's, (ii) substitution, and (iii) redesign. The first mechanism involves improved drainage through groundwater level control and channelling, reduced evaporation (through mulching), less input of soluble fertilisers, and targeted irrigation with low EC water. The second mechanism involves drip irrigation instead of surface irrigation. The third mechanism includes ridging, (plastic) mulching, and growing tolerant crops.

Most promising salinization-specific SICS (i) reduce the input of unwanted salts into the soil, (ii) decrease the content of unwanted salts in soil, and (iii) minimize the impact of unwanted salts in soil on soil functioning (Table 6.2). The greatest effects can be expected from irrigation and drainage management.

Table 6.2. *Qualitative assessment of salinization-specific SICS.*

	Components of cropping systems	Components of salinization-specific SICS	Change in profitability	Changes in soil properties		
				Physical	Chemical#	Biological
	Crop rotations					
A	Nutrient management	Amendment	51%		-27%	
		Conditioner	48%		-4%	
		Fertilization	2%		50%	
B	Irrigation management	Deficit irrigation	9%		44%	
		Flushing	24%		-39%	
C	Drainage management	Drainage	-19%		-46%	
D	Tillage management	Reduced tillage	17%		-29%	
E	Residue management	Mulching	10%		-7%	
F	Landscape management	Phytoremediation	66%		-20%	
		Rotation vs monocultivation	0%		-25%	
G	Combinations	Amendment and mulch	80%		-53%	
		Conditioner and amendment	84%		-37%	
		Mulching, irrigation and drainage	0%		-49%	

#: This refers to SAR/EC. Sign may need to be reversed depending on the context of "chemical change" (i.e. the property reduces but this is "good"). Physical and biological properties may be added empirically.

6.5 References

Abebe, T. Guenzi, A.C., Martin, B. and Cushman, J.C. 2003. Tolerance of mannitol-accumulating transgenic wheat to water stress and salinity. Plant Physiol. 131(4): 1748–1755.

Abliz, A., Tiyip, T., Ghulam, A., Halik, U., Ding, J. Sawut, M., Zhang, F., Nurmemet, I. and Abliz, A. 2016. Environ Earth Sci. 75: 260.

Abu Hammad, A. and Tumeizi, A., 2012. Land degradation: socioeconomic and environmental causes and consequences in the eastern Mediterranean. L. Degrad. Dev. 23, 216–226.

Adnan, M., Zia-ur-Rehman, M., Yousaf, B., Murtaza, G., Yousaf, A., Adnan M., and Khan, M.A. 2014. The evaluation of various soil conditioners effects on the amelioration of saline-sodic soil. J. Environ. Earth Sci. 4, 22–29.

Ado, M. N., Y. Guero, D. Michot, B. Soubeiga, T. S. Kiesse and C. Walter. Phytodesalinization of irrigated saline Vertisols in the Niger Valley by *Echinochloa stagnina*. Agric. Water Mange. 117: 229-240.

Ahmad, M.J., Arif, M., Iqbal, A., Khalid, M. and Akhtar, N. 2013. Rice Production in Salt-Affected Soils of Pakistan Using Different Reclamation Techniques. In: Soil Salinity Assessment and Reclamation, Innovative Thinking and Use of Marginal Soil and Water Resources in Irrigated Agriculture (eds. Shabbir, A. S., Abdelfattah, M. & Taha, F. K.) pp 283–293. Springer (New York Dordrecht Heidelberg London).

Ahmad, N., Qureshi, R.H., and Qadir, M. 1990. Amelioration of a calcareous saline-sodic soil by gypsum and forage plants. L. Degrad. Dev. 2: 277-284.

Ahmad, S., Ghafoor, A., Akhtar, M.E. and Khan, M.Z. 2013. Ionic displacement and reclamation of saline-sodic soils using chemical amendments and crop rotation. L. Degrad. Dev. 24: 170-178.

Alcon, F., Tapsuwan, S., Brouwer, R., and de Miguel, M.D. 2014. Adoption of irrigation water policies to guarantee water supply: A choice experiment. Environmental Science & Policy 44: 226-236.

Al-dhuhli, H.S., Al-rawahy, S.A. and Prathapar, S. 2010. Effectiveness of mulches to control soil salinity in Sorghum fields irrigated with saline water. A Monogr. Manag. Salt-Affected Soils Water Sustain. Agric. Sultan Qaboos Univ. 41–46.

Anderson, R.C., 1997. Environmental damage assessment of the Aral Sea disaster, Environmental Policy and Technology Project, Issue Paper No.1, U.S. Agency for International Development, p. 25

Apostolakis, A., Wagner, K., Daliakopoulos, I.N. and Tsanis, I.K., 2016. Greenhouse soil moisture deficit under saline irrigation and climate change. Procedia Engineering 166: 537–544.

Aquastat, 2017. FAO's information system on water and agriculture [WWW Document].

Aragüés, R. Medina, E.T., Martínez-Cob, A. and Faci, J. 2014b. Effects of deficit irrigation strategies on soil salinization and sodification in a semiarid drip-irrigated peach orchard. Agric. Water Manage. 142: 1-9.

Aragüés, R., Medina, E. T., Zribi, W., Clavería, I., Álvaro-Fuentes, J., and Faci, J. 2015. Soil salinization as a threat to the sustainability of deficit irrigation under present and expected climate change scenarios. Irr. Sci. 33: 67-79.

Aragüés, R., Medina, E.T., Clavería, I., Martínez-Cob, A. and Faci, J. 2014a. Regulated deficit irrigation, soil salinization and soil sodification in a table grape vineyard drip-irrigated with moderately saline waters. Agric. Water Manage. 134: 84-93.

Ashraf, M. and Saeed, M.M. 2006. Effect of improved cultural practices on crop yield and soil salinity under relatively saline groundwater applications. Irrigation and Drainage Systems, 20: 111-124.

Azhar, I., Adamowski, J., Prasher, S. and Albano, R. 2015. Development and application of a decision support system for optimizing cropping patterns under saline agriculture conditions in Rechna Doab, Pakistan .2015 ASABE Annual International Meeting. Paper No. 152189304.

Bacilio, M., Moreno, M. and Bashan, Y. 2016. Mitigation of negative effects of progressive soil salinity gradients by application of humic acids and inoculation with Pseudomonas stutzeri in a salt-tolerant and a salt-susceptible pepper. Applied Soil Ecology 107: 394-404.

Bai, Z. Caspari, T. Van Lynden, G., Mantel, S. and Wolters, S. 2015. Measures for preventing, mitigating and remediating soil threats in Europe: A literature review. Recare Project Report. www.recare-project.eu.

Barrett-Lennard, E.G. 2002. Restoration of saline land through revegetation. Agric. Water Manage. 53: 213–226.

Bezborodov, G.A., Shadmanov, D.K., Mirhashimov, R.T., Yuldashev, T., Qureshi, A.S., Noble, A.D. and Qadir, M. 2010. Mulching and water quality effects on soil salinity and sodicity dynamics and cotton productivity in Central Asia. Agriculture, ecosystems & environment, 138: 95-102.

Bhutta, M. N. and Smedema, L.K. 2007. One hundred years of waterlogging and salinity control in the Indus valley, Pakistan: A historical review. Irrigation and drainage, 56: S81-S90.

Bhutta, M.N. and Chaudhry, M.R., 2000. Biological control of waterlogging. In: Proceedings of Eighth ICID International Drainage Workshop. New Delhi, India, (Bio-Drainage), pp. 33–45.

Cabot, C., Sibole, J.V., Barcelo, J. and Poschenrieder, C. 2014. Lessons from crop plants struggling with salinity. Plant Sci. 226: 2–13.

Cao, Z. H., Huang, J. F., Zhang, C. S. and Li, A. F. 2004. Soil quality evolution after land use change from paddy soil to vegetable land. Environ. Geochem. Health 26: 97–103.

Castanheira, P. J. N. and Serralheiro, R. P. 2010. Impact of mole drains on salinity of a vertisoil under irrigation. Biosystems Engineering, 105: 25-33.

Chartzoulakis, K.S., 2005. Salinity and olive: growth, salt tolerance, photosynthesis and yield. Agric. Water Manage. 78: 108–121.

Chesworth, W. 2008. Encyclopedia of Soil Science, Encyclopedia of Earth Sciences Series. Springer Netherlands, Dordrecht. doi:10.1007/978-1-4020-3995-9.

Cocks, P.S. 2001. Ecology of herbaceous perennial legumes: A review of characteristics that may provide management options for the control of salinity and waterlogging in dryland cropping systems. Australian Journal of Agricultural Research 52: 137-151.

Colla, G., Rouphael, Y., Leonardi, C. and Bie, Z. 2010. Role of grafting in vegetable crops grown under saline conditions. Sci. Hort. 127: 147-155.

Colmer, T.D., Flowers, T.J. and Munns, R. 2006. Use of wild relatives to improve salt tolerance in wheat. J. Exp. Bot. 57: 1059-78.

Connor, D. J. 2004. Designing cropping systems for efficient use of limited water in southern Australia. European Journal of Agronomy, 21: 419-431.

Crescimanno, G. and Garofalo, P. 2006. Management of irrigation with saline water in cracking soils. Soil Science Society of America Journal, 70: 1774-1787.

Cuevas, J., Cañete, M.L., Pinillos, V., Zapata, A.J., Fernández, M.D., González, M. and Hueso, J.J. 2007. Optimal dates for regulated deficit irrigation in 'Algerie' loquat (*Eriobotrya japonica* Lindl.) cultivated in Southeast Spain. Agric. Water Manage. 89: 131-136.

Daliakopoulos, I.N., Apostolakis, A., Wagner, K., Deligianni, A., Koutskoudis, D., Stamatakis A. and Tsanis, I.K. (under review). Benefits of T. harzianum in soil and yield conservation of tomato crops under saline irrigation. Catena.

Daliakopoulos, I.N., Panagea, S.I., Tsanis, I.K., Grillakis, M.G., Koutroulis, A.G., Hessel, R., Mayor, A.G. and Ritsema, C.J. 2017. Yield response of Mediterranean rangelands under a changing climate. L. Degrad. Dev. 28: 1962–1972.

Daliakopoulos, I.N., Tsanis I.K., Koutroulis, A., Kourgialas N., Varouchakis, E., Karatzas P. and Ritsema C. 2016. The threat of soil salinity: a European scale review. Science of the Total Environment 574: 727-739.

Darwish, T., Atallah, T., El Moujabber, M. and Khatib, N. 2005. Salinity evolution and crop response to secondary soil salinity in two agro-climatic zones in Lebanon. Agric Water Manage. 78: 152–164.

Deng, L., Chen, M., Liu, Z., Shen, Q., Wang, H. and Wang, J. 2003. Effects of different ground covers on soil physical properties and crop growth on saline–alkaline soil. Chin. J. Soil Sci. 34: 93–97.

Díez, J. A., Caballero, R., Román, R., Tarquis, A., Cartagena, M. C. and Vallejo, A. 2000. Integrated fertilizer and irrigation management to reduce nitrate leaching in central Spain. Journal of Environmental Quality 29: 1539-1547.

Domínguez-Beisiegel, M., Herrero, J. and Castañeda, C., 2013. Saline wetlands' fate in inland deserts: an example of 80 years' decline in Monegros, Spain. L. Degrad. Dev. 24: 250–265.

English, M.J., Solomon, K.H. and Hoffman, G.J. 2002. A paradigm shift in irrigation management. Journal of Irrigation and Drainage Engineering 128: 267-277.

FAO. 2002. Agricultural drainage water management in arid and semi-arid areas. FAO Irrigation and drainage paper 61. Annex 1. Crop salt tolerance data http://www.fao.org/docrep/005/y4263e/y4263e0e.htm

Finlayson, J., Bathgate, A., Nordblom, T., Theiveyanathan, T., Farquharson, B., Crosbie, R., Mitchell, D. and Hoque, Z. 2010. Balancing land use to manage river volume and salinity: Economic and hydrological consequences for the Little River catchment in Central West, New South Wales, Australia. Agricultural Systems 103: 161–170.

Forkutsa, I., Sommer, R., Shirokova, Y.I., Lamers, J.P.A., Kienzler, K., Tischbein, B., Martius, C. and Vlek, P.L.G. 2009. Modeling irrigated cotton with shallow groundwater in the Aral Sea Basin of Uzbekistan: II. Soil salinity dynamics. Irr. Sci. 27: 319-330.

Gabriel, J.L., Vanclooster, M. and Quemada, M. 2014. Integrating water, nitrogen, and salinity in sustainable irrigated systems: Cover crops versus fallow. Journal Irrigation and Drainage Engineering, 140: A4014002-1 – A4014002-11.

Geeson, N.A., Brandt, C.J. and Thornes, J.B. 2003. Mediterranean desertification: a mosaic of processes and responses. John Wiley & Sons, Chichester, UK

Ghassemi, F., Jakeman, A.J. and Nix, H.A. 1995. Salinisation of land and water resources: human causes, extent, management and case studies. CAB international.

González-Vázquez. C., Grande, J.A. Barragán, J., Ocaña, J.A., and De La Torre, M.L. 2005. Nitrate accumulation and other components of the groundwater in relation to cropping system in an aquifer in Southwestern Spain. Water Resour. Manage. 19: 1-22.

Greiner, R. 1997. Optimal farm management responses to emerging soil salinisation in a dryland catchment in Eastern Australia. L. Degrad. Dev. 8: 71–93.

Hanson, B., Hopmans, J.W. and Simunek, J. 2008. Leaching with subsurface drip irrigation under saline, shallow groundwater conditions. Vadose Zone Journal 7: 810–818.

Heuperman, A. 1999. Hydraulic gradient reversal by trees in shallow water table areas and repercussions for the sustainability of tree-growing systems. Agric. Water Manage. 39: 53–167.

Hira, G.S. and Thind, H.S. 1987. A plantation technique for sal-affected soils with shallow water table. Landscape and Urban planning, 14: 445-446.

Hoffman, G.J. and Shannon, M.C. 2007. Salinity. In: Lamm, F.R., Ayars, J.E., Nakayama, F.S. (eds.) Microirrigation for Crop Production Design, Operation and Management. Developments in Agricultural Engineering 13: 131-160.

Hu, L., Lu, H., Liu, Q., Chen, X. and Jiang X. 2005. Overexpression of mtlD gene in transgenic *Populus tomentosa* improves salt tolerance through accumulation of mannitol. Tree Physiol. 25: 1273-81.

Huang, J., Yu, H., Guan, X., Wang, G. and Guo, R. 2016. Accelerated dryland expansion under climate change. Nature Climate Change 6: 166-171.

Huang, Q., Yin, Z., Tian, C. 2001. Effect of two different straw mulching methods on soil solute salt concentration. Arid Land Geogr. 24, 52–56.

Hueso, J.J. and Cuevas, J. 2008. Loquat as a crop model for successful deficit irrigation. Irr. Sci. 26: 269-276.

Hueso, J.J. and Cuevas, J. 2010. Ten consecutive years of regulated deficit irrigation probe the sustainability and profitability of this water saving strategy in loquat. Agric. Water Manage. 97: 645-650.

International Programme for Technology and Research in Irrigation and Drainage (IPTRID), 2002. Biodrainage— Principles, Experiences and Applications, Knowledge Synthesis Report No. 6, FAO, Rome.

Kafi, M., Asadi, H. and Ganjeali, A. 2010. Possible utilization of high-salinity waters and application of low amounts of water for production of the halophyte *Kochia scoparia* as alternative fodder in saline ecosystems. Agric. Water Manage. 97: 139-147.

Kaur, R., Paul, M. and Malik, R. 2007. Impact assessment and recommendation of alternative conjunctive water use strategies for salt affected agricultural lands through a field scale decision support system – a case study. Env. Monit. Assess. 129: 257–270.

Khan, M.A., Ansari, R., Ali, H., Gul, B. and Nielsen, B.L. 2009. Panicum turgidum, a potentially sustainable cattle feed alternative to maize for saline areas. Agric. Ecosys. Environ. 129: 542–546.

Khan, S. O'Connell, N. Rana, T. and Xevi, E. 2007. Hydrologic-economic model for managing irrigation intensity in irrigation areas for water table and soil salinity targets. Environ. Model Assess. 13: 115-120.

Kitamura, Y., Yano, T., Honna, T., Yamamoto, S. and Inosako, K. 2006. Causes of farmland salinization and remedial measures in the Aral Sea basin—Research on water management to prevent secondary salinization in rice-based cropping system in arid land. Agric. Water Manage. 85: 1-14.

Konukcu, F., Gowing, J.W. and Rose, D.A. 2006. Dry drainage: A sustainable solution to waterlogging and salinity problems in irrigation areas? Agric. Water Manage. 83: 1-12.

Kotb, T.H., Watanabe, T., Ogino, Y. and Tanji, K.K. 2000. Soil salinization in the Nile Delta and related policy issues in Egypt. Agric. Water Manage. 43: 239-261.

Koubouris, G.C., Tzortzakis, N., Kourgialas, N.N., Darioti, M. and Metzidakis, I. 2015. Growth, photosynthesis and pollen performance in saline water treated olive plants under high temperature. Int. J. Plant Biol. 6: 28-32.

Lefroy, E.C. and Stirzaker, R.J. 1999. Agroforestry for water management in the cropping zone of southern Australia. Agroforestry Syst. 45: 277-302.

Li, J., Pu, L., Zhu, M. and Zhang, R., 2012a. The present situation and hot issues in the salt-affected soil research. Acta Geogr. Sin. 67: 1233–1245.

Li, Q., Ma, L., Wang, X., Wei, M., Shi, Q. and Yang, F. 2012b. Effects of short-term summer fallow and rotations on soil quality under plastic greenhouse cultivation. J Food Agric Environ. 10: 1106-1110.

Li, X. and Zhang, Z. 1999. Effect of straw mulching on soil water and salt movement. Chin. J. Soil Sci. 30: 257–258.

Li, X., Zhang, Z., Liu, X., Li, Y. 2000. Effect of straw mulching on soil water and salt moving. J. Shandong Agric. Univ. 31: 38–40.

Lozano-García, B., Parras-Alcántara, L. and Muriel-Fernández, J. L. 2011. Soil tillage effects on monovalent cations (Na+ and K+) in vertisols soil solution. Catena 84: 61-69.

Maas, E.V. 1986. Salt tolerance of plants. Applied Agricultural Research 1:12-26.

Mahdy, A. M. 2011. Comparative effects of different soil amendments on amelioration of saline-sodic soils. Soil Water Res. 6: 205–216.

Manjunatha, M.V., Oosterbaan, R. J., Gupta, S. K., Rajkumar, H. and Jansen, H. 2004. Performance of subsurface drains for reclaiming waterlogged saline lands under rolling topography in Tungabhadra irrigation project in India. Agric. Water Manage. 69: 69-82.

Marchand, M. and H. Abd El Hadi. 2002. Long-term experiments comparing the impact on soils and field crops of potassium chloride vs. potassium sulfate. Acta Hort. 573: 49-54.

Mateo-Sagasta, J. and Burke, J. 2011. Agriculture and water quality interactions: a global overview. SOLAW Backgr. Themat. Rep. - TR08.

Matthews, N., Grové, B., Barnard, J.H. and Rensburg, L.D. 2010. Modelling the economic tradeoffs between allocating water for crop production or leaching for salinity management. Water SA, 36: 37-44.

Mbarki, S., Cerdà, A., Brestic, M., Mahendra, R., Abdelly, C. and Pascual, J.A. 2017. Vineyard compost supplemented with Trichoderma Harzianum T78 improve saline soil quality. L. Degr. Dev. 28: 2028-2037.

Mendes, J.P. and Carvalho, M.C. 2009. Controlo da salinidade do solo com recurso à sementeira directa. Rev. de Ciências Agrárias 32: 360-369.

Morris, J.D. and Collopy, J.J. 1999. Water use and salt accumulation by *Eucalyptus camaldulensis* and *Casuarina cunninghamiana* on a site with shallow saline groundwater. Agric. Water Manage. 39: 205-228.

Mudge, K., Janick, J., Scofield, S. and Goldschmidt, E. E. 2009. A History of Grafting. In: Horticultural Reviews, Volume 35 (ed J. Janick), John Wiley & Sons, Inc., Hoboken, NJ, USA.

Nayak, A. K., Sharma, D. K., Mishra, V. K., Minhas, P. S. and Verma, C. L. 2008. Reclamation of saline-sodic soil under a rice–wheat system by horizontal surface flushing. Soil use and management 24: 337-343.

Nevo, E. and Chen, G. X. 2010. Drought and salt tolerances in wild relatives for wheat and barley improvement. Plant Cell Environ. 33: 670–685.

Oron, G., De Malach, Y., Gillerman, L., David, I. and Lurie, S. 2002. Effect of water salinity and irrigation technology on yield and quality of pears. Biosystems Engineering 81: 237-247.

Panagea, I.S., Daliakopoulos, I.N., Tsanis, I.K. and Schwilch, G., 2016. Evaluation of promising technologies for soil salinity amelioration in Timpaki (Crete): a participatory approach. Solid Earth 7: 177-190.
Pang, H. and Xu, F. 1998. Study on the tillage methods under straw mulching in Weibei Arid Area. Res. Agric. Mod. 19: 249–251.
Pang, H.1999. The effect of straw mulching on the soil environment and wheat yield traits. Chin. J. Soil Sci. 30: 174–175.
Pang, H.C., Li, Y.Y., Yang, J.S. and Lianget, Y.S. 2010. Effect of brackish water irrigation and straw mulching on soil salinity and crop yields under monsoonal climatic conditions. Agric. Water Manage. 97: 1971-1977.
Paranychianakis, N.V. and Chartzoulakis, K.S. 2005. Irrigation of Mediterranean crops with saline water: from physiology to management practices. Agric. Ecosyst. Environ. 106: 171–187.
Pedreño, J. N., Gómez, I., Moral, R. and Mataix, J. 1996. Improving the agricultural value of a semi-arid soil by addition of sewage sludge and almond residue. Agric. Ecosyst. Environ. 58: 115–119.
Pinillos, V., Chiamolera, F.M., Ortiz, J.F., Hueso, J.J. and Cuevas, J. 2016. Post-veraison regulated deficit irrigation in 'Crimson Seedless' table grape saves water and improves berry skin color.
Pitman, M. G. and Laüchli, A. 2004. Global impact of salinity and agricultural ecosystems. In: Lau ̈chli, A. and Lu ̈ttge, U. (eds.) Salinity: Environment-Plants-Molecules. Kluwer Academic Publishers, Dordrecht. pp. 3–20.
Pulatov, A., Egamberdiev, O., Karimov, A., Tursunov, M., Kienzler, S., Sayre, K., Tursunov, L., Lamers, J.P.A. and Martius, C. 2012. Introducing conservation agriculture on irrigated meadow alluvial soils (Arenosols) in Khorezm, Uzbekistan. In: Martius C., Rudenko I., Lamers J., Vlek P. (eds) Cotton, Water, Salts and Soums. Springer, Dordrecht
Qadir, M. and Oster, J.D. 2004. Crop and irrigation management strategies for saline-sodic soils and waters aimed at environmentally sustainable agriculture. Science of the total environment 323: 1-19.
Qadir, M., Noble, A.D., Schubert, S., Thomas, R.J. and Arslan, A., 2006a. Sodicity-induced land degradation and its sustainable management: problems and prospects. L. Degr. Dev. 17: 661-676.
Qadir, M., Qureshi, R.H., Ahmad, N. and Ilyas, M. 1996. Salt-tolerant forage cultivation on a saline-sodic field for biomass production and soil reclamation. L. Degrad. Dev. 7: 11–18.
Qadir, M., Schubert, S., Noble, A.D., Saqib, M. and Saifullah. 2006b. Amelioration strategies for salinity-induced land degradation. CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources, 1, art. no. 069.
Qadir, M., Tubeileh, A., Akhtar, J., Larbi, A., Minhas, P.S. and Khan, M.A. 2008. Productivity enhancement of salt-affected environments through crop diversification. L. Degrad. Dev. 19: 429-453.
Qiao, H., Liu, X., Li, W., Huang, W., Li, C. and Li, Z. 2006. Effect of deep straw mulching on soil water and salt movement and wheat growth. Chin. J. Soil Sci. 37: 885–889.
Qureshi, A.S., Qadir, M., Heydari, N., Turral, H. and Javadi, A. 2007. A Review of management strategies for salt-prone land and water resources in Iran. International Water Management Institute, Colombo, Sri Lanka, IWMI Working Paper 125.
Raats, P.A. 2014. Salinity management in the coastal region of the Netherlands: A historical perspective. Agric. Water Manage. 157: 12–30.
Rabhi, M., Ferchichi, S., Jouini, J., Hamrouni, M.H., Koyro, H-W., Ranieri, A., Abdelly, C. and Smaoui, A. 2010. Phytodesalination of a salt-affected soil with the halophyte Sesuvium portulacastrum L. to arrange in advance the requirements for the successful growth of a glycophytic crop. Bioresource Technology 101: 6822-6828.
Rengasamy, P. 2006. World salinization with emphasis on Australia. J. Exp. Bot. 57: 1017–1023.
Ritzema, H.P. 2016. Drain for Gain: Managing salinity in irrigated lands — A review. Agricultural Water Management 176: 18–28.
Ritzema, H.P., Satyanarayana, T.V., Raman, S. and Boonstra, J. 2008. Subsurface drainage to combat waterlogging and salinity in irrigated lands in India: Lessons learned in farmers' fields. Agric. Water Manage. 95: 179-189.
Roy, S.J., Negrao, S. and Tester, M. 2014. Salt resistant crop plants. Curr. Opin. Biotechnol. 26: 115-124.

Satyanarayana, T.V., Lakshmi, G.V., Hanumanthaiah, C.V., Srinivasulu, A., Ratnam, M., 2003. Feasible subsurface drainage strategies to combat waterlogging and salinity in irrigated agricultural lands in Andhra Pradesh. Indo-Dutch Network Bridging Project on Drainage and Water
Shaaban, M., Abid, M. and Abou-Shanab, R.A. 2013. Amelioration of salt affected soils in rice paddy system by application of organic and inorganic amendments. Plant Soil Environ. 59: 227–233.
Sharma, D.P. and Gupta, S.K. 2006. Subsurface drainage for reversing degradation of waterlogged saline lands. L. Degrad. Dev. 17: 605-614.
Sharma, P.K. and Bhagat, R.M. 1993. Puddling and compaction effects on water permeability of texturally different soils. J. Indian Soc. Soil Sci. 41: 1–6.
Slavich, P.G., Smith, K.S., Tyerman, S.D. and Walker, G.R. 1999. Wateruse of grazed saltbush plantations with saline water table. Agric. Water Manage. 39: 169-186.
Sparks, D.L. 2003. Environmental soil chemistry. Academic Press.
Suárez, D. L. 2001. Sodic soil reclamation: Modelling and field study. Aust. J. Soil Res. 39: 1225–1246.
Sun, J., Tian, Y., Gao, L., Peng, X. and Tong, E. 2014. Effects of straw biological reactor and microbial agents on physicochemical properties and microbial diversity of tomato soil in solar greenhouse. Trans. Chin. Soc. Agric. Eng. 6: 153–164 (in Chinese with English abstract).
Tanji, K.K. and Wallender, D.D. 1990. Nature and extent of agricultural salinity and sodicity. In: Tanji, K.K. and Wallender, D. D. (eds.) Agricultural Salinity Assessment and Management. Manuals and Reports on Engineering Practices No. 71. American Society of Civil Engineers, New York. pp. 1–25.
Tian, K. and Lei, Y. 1994. Integrated effect of water, fertilize and salinity on the wheat yield in desalinized fluvo-aquic soil in Heilonggang Region. Res. Agric. Mod. 15: 364–368
Trnka, M., Kersebaum, K.C., Eitzinger, J., Hayes, M., Hlavinka, P., Svoboda, M., Dubrovský, M., Semerádová, D., Wardlow, B., Pokorný, E., Možný, M., Wilhite, D. and Žalud, Z. 2013. Consequences of climate change for the soil climate in Central Europe and the central plains of the United States. Clim. Change 120: 405–418.
Tsanis, I.K., Daliakopoulos, I.N., Koutroulis, A.G., Karatzas, G.P., Varouchakis, E. and Kourgialas, N. 2015. Soil salinization. In: Stolte, J., Tesfai, M., Øygarden, L. (eds.). Soils in Europe-Threats, functions and ecosystem services. Recare Project Report. www.recare-project.eu.
Van Beek, C.L. and Tóth, G. 2012. Risk assessment methodologies of soil threats in Europe. JRC Scientific and Policy Reports EUR 24097.
Van Weert, F., van der Gun, J. and Reckman, J. 2009. Global overview of saline groundwater occurrence and genesis, International Groundwater Resources Assessment Centre. Utrecht.
Vinod, K. K., Krishnan, G. S., Babu, N. N., Nagarajan, M. and Singh, A. K. 2013. Improving salt tolerance in rice: looking beyond the conventional. In: Salt Stress in Plants: Signalling, Omics and Adaptations (eds P. Ahmad, M. M. Azooz, and M. N. V. Prasad) New York, NY: Springer, pp 219–260.
Vlek, P.L.G., Martius, C., Schoeller-Schletter, A. and Wehrheim, P. 2002. Economic and ecological restructuring of land- and water use in the Region Khorezm (Uzbekistan): a pilot project in development research, Project Proposal, ZEF/ UNESCO Khorezm Project, ZEF, Bonn, Germany, p. 90.
Wang, X., Yang, J., Liu, G., Yao, R. and Yu, S. 2015. Impact of irrigation volume and water salinity on winter wheat productivity and soil salinity distribution. Agric. Water Manage. 149: 44-54.
Weaver, T. B., Hulugalle, N. R., Ghadiri, H. and Harden, S. 2013. Quality of drainage water under irrigated cotton in vertisols of the lower Namoi Valley, New South Wales, Australia. Irrigation and Drainage 62: 107-114.
Wichelns, D. and Oster, J. D. 2006. Sustainable irrigation is necessary and achievable, but direct costs and environmental impacts can be substantial. Agric. Water Manage. 86: 114-127.
Yamazaki, F., 1976. Farmland Engineering, vol. 1. Tokyo University Publication, Tokyo (in Japanese).
Yensen, N.P. and Biel, K.Y. 2006. Soil remediation via salt-conduction and the hypotheses of halosynthesis and photoprotection. Tasks for Vegetation Science Series-40. Ecophysiology of High Salinity Tolerant Plants, pp. 313–344.

Young, R., Huth, N., Harden, S. and McLeod, R. 2014. Impact of rain-fed cropping on the hydrology and fertility of alluvial clays in the more arid areas of the upper Darling Basin, eastern Australia. Soil Research 52: 388-408.

Young, R.A. 2005. Determining the economic value of water: concepts and methods. World Bank, Washington, DC.

Zhang, N., Si, H., Wen, G. Du, H.H., Liu, B.L. and Wang, D. 2011. Enhanced drought and salinity tolerance in transgenic potato plants with a BADH gene from spinach. Plant Biotechnol. Rep 5: 71-77.

Zhou, Q., Wu, F., and Zhang, Q. 2015. Is irrigation water price an effective leverage for water management? An empirical study in the middle reaches of the Heihe River basin. Physics and Chemistry of the Earth, Parts A/B/C, 89: 25-32.

Zuazo, V.H.D., Raya, A.M. and Ruiz, J.A. 2004. Impact of salinity on the fruit yield of mango (*Mangifera indica* L. cv. "Osteen"). Eur. J. Agron. 21: 323–334.

Zuccarini, P. 2008. Ion uptake by halophytic plants to mitigate saline stress in *Solanum lycopersicon* L., and different effect of soil and water salinity. Soil & Water Res., 3: 62–76.

7 Soil-improving cropping systems for soil erosion

J. Stolte, R. Hessel, L. Øygarden, O. Green, A. Ferreira, G. Edwards, J. Poesen and M. Riksen

7.1 Background

Soil erosion can be defined as a three-phase process that consists of: (i) the detachment of individual soil particles from the soil mass; (ii) their subsequent transport by an erosive agent; and, ultimately, (iii) their deposition when the erosive agent lacks sufficient energy for further transport (Morgan, 2005). In the case of soil erosion by water, both rainsplash and water running over the soil surface detach and then move the detached particles, but running water is the principal transporting agent. Which detachments process dominates depends among others on scale; at plot scale it is usually rainsplash, but at catchment scale it is running water. Poesen et al (2003) and Poesen (in press), for example, reported that at watershed scale gully erosion may contribute more than 80% to sediment production.

Soil erosion by wind is causing severe soil degradation, mainly in arid and semi-arid areas (Woodruff and Siddoway, 1965; Kalma *et al.*, 1988). However, wind erosion can also be an important process in temperate climates if conditions are conducive: bare soil, dry conditions and strong wind Funk *et al.* (2002), for example, report that wind erosion is a serious problem in the northeastern parts of Germany because the months of highest wind erosivity (March & April) coincides with seedbed preparation for crops like sugar beet and maize. It is estimated that ca. 28% of the global land area that experiences land degradation suffers from wind-driven soil erosion process (Oldeman, 1994). A total land area of 549 Mha is potentially affected by wind erosion, of which 296 Mha could be severely affected (Lal, 2001). The movement of soil occurs when forces exerted by wind overcome the gravitational and cohesive forces of soil particles on the surface of the ground (Bagnold, 1941), and the surface is mostly devoid of vegetation, stones or snow (Shao, 2008).

Maintaining or improving soil quality and soil health is crucial for crop production, and using soil-improving cropping systems (SICS) can greatly contribute to prevention of loss of fertile soil through erosion. Here, we explore the effects of SICS on erosion and soil quality.

7.2 Purpose

The aim of this review is to perform a literature search on soil-improving cropping system reducing soil degradation caused by erosion. This chapter deals with both erosion by wind and by water, though we realize that processes are different, and so are cropping systems. For erosion by water, Maetens et al. (2012) presented a review on effectiveness of soil conservation techniques in reducing plot runoff and soil loss in Europe and the Mediterranean. Riksen et al. (2003) analysed the soil conservation policy measures for erosion by wind for Northwestern Europe.

In many cases, measures are needed to combat the effects and causes of soil erosion. Such measures can be subdivided into two major parts: physical measures that seek to prevent, control and restore, and political measures that are of major influence on the adoption of physical measures. Here we focus on physical measures and in particular on SICS.

7.3 Results and discussion

7.3.1 General principles of SICS

Many different methods exist that aim to control and prevent soil erosion in agricultural areas. These methods seek to decrease detachment and/or transport capacity. Decreasing detachment is preferable to decreasing transport capacity, because in that case soil is kept in place, there are no deposition problems elsewhere, and there are no problems with sediment enrichment, which occurs when deposition takes place (Toy et al., 2002). Detachment can be limited by decreasing erosivity of the eroding agent, or by decreasing erodibility of the soil. For wind erosion, the relative effect of reducing erosivity is much larger than that for water erosion, so that wind erosion measures are usually aimed at decreasing erosivity (Toy et al., 2002).

A very important principle is to maintain ground cover, which decreases both erosivity and erodibility. For water erosion, another important principle is to control the runoff of water in such a way that runoff is no longer erosive, which can be done by decreasing and slowing down discharge. For wind erosion, decreasing wind speed at the soil surface is likewise very important, and is usually achieved by using vegetation. Such measures seek to prevent erosion through appropriate farming systems, and mechanical measures are only needed if these systems are not effective enough, which is more likely to be the case once erosion has become a problem. Many anti erosion measures also conserve water, either by intention, or because reducing the amount of water is a good way to decrease erosion. These measures have therefore in the past often been called Soil and Water Conservation measures (SWC measures). In the last years, this term has more or less been replaced by Sustainable Land Management (SLM) measures (Liniger and Critchley 2007). Not all SLM measures are SICS, but many of them are. Key elements of SICS are:

1. Vegetative covers (including crop rotations, cover crops etc.)
2. Tillage, mulching, soil and water management
3. Structural landscape elements (grass strips, grassed waterways, alley farming etc.)

7.3.2 Vegetative covers

Crops and crop management are an integral part of SICS and include choice of crop, fallow period, planting patterns and soil cover by vegetation.

Some crops are inherently more sensitive to erosion than others, especially row crops and crops that have low cover during the most erosive period of the year. From the viewpoint of erosion control, crops should be chosen that quickly produce significant ground cover. Row crops

could be combined with cover crops. However, like for vegetative strips, the cover crop might compete for water with the main crop. Furthermore, crops are often planted in rows to allow use of machinery. The choice of an appropriate land use is of vital importance in erosion control. Which land use is most appropriate depends on erosion risk, but also on factors such as rainfall, length of the growing season, temperature, acidity and fertility of the soil.

Cover crops can be seen as a special case of mulching (discussed below), and have similar benefits, but can also provide yield in their own right. Some cover crops such as alfalfa and clover also replenish the nitrogen supply of the soil. Cover crops should be established easily, provide quick ground cover and eliminate other vegetation (Lal, 1990). Usually, low growing legumes are used. Which species is most suitable depends on local conditions such as climate, soil and farming system.

For water erosion, short cover is preferable, while for wind erosion high cover is better (Toy et al., 2002). Cover is especially crucial at the time of the year when erosivity is greatest, since it is an effective protection to falling rain and to wind, while cover at the soil surface increases flow resistance. As pointed out by Troeh et al. (1991) the use of fertilisers can also be an effective way to decrease runoff and erosion since well-fertilised crops grow more vigorously, providing more cover and therefore protecting the soil. Fallow crops can allow the soil to recuperate if the fallow period is long enough. An alternative to fallow might be crop rotation, since not all crops have the same effect on soil properties, and different crops have different erosion rates.

7.3.3 Tillage, mulching, and soil and water management

Tillage prepares a suitable seedbed, increases soil roughness and helps to control weeds. However, it has the adverse effects to make the soil more susceptible to erosion, and to cause compaction of the soil because of the weight of tillage implements such as tractors. Besides, on some soils, tillage can destroy structure instead of increasing it. In such conditions conservation tillage or even no tillage is needed. However, such systems often use herbicides for weed control, which might be detrimental to the environment. Because of different environmental conditions and different crops in different areas, the tillage method that should be used to minimise erosion will differ too, and may e.g. include no-tillage, reduced-tillage, ridge-furrow systems, hillocks and mulch tillage (Lal, 1990).

Tillage along the contour will modify flow pattern and reduce the grade. If tillage is exactly on the contours furrows would fill and overflow, but in reality there is always some grade, resulting in flow to lower areas where plough ridges might be overtopped, causing erosion. Nevertheless, net erosion is decreased and the value for the P-factor depends on slope and height of plough ridges (Renard et al., 1997). Contouring is less effective for larger storms. Studies on plots showed that erosion was more affected than runoff (see e.g. Hessel & Tenge, 2008; Maetens et al 2012). To prevent erosion in places where water concentrates grassed waterways can be used. Contouring is less effective with increasing grade and loses effectiveness for long slopes.

Tillage performed on dry soils can lead to pulverisation of the aggregates, making the soil more susceptible to erosion (Munkholm, 2011). Dexter and Bird (2001) synthesised previous studies to determine a range of water contents under which soils could be worked to create desirable soil structures without requiring excessive energy inputs. Edwards et al. (2016) showed that through monitoring and evaluating soil conditions, tillage operations can be planned so that the risk of erosion decreases. Nielsen et al. (2015) showed how varying soil types can affect the efficiency of soil tillage operations, whilst Suomi and Oksanen (2015) showed that through the utilisation of sensors and actuators operations could be tailored to site specific conditions.

Holland (2004) showed that conservation tillage provides a wide range of benefits to the environment, and still has the potential to allow farmers to continue cropping profitably. By preserving soil and maintaining it in optimum condition crop yields are sustained. Lemken Gmbh & Co. (2010) was the first commercial company to address this problem with their electrical driven power harrow designed for easily optimizable seedbed tillage operation [patent ref.: DE102010013407 A1] enabling the possibility for site specific tillage intensity application.

Mulching is the covering of the soil, usually with plant residue, which acts as a buffer because it dampens the effect of the environment on the soil. The cover protects the soil from raindrop impact, reduces the velocity of wind and water and can enhance soil structure, thereby greatly reducing erosion. It also decreases soil temperature and increases soil moisture. It may be used as an alternative to cover crops, since it protects the soil but does not compete for water. However, it might encourage weed growth, unless the mulch application rate is high. Mulch should cover about 75% of the soil (Morgan, 1986). Mulches are usually made from crop residue, or from plant material brought in from elsewhere, but they can also be made from inorganic materials and gravel (Lal, 1990).

Prosdocimi et al. (2016) examined the use of barley straw as mulching treatment in reducing soil erodibility in Mediterranean vineyards. Keesstra et al (2016) examined the effects of soil management on erosion in apricot orchards. Sheehy et al. (2015) showed that no-till and minimum tillage had a positive effect on soil aggregation in Northern European agroecosystems. This may reduce erosion risk, although their study was focusing on the effect on carbon stock. Cerdan et al. (2010) concluded from their review on erosion plot data through Europe that land use has an overwhelming effect on erosion rates: soil losses on conventionally tilled arable land are often more than a magnitude higher than for permanent vegetated plots. Van den Putte et al. (2010) performed a European wide inventory on the effects of conservation tillage on yields, and concluded that the yield reduction was on average 4.5%. The degree of yield reduction, however, varies strongly (from 0-30%, depending on crop type, tillage technique, soil texture and crop rotation). Seufert et al (2012) showed in their meta-analysis on organic versus conventional agriculture a yield ratio of 0.75, but with large differences between crop types (Figure 7.1). This suggests indeed that responses to agro-management techniques greatly differ between crop types.

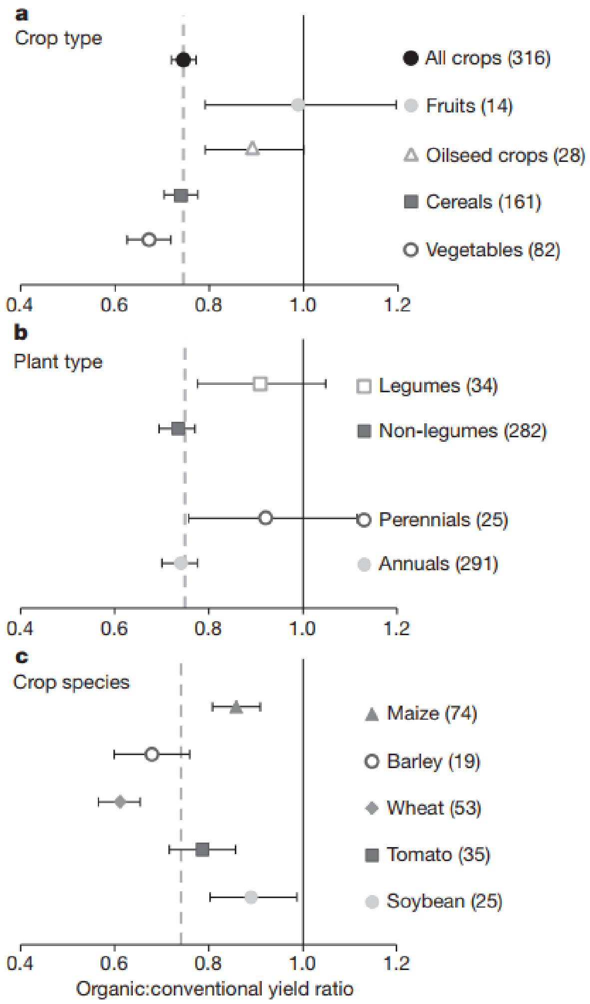


Figure 7.1. *Influence of different crop types, plant types and species on organic-to-conventional yield ratios. a–c, Influence of crop type (a), plant type (b) and crop species (c) on organic-to-conventional yield ratios (from Seufert et al., 2012).*

Pittelkow et al. (2015) performed a comparable study for non-till versus conventional agriculture, and concluded that, when averaged across all observations, the implementation of no-till leads to a significant decrease in yield (5.1%), depending also on the duration of the practice (Figure 7.2). Many studies have been performed and are still ongoing on effective land use systems to reduce and prevent soil erosion. Tilman et al. (2002) raised the question how society can accomplish the dual objectives of improving yield and preserving the quality and quantity of ecosystem services by land and water resources. For soil fertility, they present a list of causes and possible remediation measures.

Soil management is intended to enhance the structure of the soil, as well as the roughness of the soil surface. This may be achieved by applying organic matter, which increases cohesion,

water retention and aggregate stability. It can also be achieved by applying soil stabilisers, such as organic products, polyvalent salts and synthetic polymers (Morgan, 1986; Lal, 1990). Most of these are, however, too expensive to use at large scale, and can therefore only be used for specific local problems.

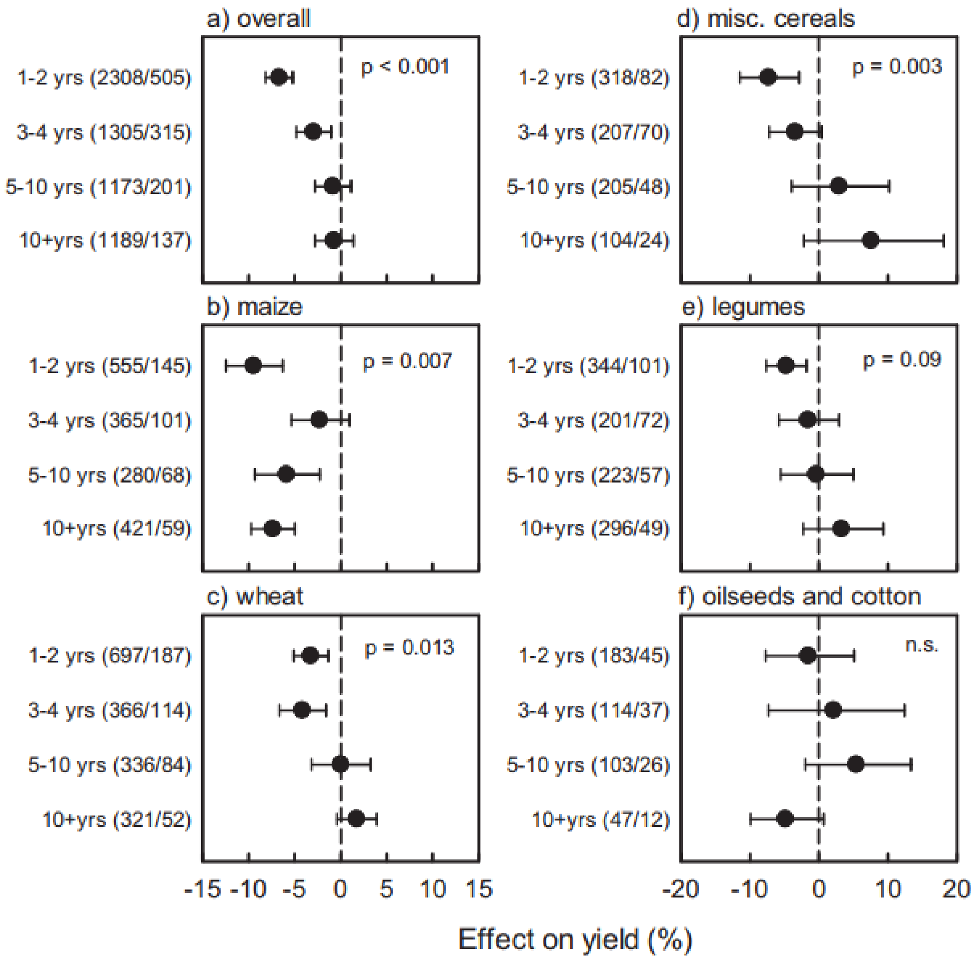


Figure 7.2. *The influence of no-till duration on the yield impacts of no-till relative to conventional tillage for different crop categories. Misc. cereals include barley, millet, oat, rye, sorghum, tef, and triticale. The number of observations and total number of studies included in each category are displayed in parentheses. Error bars represent 95% confidence intervals. Significant differences by no-till duration are indicated by p-values based on randomization tests. n.s. = non-significant (From Pittelkow et al., 2015).*

Water management is intended to effectively use the available water from crop production, while excess water is drained safely, to prevent water erosion and water logging. Drainage ditches can be used to prevent that water from upslope enters agricultural land. The drainage ditches should not only convey runoff from the cropland, but they should also dispose of it in

a safe manner, e.g. into a grassed waterway, which in turn drains into the natural drainage system. The grassed waterways increase flow resistance and therefore slow down flow, while the resistance to erosion is also increased. Ditches and ridges may also be used to store water on the slope, instead of discharging it. This should result in increased infiltration and increased soil moisture. Care must be taken that such structures are on the contour, because else they are likely to fail, resulting in concentrated erosion and gullying. Irrigation may induce water erosion, depending on irrigation system. Flood irrigation and center-pivots are well known for inducing erosion. Ferreira (2001) and Silva et al. (2005) reported erosion rates significantly higher than those produced by natural rainfall, in a semi-arid southern Alentejo (Portugal). The erosion values for irrigation center-pivots were found unsustainable both for conventional and minimum tillage (Ramos et al., 2010).

Figueiredo et al (2009) highlight the importance of maintaining soil stoniness to reduce soil erosion, a practice well disseminated in small farm in the north and center of Portugal. The author found that stoniness may reduce erosion by 38-60%, compared with bare soil without rock fragments. Martins et al. (2007) present information on different types of pastures in a semi-arid environment in southeastern Alentejo (Portugal). The authors came to the conclusion that soil tillage was responsible for a significant increase in erosion rates. Fertilization also plays an important role, with the use of mineral fertilizers presenting higher overland flow rates when compared with the areas where organic fertilizers were applied. The use of treated sewer sludge was proven to be beneficial in the reduction of overland flow and erosion rates.

7.3.4 Structural landscape elements

Structural landscape elements include vegetative strips, which can be divided in grass strips, hedges and strip cropping. Strips should be placed on the contour, or perpendicular to the wind direction. In strip-cropping, low cover strips are alternated with high cover strips (such as grasses and legumes). Crops can be rotated, but some strips might also have perennial vegetation (buffer strips). Strip-cropping reduces the rate of sediment movement down the slope, where deposition high on the slope is more beneficial than lower down. The principle is that the strip reduces transport capacity of runoff. The quantitative effect of the strips depends on the sediment load generated from the erodible strips relative to the transport capacity on the less erodible strips. Most deposition in strips occurs along the upper edge. Strips loose their effect if they are so wide that rilling occurs, but should be wide enough to allow adequate filtering of the sediment-laden water. Strips might not decrease total runoff much, but discharge it at lower velocity. However, on very long slopes the accumulation of water might be too much to handle with strip cropping (Troeh et al., 1991).

To prevent wind erosion, shelterbelts are often used. They should be placed at right angles to the major wind direction. If wind direction is very variable, grids of shelterbelts might be necessary. The belt should not be so dense that air cannot move through it. On the other hand, it should be dense enough to result in a large decrease of wind speed. Shelterbelts not only reduce wind erosion, but can also protect livestock and reduce fuel use if they are used to

protect houses (Troeh et al., 1991). Windbreak effectiveness extends as far leeward as 15 to 20 times the height of the windbreak, and windward for about twice its height (Troeh et al., 1991).

7.3.5 Summary of effect SICS

Many soil conservations can be seen as part of SICS. There cannot be any doubt that generally speaking conservation measures are effective against soil erosion. There is a multitude of studies that have proved this for a variety of SLM measures, as e.g. summarised by Lal (1990). Although many of these studies were performed on erosion plots, which cannot fully replicate field conditions, there is just too many data to deny the effectiveness of such measures, if properly maintained. In many cases, properly maintained measures were found to decrease erosion rates by at least an order of magnitude, although generally the effectiveness of conservation measures decreases with increasing magnitude of the erosive event. Combinations of several types of SLM are usually more effective than single conservation measures.

However, quantification of the effectiveness of conservation measures and SICS with respect to erosion reduction is difficult for several reasons:

- Effectiveness of measures will vary between erosion events (Hessel and Tenge, 2008), depending e.g. on the size of the events, the intensity of the event, and the sequence of events (antecedent conditions, e.g. soil moisture content at start of the event)
- Effectiveness of measures will vary between fields (Hessel and Tenge, 2008), even if these fields appear to be identical regarding SICS and site conditions.
- It depends on environmental conditions, but information on these conditions is often not complete (Maetens et al., 2012). Environmental conditions (geomorphology, soil, climate, geology) are also highly variable in space
- It may change over time. For example, Maetens et al (2012) found that reduced tillage and contour tillage effects tend to decrease over time, although to a stronger degree for runoff than for soil loss

To illustrate this variability, the Table 7.1 summarises some literature data relevant for eastern Africa. It shows that there is large variability, but nevertheless two general conclusions can be drawn: 1) All measures reduced runoff and soil loss, and 2) soil loss was reduced more than runoff.

Table 7.1. *Effects of different measures on runoff and soil loss. Based on Hessel & Tenge (2008), who consulted a number of publications (see their publication for references). Values are expressed as the value in a field with measure divided by the values in a field without measures. Hence, values below 1 indicate that erosion is decreased by the measure. The range of reported values is given here; original values can be found in Hessel and Tenge (2008).*

Measure	Effect on runoff	Effect on soil loss
Terraces with grass (Fanja Yuu and bench terraces)	0.20-0.67	0.04- 0.13
Grass strips (buffer strips)	0.31-0.85	0.07-0.60
Strip cropping	0.64 (only 1 value found)	0.25-0.65
Mulching	0.15-0.30	0.01-0.24
Minimum tillage	0.8 (only 1 value found)	0.44 (only 1 value found)
Contour ploughing	No information found	0.50-1.00

7.4 Conclusions

Erosion refers to the transport of soil particles by water and wind, and the subsequent deposition of the soil particles elsewhere. Erosion may affect food and biomass production directly through removal of seeds and damage to plants, and indirectly through the loss of fertile topsoil. Erosion negatively affects the storage, filtering, buffering and transformation capacity of the soil, and the habitat function. The risk of erosion is high on sloping land, with erodible soil and low soil cover, during heavy rains or strong winds.

Erosion-specific SICS prevent erosion or lower erosion rates. Erosion-specific SICS are water erosion and wind erosion specific, and involve mainly substitution and redesign mechanisms. The substitution mechanisms relate to minimum or zero tillage instead of conventional tillage, and mulching. Using organic manures and green manures improves soil aggregate stability and water holding capacity and thereby lowers soil erodebility. The redesign mechanism relates to the replacement of annual short cycle crops by perennial crops, relay cropping, strip cropping, cover crops, agroforestry, as well as to the management of landscape elements (terracing, contour planting and ridging, planting hedges, permanent cropping strips, field borders, etc.).

Key elements of SICS are:

1. Vegetative covers (including crop rotations, cover crops etc.)
2. Tillage, mulching, soil and water management
3. Structural landscape elements (grass strips, grassed waterways, alley farming etc.)

Most promising erosion-specific SICS are highly site (morphology), climate (high rainfall areas) and soil specific. Erosion-specific SICS involve a whole range of actions, including a permanent groundcover (crops, mulches), reduced tillage, contour ridging, terracing, drainage,

agroforestry (Table 7.2), which in general also have a positive impact on soil carbon sequestration, landscape appearance and resource use efficiency.

Table 7.2. *Qualitative assessment of erosion-specific SICS (+ indicates significant positive effect; - indicates significant negative effect; -/+ indicates variable effect; no scores means no functional effect).*

	Components of cropping systems	Components of Erosion-specific SICS	Change in profitability	Changes in soil properties		
				Physical	Chemical	Biological
A	Crop rotations	Permanent cropping or +inter/relay/cover cropping +strip cropping, agroforestry	-/+[10]	+	+/-	+
B	• Nutrient management	Optimal				
C	• Irrigation management	Optimal; no flood irrigation				
D	• Drainage management	optimal				
E	• Tillage management	Reduced & contour tillage	+	+	+/-	+
F	• Pest management	Optimal				
G	• Weed management	Optimal				
H	• Residue management	Mulching	+/-	+	+/-	+
J	• Mechanization management	Contour traffic	-/+	+	+/-	+
K	• Landscape management	Agroforestry, terracing, contour treelines	+	+	+/-	+

7.5 References

Bagnold, R.A., 1941. The Physics of Blown Sand and Desert Dunes. Methuen & Co., London, 265 pp.

Cerdan, O.,G. Govers, Y. Le Bissonnais., K. Van Oost, J. Poesen, N. Saby, A. Gobin, A. Vacca, J. Quinton, K. Auerswald, A. Klik. F.J.P.M. Kwaad, D. Raclot, I. Ionita, J. Rejman, S. Rousseva, T. Muxart, M. J. Roxo, T. Dostal. 2010. Rates and spatial variations of soil erosion in Europe: A study based on erosion plot data. Geomorphology 122:167–177.

Dexter, A.R., Bird, N.R.A. 2001. Methods for predicting the optimum and the range of soil water contents for tillage based on the water retention curve. Soil & Tillage Research 57 (2001) 203-212.

Edwards, G., White, D.R., Munkholm, L.J., Sørensen, C.G., Lamandé, M. 2016. Modelling the readiness of soil for different methods of tillage. Soil & Tillage Research 155 (2016) 339–350

Ferreira, A.J.D., Coelho, C.O.A., 2001. Reflexão sobre alguns métodos e técnicas de campo, usados no estudo dos processos erosivos actuais. In Metodologias de estudo de processos de erosão dos solos, (Reflection on methods and field techniques, used to study erosion processes. In Methodologies for studying soil erosion processes) FLUP, Porto, pp. 67–84 (in Portuguese).

Funk, R., Skidmore, E.L. & Hagen, L.J. (2002) Comparsion of wind erosion measurements in Germany with simulated soil losses by WEPS. Proceedings of the ICAR5/GCTE-SEN Joint Conference, Lubbock, Texas, pp 235-238

[10] Change in profitability strongly depends on the accounting period; in the short term costs may exceed benefits, due to the costs involved, change in crop types, and smaller cropping area; on the longer term benefits may exceed costs.

Holland, J.M. 2004. The environmental consequences of adopting conservation tillage in Europe: reviewing the evidence. Agriculture, Ecosystems and Environment 103:1–25.

Hessel, R., A Tenge. 2008. A pragmatic approach to modelling soil and water conservation measures with a catchment scale erosion model. Catena 74, 119-126

Kalma, J. D., Speight, J. G. and Wasson, R. J. 1988. Potential wind erosion in Australia: A continental perspective. J. Climatol., 8: 411–428. doi:10.1002/joc.3370080408

Keesstra, Saksia, Paulo Pereira, Agata Novara, Eric C. Brevik, Cesar Azorin-Molina, Luis Parras-Alcántara,AntonioJordán, ArtemiCerdà. 2016 Effects of soil management techniques on soil water erosion in apricot orchards. Science of the Total Environment 551–552; 357–366

Lal, R., 1990. Soil erosion in the tropics: principles and management. McGraw-Hill, New York

Lal, R. 2001. Soil degradation by erosion. Land Degrad. Dev., 12: 519–539. doi:10.1002/ldr.472

Liniger, H., Critchley, W. (Eds.), 2007. Where the land is greener. Case studies and analysis of soil and water conservation initiatives worldwide. World Overview of Conservation Approaches and Technologies (WOCAT), Bern.

Maetens, W., Poesen, J. and Vanmaercke, M., 2012. How effective are soil conservation techniques in reducing plot runoff and soil loss in Europe and the Mediterranean? Earth-Science Reviews 115(1-2): 21-36. doi:10.1016/j.earscirev.2012.08.003

Martins, J.C., F.P. Pires, A.V. Oliveira, A. M. Campos, C. Horta, M.G. Serrão, M. L. Fernandes. 2007. Experimental evaluation of runoff and erosion in an Haplic Luvisol of Mértola: impacts of fertilization in pastures (in Portuguese). Avaliação do escoamento e erosão num luvissolo, Revista de Ciências Agrárias ISSN 0871-018X. 30:2, p. 278-290.

Morgan, R.P.C., 2005. Soil Erosion and Conservation, third ed. Blackwell Science Ltd., 304ISBN: 1-4051-1781-8.

Munkholm, L.J. 2011. Soil friability: A review of the concept, assessment and effects of soil properties and management. Geoderma 167-168 (2011) 236–246.

Nielsen, Uffe N., Wall, Diana H., Six, Johan. 2015. Soil Biodiversity and the Environment. Annual Review of Environment and Resources 2015 40:1, 63-90

Oldeman, L.R. 1994. Global extend of soil degradation. Published in: ISRIC Bi-Annual Report 1991-1992, pp 19-36.

Poesen, J., J Nachtergaele, G Verstraeten, C Valentin (2003) Gully erosion and environmental change: importance and research needs. Catena 50: 91-133

Poesen, J. (in press) Soil erosion in the Anthropocene: research needs. Earth Surface Processes and landforms. Doi: 10.1002/esp.4250

Prosdocimi,M., Cerdà, A., Tarolli, P. 2016. Soil water erosion on Mediterranean vineyards: A review. Catena 141:1–21.

Ramos, T.B.; Martins, J.C.; Pires, F.P.; Luz, P.B.; Castanheira N.L., Reis, J.L.; Santos, F.L.. 2010 Tillage techniques impact on soil conservation and on maize yield irrigated by center pivot in a Fluvisol. Revista de Ciências Agrárias. ISSN 0871-018X. 33:1, p. 247-256. (in Portuguese)

Renard, K.K., Foster, G.R., Weesies, G.A., McCool, D.K., Yoder, D.C. 1997. Predicting soil erosion by water: A guide to conservation planning with the revised Soil Loss Equation (RUSLE). U.S. department of Agriculture, Agriculture Handbook No. 703, 404 pp.

Pittelkow, C.M., Linquist, B.A., Lundy, M.E., Liang, X., Van Groenigen, K.J., Lee, J., Van Gestel, N., Six, J., Venterea, R.T., Van Kessel, C. 2015. When does no-till yield more? A global meta-analysis. Field Crops Research 183:156–168

Shao, Y. 2008. Physics and modelling of wind erosion. Atmospheroc and Oceanographic Sciences Library 37, Springer, ISBN 978-1-4020-8894-0.

Sheehy, Jatta; Regina Kristiin;, Alakukku, Jaura; Sixb, Johan. 2015. Impact of no-till and reduced tillage on aggregation and aggregate-associated carbon in Northern European agroecosystems. Soil & Tillage Research 150:107–113.

Silva, L.L.; da Silva, J.R.M.; da Luz, P.B. 2005. Avaliação Qualitativa do Risco de Escorrimento e Erosão na Rega por Rampa Rotativa utilizando Sistemas de Informação Geográfica. (Qualitative Evaluation of

SoilCare
SOILCARE FOR PROFITABLE AND SUSTAINABLE CROP PRODUCTION IN EUROPE

Drainage and Erosion Risk in Rotating Ramp Irrigation using Geographic Information Systems). I Congresso Nacional de Rega, 5-7 December, Beja, Portugal. (in Portuguese)

Seufert V, N Ramankutty, JA Foley (2012). Comparing the yields of organic and conventional agriculture. Nature 485, 229-232

Suomi, P., Oksanen, T. 2015. Automatic working depth control for seed drill using ISO 11783 remote control messages. Computers and Electronics in Agriculture 116:30–35

Tilman, D, Cassman, K.G., Matson, P.A., Naylor, R., Polasky, S. 2002. Agricultural sustainability and intensive production practices. NATURE, VOL 418. doi:10.1038/nature01014.

Troeh, F.R., Hobbs, J.A. and Donahue, R.L. (1991) "Chapter Six: Predicting Soil Loss" Soil and Water Conservation, 2nd ed. Prentice-Hall, Englewood Cliffs, New Jersey, USA.

Toy, T.J., Foster, G.R., Renard, K.G. 2002. Soil erosion: Processes, prediction, measurement, and control. John Wiley and Sons, Inc. 338 p.

Van den Putte, A., Govers, G.,, Diels, J.,,Gillijns, K., Demuzere, M. 2010. Assessing the effect of soil tillage on crop growth: A meta-regression analysis on European crop yields under conservation agriculture. Europ. J. Agronomy 33:231–241.

Woodruff, N.P. and Siddoway, F.H., 1965. A Wind Erosion Equation. SSSAJ Vol. 29 No. 5, p. 602-608

8 Soil-improving cropping systems for soil compaction

A. Alaoui, A. Berti, G. Edwards, O. Green, T. Seehusen and P. Yang

8.1 Background

Soil compaction has been defined as "*The densification and distortion of soil by which total and air-filled porosity are reduced, causing deterioration or loss of one or more soil functions*" (Van den Akker, 2008). Soil compaction affects physical soil condition (aeration and water movement in soil, soil mechanical resistance, water availability), can greatly affect plant growth, cause severe yield loss (Lebert et al., 2004; Voorhees 2000) and extra costs for farmers (McGarry 2003). Compaction beneath the ploughed layer, due to use of heavy machinery under unfavourable conditions, may be nearly permanent since the effects of freezing, drying and biological activity are limited and techniques to remediate compacted subsoils are scarce (Lebert et al., 2007). Because of its persistence, subsoil compaction is therefore a long term threat to soil productivity and thereby also an ethical aspect (Håkansson 1994; Håkansson et al., 1987). Soil compaction is one of the most important factors in soil physical degradation (Pagliai et al., 2003), affecting ca. 68 Mha worldwide, of which more than half of it is in Europe (Hamza and Anderson 2005). The vast majority of soil compaction in modern agriculture is caused by vehicular traffic (Soane and Van Ouwerkerk, 1994, Flowers and Lal, 1998). Increasing costs of production lead to increasing economic pressure and the increase in machinery size (Flower and Lal, 1998). In addition, trampling by animals, reduced use of organic matter, frequent use of chemical fertilizers and ploughing at the same depth for many years seem conducive to soil compaction.

The sensitivity of soils to compaction depends on soil properties such as texture and moisture, organic carbon content, and on several external factors such as climate and land use (Jones et al., 2012). At the European level, based on SPADE-8 database (Koue et al., 2008), nearly 29% of the profiles of non-organic soils reached critically high densities (Stolte et. al, 2015, Schjønning et al., 2015). The extend of soil compaction depends upon technical factors (e.g. type of machinery, wheel load, tyre contact area), natural factors (e.g. moisture content) and management related factors (e.g. number of passes) (Alakukku et al., 2003).

8.2 Purpose

The review aims to review literature related to soil-improving cropping systems that prevent, mitigate or remediate the impacts of soil compaction; which strategies and agronomic techniques can be used to remediate compacted soils and ameliorate soil quality?

8.3 Results and Discussion

8.3.1 Main factors affecting soil compaction

Soil compaction is strongly related to soil and environment conditions and to the type of exploitation of the agricultural land. Critical factors for soil compaction are:

- Soil water content
- Soil texture and movement of fine clay from top soil to subsoil
- Traffic and tillage
- Trampling by animals

a) Soil water content

Soil moisture status is an important factor influencing soil compaction processes (Soane and van Ouwerkerk, 1994). Soils become stronger with aggregation and their susceptibility for compaction is dependent upon water content (Horn and Rostek 2000). Zhang et al. (2006) applied two treatments, C1 and C2 (corresponding to an increase in bulk density by 10 and 20%, respectively) to two silty loam sites, Heyang (Chromic Cambisol) and Mihzi (Calcic Cambisol). They found that water retention curves for both the surface (0–0.05 m) and subsurface (0.10–0.15 m) layers at the two sites were significantly changed by tested levels of soil compaction. They found that a high level of compaction (C2) significantly decreased the water content of the surface layer at tensions of <2 kPa for Heyang and ≤8 kPa for the Mihzi site.

The structure of soils can also change through drying and wetting cycles (hydraulic stress) (Peng et al., 2007). In non-rigid soils, two shrinkage components with vertical and horizontal directions can be quantified. Vertical shrinkage results in soil subsidence, while horizontal shrinkage produces soil cracks. Pre-existing soil cracks induce surface water to reach more directly the water level as preferential flow (Liu et al., 2003) and potentially lead to unintended contamination of ground- and surface waters by agrochemicals (e.g., Borggaard and Gimsing, 2008; Jarvis, 2007).

Hassan et al. (2007) in the sub-humid region of Pakistan found that the effect of the same intensity of soil compaction on wheat yield was significantly higher in a dryer and warmer year than in better climatic conditions, due to a reduction on water and nutrient uptake. Alblas et al., 1994) measured on a sandy soil in The Netherlands in a dry year a reduction of 38% in silage maize yield due to a reduced rooting depth caused by subsoil compaction. On the other hand, in cooler or wetter areas the effect of compaction can be less evident. In rainy Scotland, despite the presence of a dense impermeable layer at depths of 40–50 cm, the yields of arable crops are little affected in most seasons (Batey, 2009), because the summer crops can obtain enough water from the restricted rooting depth.

The interaction with climate may account for the perception of severity of soil compaction. While in dry conditions the perceived effects are mainly related to water stress and can be partially masked by irrigation, in wetter climates the adverse effects of compaction are mainly related to reduced drainage and water logging and erosion.

In the northern latitudes (e.g. Norway) the growing season is shorter, the soil remains longer at field capacity in spring and returns earlier to field capacity in autumn than in other regions. Late harvest due to unfavourable conditions can lead to reduced cereal yields and quality (Sander et al., 1987; Sogn and Hauge 1976), while performing field operations under suboptimal circumstances may lead to a high risk of damaging the soil structure also in the subsoil (Botta et al., 2002; Hamza and Anderson 2005; Raper 2005).

b) Movement of clay from top soil to subsoil

Compaction may also be found at considerable depth under natural situations (Batey, 2009). In non-calcaerous soils, fine clay particles may be dispersed in the top soil and move downward where they flocculate and may form a dense B2 horizon (Sullivan & Montgomery, 1998). In Vertisols, movements of finely aggregated topsoils through cracks during tillage operations or intense water infiltrations, can lead to subsoil compaction, especially when the soil rewets and expands.

c) Tillage and traffic

Direct effects of soil tillage on soil compaction are questionable and contradictory results had been obtained when tillage systems effects on soil bulk density had been reviewed (Alvarez and Steinbach, 2009). The direct effects of ploughing are mostly related to the formation of dense sub-surface soil horizons when the same ploughing depth is used in subsequent years. This can be managed through deep ripping or deep cultivation (Hamza and Anderson, 2005). However, most of the effects of soil tillage are indirect, related to the high number of passes and to the high wheel load of machinery, to the effects on stability of soil structure and to the effects on the dynamic of soil organic matter and on soil water retention.

The ground area trafficked with heavy machinery can exceed 100% during a single cropping cycle in conventional systems with multiple passes (Soane et al., 1982); in reduced tillage systems the percentage of area subjected to traffic is reduced (ca. 60%) but, even in genuine no-till systems with one pass at sowing, the trafficked area can be over 30% (Tullberg, 1990).

Soil loosening is one common solution to alleviate compaction and to improve soil structure. But loosening, especially of the subsoil, is both labour and energy intensive (Botta et al., 2002; Wolkowski 1990). The original, natural soil structure is complicated to rebuild and soil tillage may result in a soil structure that is inferior since the aggregates become more blocky and less porous (Arvidsson and Håkansson, 1996; Horn et al., 1995) and the pore functions, once deteriorated, are hardly renewable (Horn and Fleige 2009). After loosening, the aggregate

stability can be lower and the soil can be more sensitive against compaction and may re-compact quickly after loosening (Chamen et al., 2003; Chen and Weil 2010; Spoor 2006). Soil loosening should therefore ideally be a combination of mechanical (tillage) and biological (plant roots) measures to rebuild the soil structure.

Clay soils may be readily compacted through heavy machinery. However, compacted clay soils may regenerate more easily and rapidly than compacted sand and sandy loams, through drying-wetting and in the top layer freeze-thaw cycles. However, subsoil compaction proves to be at least partly very persistent (Berisso et al., 2012)

d) Trampling by animals

Soil structure may be significantly altered due to either mechanical stress (e.g., tractor traffic, tillage and cattle trampling) or hydraulic stress from natural wetting and drying cycles. Stock trampling affects soil in different ways, depending on several conditions: (i) trampling intensity; (ii) soil moisture (iii) soil type; (iv) plant type; (v) field slope and (vi) land use type (e.g., Zhao et al., 2010; Krümmelbein et al., 2006). Trampling-induced soil compaction is characterized by its spatial heterogeneous distribution. It mainly affects pore geometry (or structure) at the soil surface (Nie et al., 2001; Vzzotto et al., 2000) and topsoil matrix (Alaoui and Helbling, 2006) and may degrade the ecological status of the soil by reducing the number of earthworms which improve infiltration (e.g., Hills, 1971). The depth of soil compaction induced by pugging depends on animal weight, soil moisture, hoof size and kinetic energy.

The major trend concerning pastoral agriculture is the exponential increase in stocking numbers and densities and this has been correlated with changing flood risk. For example, in Wales, 72 per cent of agricultural land was estimated to be under grassland production in 2005, almost exclusively to support sheep farming. Sheep numbers in the UK increased from 19.7 million in 1950 to 40.2 million in 1990 while they were only about 8 million in the 1860s (Fuller and Gough, 1999). Such changes have been correlated with runoff and flow regimes in the River Derwent (Evans, 1996) where sheep numbers doubled between 1944 and 1975, and which coincided with an increased runoff rate of 25%. Similarly, increasing flow peaks in the upper catchment of the River Lune was qualitatively related to this increased stock densities (Orr and Carling, 2006). Within the Yorkshire Ouse catchment, over 40% of sites investigated after the autumn 2000 floods had high soil degradation, and this was estimated to have caused an increased runoff rate of between 0.8% and 9.4% (Holman et al., 2003). Heathwaite et al. (1989) found that 7% of rainfall was converted to runoff in ungrazed fields, while this increased to 53% in grazed fields. Furthermore, Heathwaite et al. (1990) found that infiltration capacity was reduced by 80% on grazed areas compared to fields with no stock. Stock reduces the vegetation cover, which may lead to soil surface crusting and reduced overland flow resistance (Ferrero, 1991). It may also lead to a decrease in the evapotranspiration or lead to the partitioning of water into the slower subsurface through flow pathways (Owens et al., 1997).

All these processes may impact upon runoff generation and, possibly, downstream flood risk (Pattison and Lane, 2011).

8.3.2 Indicators for soil compaction

The effect of soil compaction on soil structure can be assessed with several types of parameters: bulk density and total porosity (Boone, 1988; da Silva, Kay, & Perfect, 1994), macroporosity (Alakukku, 1996), penetration resistance (Pagliai, 1998), air permeability (Ball, 1981; Reszkowska, Krümmelbein, Gan, Peth, & Horn, 2011), saturated hydraulic conductivity (Alakukku, 1996), pre-consolidation pressure (Horn, 1981; Kirby, 1991), dye surface density (Kulli, Gysi, & Flühler, 2003; Alaoui & Helbling, 2006), and infiltration capacity (Alaoui & Helbling, 2006; Blanco-Canqui, Claassen, & Stone, 2010).

The most-used ones are soil cohesion, soil structural strength, bulk density, water potential, and pre-compressive stress. However, these parameters are general indicators that integrate information about the total change in the volume of voids of soil under consideration, but they cannot account for changes in the volume distribution of these voids, their connectivity, or the changes in this connectivity (e.g., Vogeler, Horn, Wetzel, & Kruemmelbein, 2006). To overcome this problem, the actual bulk density is expressed as a percentage of the reference-compaction state of a given soil known as "degree of compactness" or "relative compactness" (Håkansson, 1990; Håkansson and Lipiec, 2000). In the same way, the pore space can be quantified by the void ratio frequently used in soil mechanics and soil physics and defined as the volume of the pores per unit volume of solid. The fact that the denominator is constant enables the void ratio of different types of pores to be compared, even in soil where pore space may vary with shrinkage/swelling processes or under compaction/shearing (e.g., Dexter et al., 2008). Zhang et al. (2006) also used water volume ratio expressed as the volume of water per unit volume of solid phase, which does not depend on the changes in soil bulk density and is appropriate variable to use for swelling soils. These relative compaction parameters are more useful than bulk density or total porosity in studies of the effects of field traffic on soil structure and consequently on root and crop response (Canarache, 1991; Håkansson and Lipiec, 2000). By using the relative compaction instead of the bulk density performance and applying the concept of the least limiting water range, LLWR (defined as the ideal soil water content range, in which the limitations for root growth were due to the availability of water, air, and PR were minimal) are enhanced (da Silva et al., 1997).

However, not all soil parameters that are affected by compaction have necessarily negative influence on soil function and plant growth (Horn and Fleige 2009) but may have negative influence on the ecosystem itself. Anyhow the effect of compaction may also depend on the weather conditions (Alakukku 2000) and a certain degree of compaction may increase plant growth due to better soil- root contact. Sensitivity against compaction in general differs among crops. Cereals, especially wheat and barley are comparatively insensitive against compaction, whereas crops like peas and oil seed are more sensitive (Arvidsson et al., 2012). Especially crops

with their yield organs in the soil as potato are sensitive and compaction may not only hamper growth but also misshapen yield organs.

8.3.3 Soil-improving cropping systems for soil compaction

a) organic matter management

Organic matter affects soil compactibility mainly through the binding effect on soil mineral particles (Theng and Oades, 1982; Zhang, 1994), reducing of aggregate wettability (Zhang and Hartge, 1992) and increasing the mechanical strength of soil aggregates. The effects are anyway strongly dependent on type of organic amendments, on soil type and environmental conditions, such as temperature and soil moisture (Hamza and Anderson, 2005). The effects are normally evident in topsoil, where incorporation of plant residues and manure applications is normally done. Organic materials normally have lower bulk density and greater porosity than mineral soils, thus their addition to soils would improve soil bulk density and porosity. Furthermore, the elasticity of manure prevents the transmission of the stresses toward the subsoil in the lower depths (Soane, 1990) thus acting as a buffer to decrease the impact of farm machinery on subsoil.

The incorporation of OM should also foresee for improving subsoil compaction, but the injection of organic material into the rooting zone requires deeper tillages, thus increasing costs and the mechanical effect of tractor tyres.

b) controlled traffic and management of trafficability

Controlled traffic farming (CTF) is a management strategy to minimise traffic-induced soil compaction, which is being implemented worldwide (Raper, 2005). CTF is defined as a "crop production system in which the crop zone and the traffic-lanes are distinctly and permanently separated" (Taylor, 1983) using in-field machinery equipped with navigation-aids and auto-steering systems (Bochtis and Vougioukas, 2008; Raper, 2005). Gasso et al. (2013), reviewing the available literature on environmental effects of CTF, found that this approach can lead to an increase of crop yields associated to a consistent reduction on GHG emissions (particularly for methane and nitrous oxide) and on water runoff and has a positive indirect impact associated with use of fertilisers, pesticides, seeds and fuels.

Trafficability is defined as the ability of the soil to support and withstand traffic, causing only minimal or reversible structural damage (Rounsevell and Jones, 1993). Whereas CTF aims to confine damage causing by trafficking to specific areas, managing trafficability aim to modify vehicle operational setups to limit the applied pressure or limit any field traffic to specific time windows. To estimate the potential risk of an operation causing compaction of the soil a common technique is to compare the stresses caused by trafficking to the soil strength (Schjønning et al., 2012). Söhne (1953, 1958) first suggested a simple analytical model for the stress propagation within the soil profile based on the work of Boussinesq (1885) and Fröhlich (1934). An important input into the stress propagation model is the boundary conditions at the

soil-tyre interface, (Keller, 2005). Schjønning et al., (2008) suggested a further model, referred to as FRIDA, which describes the stress distribution in the tyre foot print. The model can be parameterised using the physical description of the tyres, i.e. tyre width, tyre section height, tyre rim diameter, and tyre pressure, (Schjønning et al., 2006). This model, coupled with the stress propagation model, has been tested against measurements made using in-situ sensors placed within the soil profile (Lamandé and Schjønning 2011, Lamandé and Schjønning 2008, Keller et al., 2007).

Models of the driving forces acting at the soil-tire interface related to slip (Steiner, 1979; Osetinsky & Shmulevich, 2004), to the deformation of the tires and the topsoil during tractive performance (Schwieger, 1996), or to the management of the energy requirement of the tractor (Pichlmaier, 2012) described the stresses and deformation distribution of soil and tire but did not provide direct information useful to the practitioner.

Based on field and laboratory studies, computerized simulations emerged as appropriate tools to evaluate the effects of heavy-load machines on soil compaction. They included mapping on a national scale with the Soil Compaction Model (SOCOMO) (van den Akker, 1997, 2004) or on a plot scale (Diserens, Chanet, & Marionneau, 2010) and can be designed as practical applications such as Terranimo (Stettler et al., 2010) and Tyres/tracks And Soil Compaction (TASC) (Diserens & Spiess, 2004). A practical forecasting module integrated into TASC V3.0 (Battiato, 2014; Diserens & Battiato, 2013) calculates the slip-rate limit beyond which the topsoil failure occurs with the corresponding traction force.

Both tools TASC and Terranimo (Stettler et al., 2014) are web-based tools used to estimate the trafficability of soil during field operations. The tools estimate the stress distribution in the soil-tyre interface using the wheel loads and tyre characteristics and compares how this stress is propagated through the soil profile with an estimation of the pre-compression stress. If the stress caused by the wheel loading is below the pre-compression stress then the soil is said to be trafficable. The interface is built such that these calculations are not shown to the user, rather a more simplistic representation is used indicating the risk of compacting the soil as either low, moderate or high. In this way farm managers can gauge the best conditions in which to execute operations.

The spatial variance of soils on a field level can result if different parts of the fields being trafficable at different times. Edwards et al. (2016) showed how spatial variance can affect a field's overall trafficability, and how operations should be limited to specific windows when the entire field is ready. An alternative approach can be applied if the weight of the vehicle varies during the operation, when the route of the vehicle may be managed to limit soil damage. Bochtis et al. (2011) proposed a decision support system which took a soil strength map as an input and then planned a slurry operation, such that the vehicle drove over the strongest parts of the field when it was heaviest and the weakest parts of the field when it was lightest. In this

way, the operational time windows can be further expanded to allow for more in field operation time.

c) tillage measures

The main physical way for eliminating soil compaction is deep ripping or deep cultivation. This type of tillage has become a widely used management technique, allowing the destruction of hard pans and ameliorating hard setting soils (Hamza and Anderson, 2005). The effect on crop yield is, however, strongly variable: Mark and Soane (1987 – cited in Chamen et al., 2015) assessed crop yield responses to subsoil loosening on 25 sites on a wide range of soils in UK. In spring crops, six showed an increase of yield but four resulted in a negative response. Of the 17 winter crop sites, none provided a positive response while four resulted in decreased yields.

The effects of loosening operation can mainly be related to the amount of subsequent traffic and to the distribution of binding or flocculating agents, such as organic matter and gypsum. After the tillage, the open soil condition is particularly vulnerable to re-compaction by subsequent traffic (Spoor, 1995), and precipitation of fine clay and colloids through wetting–drying cycles can favour recompaction in clay soils (Hamza and Anderson, 2005). A further aspect to be considered are the effects of mechanical loosening on earthworms and other components of soil macrofauna. Lees et al. (2016) showed that both earthworm number and biomass were markedly reduced with mechanical loosening and that the effect lasted for at least two years from tillage. Earthworms can contribute significantly to soil regeneration after compaction (Langmaak et al., 2002), in particular increasing infiltration rate (Capowiez et al., 2012) and leaving pores that can be subsequently used by roots to increase their penetration in soils.

It is worth noting that cultivation pans play also a role in protecting subsoil from compaction by spreading the stress (e.g. wheeling) over a wider area (Chamen et al., 2003; Spoor et al., 2003; Wiermann and Horn 2000). These pans need to be disturbed only if they effectively reduce root growth, soil aeration and drainage (Spoor et al., 2003). When soil loosening is needed, these Authors recommended creating fissures or cracks through compacted zones to restore rooting and drainage, rather than massive disruption, with minimal loss of bearing capacity and leaving to subsequent biological and weathering activity to complete the remediation process.

d) rotational effects.

The evolution of compaction during cropping cycles is related to the type of crop grown and their agronomics and to land conditions during cropping operations. In most situations, the risk of compaction with grain and seed crops is lower than with root crops (Batey, 2009). Most grain crops are harvested collecting only above ground plant parts in the summer period, when soil is frequently dry and firm. Furthermore, the weight of harvested materials to be transported out from the field is relatively small (in the order of 10 t ha^{-1}). On the other hand, many root

crops are harvested later, with some mechanical operation on soils to lift the product and the yield is substantially higher (30-50 t ha^{-1}). This could lead to a higher soil disturbance and to compaction of both top and sub-soil.

Apart from mechanical operation required for cropping, crops show an intrinsic different ability to penetrate soils and to alleviate the effects of compaction through stabilisation of structure and direct effect on compacted soils. Roots with greater diameter (often tap-rooted dicots) are more capable of penetrating compacted soil layers than roots with smaller diameter (usually fibrous-rooted monocots) (Chen and Weil, 2011), however Busscher et al. (2000) showed in soybean that there is a consistent difference between genotypes for root growth in compacted soils, thus suggesting that genetic improvement could potentially reduce dependence on tillage in soils with hard layers.

Diurnal changes in root diameter loosen and break down any compacted soil layer around them. Radish and lupin exhibit diurnal fluctuations in root diameter, in response to the variation of transpiration during the day (Hamza et al., 2001), destabilising soil and loosening compaction.

Roots can then contribute to a ''biological drilling'' (Cresswell and Kirkegaard, 1995), due to the creation of bio-pores tap roots and the subsequent use of these biopores as low resistance pathways by the roots of succeeding crops (Chen and Weil, 2010). Including in the rotation species with a deep tap root system is then an important option to minimize the effects of soil compaction (Ishaq et al., 2001).

In recent years, there was a growing interest on cover crops with a deep and strong root apparatus, which can alleviate soil compaction. Tap-rooted fodder radish and rapeseed cover crops have been shown able to leave deep root channels in compacted soils, enhancing root development of subsequent crops (Williams and Weil, 2004; Chen and Weil, 2011).

8.3.4 Meta-analysis on soil-improving cropping systems for soil compaction

In this paragraph, we summarize the results of a meta-analysis of published effects on soil-compaction-relieving-measures on crop yield, soil bulk density and soil penetration (Yang et al. in prep). Cereals has been chosen as test crops because of their global importance and their dominance in the experimental studies. The objectives of the meta-analysis were (1) to examine the effects of soil-compaction-relieving-measures on crop yields, soil bulk density, and soil penetration, based on results of published studies; (2) to relate variations in the effects of remediation measures to variations in levels of these measures; and (3) to quantify possible interactions between yield changes and soil bulk density/penetration changes.

The dataset consisted of 712 compared yield observations, 514 compared soil bulk density observations and 418 compared soil penetration observations. There were 368 compared observations for wheat, 205 compared observations for maize, 16 compared observations for oat and 123 compared observations for barley.

a) *Organic matter management (residue cover and manure application)*

The management of organic matter inputs mainly affected crop yields (Figure 8.1). In particular, residue cover significantly increased crop yields by 10.5%, while the effect of manure application was not significant, even if the mean effect was similar to that of residues. The absence of a significant effect for manure is probably related to a strong interaction with soil type, with a differential importance of structuration and nutritional effects depending on texture and nutrient status of the soil.

Both residues and manure had no significant effects on soil bulk density or soil penetration (Figure 8.2).

b) *Controlled traffic and management of trafficability*

Compared with random traffic, controlled traffic has significant effects on both crop yields and soil physical properties. It significantly increased crop yield by 38.1% (Figure 8.1) and decreased soil bulk density by 5.9% (topsoil) and 4.4% (subsoil), respectively (Figure 8.2). Controlled traffic also has significant effects on soil penetration, decreasing the required pressure by 37.7% in topsoil and by 29.9% in subsoil (Figure 8.3).

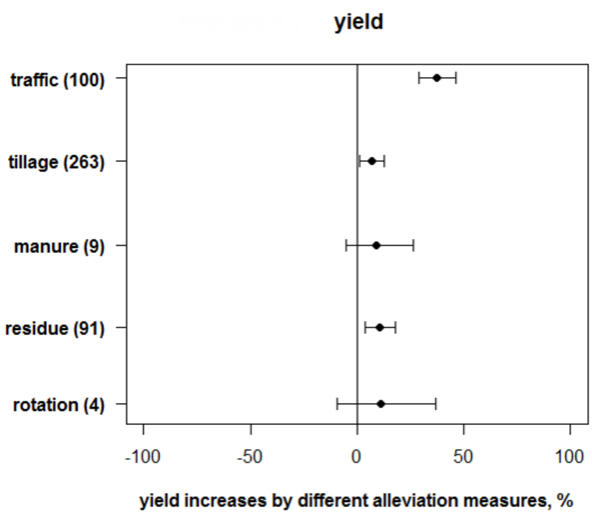


Figure 8.1. *Relative effects of soil compaction alleviation measures on crop yield. Dots show means, error bars represent 95% confidence intervals. The numbers of observations are displayed in parentheses. Traffic = controlled traffic, tillage = effects of overall tillage measures, rotation = crop rotation, residue = residue cover, manure = manure application.*

When partly controlled traffic and totally controlled traffic were analysed separately, their effects on crop yields and soil physical properties are similar, but with varying degrees. Both increased crop yield and decreased soil bulk density and soil penetration. However, only totally

controlled traffic gave relevant effects on the parameters considered, while the effects of partly controlled tillage were not significant, even if showing the same behaviour as totally controlled traffic. As expected, for both totally/partly controlled traffic, effects on soil bulk density and soil penetration were higher in topsoil than subsoil.

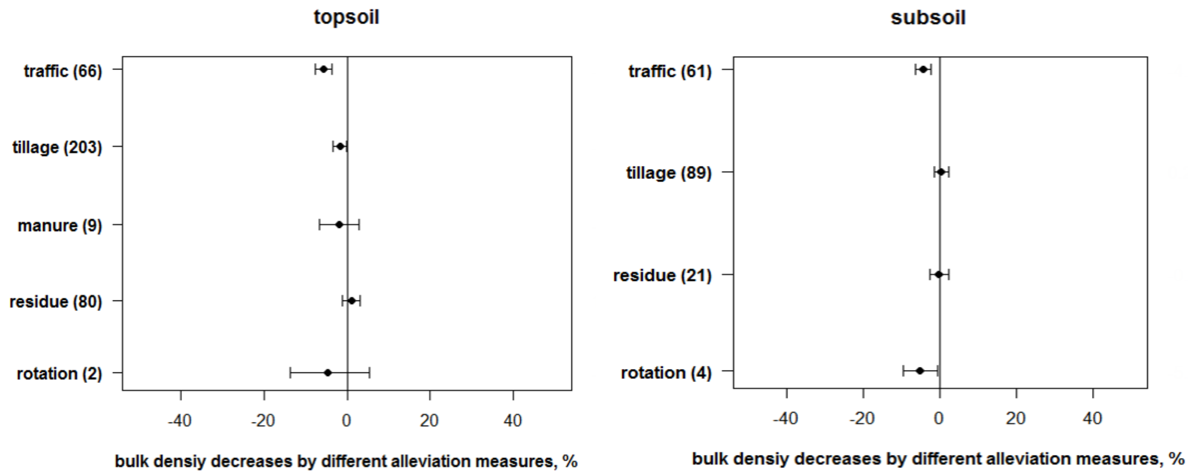


Figure 8.2. *Relative effects of soil compaction alleviation measures on soil bulk density of the top soil and subsoil. Dots show means, error bars represent 95% confidence intervals. The numbers of observations are displayed in parentheses. For legend see Figure 8.1.*

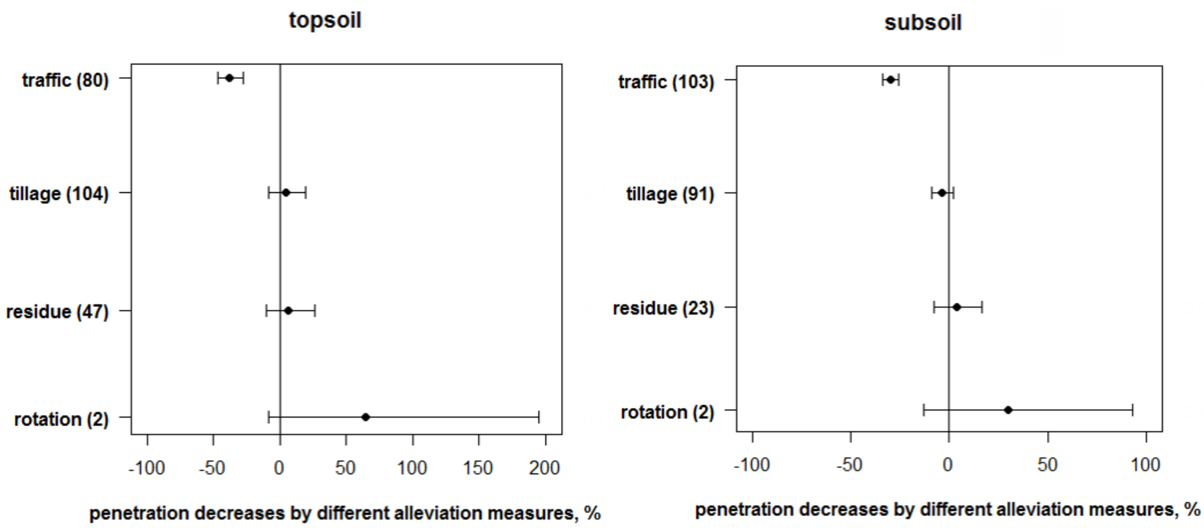


Figure 8.3. *Relative effects of different alleviation measures on soil penetration. Dots show means, error bars represent 95% confidence intervals. The numbers of observations are displayed in parentheses. For legend see Figure 8.1.*

c) Tillage measures

The average effect size of tillage on yields was positive (+6.3% - Figure 8.1) but no significant effects on either soil bulk density or soil penetration were observed (Figure 8.2; Figure 8.3). Tillage measures could be divided into 3 groups (Figures 8.4, 8.5 and 8.6), depending on their effects on crop yields and soil physical properties.

- Positive effect group: both deep tillage and subsoiling significantly increased crop yields, by 7.5% and 8.0% respectively. Deep tillage significantly decreased top soil bulk density by 8.5% while subsoiling significantly decreased subsoil bulk density by 3.6%. Their decreasing effects on both topsoil and subsoil penetration are all significant, varying from 10.2% to 24.2%.
- Negative effect group: no tillage and minimum tillage have negative effects on both soil bulk density and soil penetration, but their effects on crop yield are not significant. Compared to minimum tillage, no tillage has worse effects on soil bulk density (4.2% increase in topsoil) and soil penetration (40.0% and 6.8% in top soil and subsoil respectively). Minimum tillage increased subsoil bulk density by 4.52%, but has no significant effects on topsoil bulk density or soil penetration.
- No significant effect group: harrowing, rotary, and other tillage measures. They have no significant effects on neither crop yield nor soil bulk density/penetration.

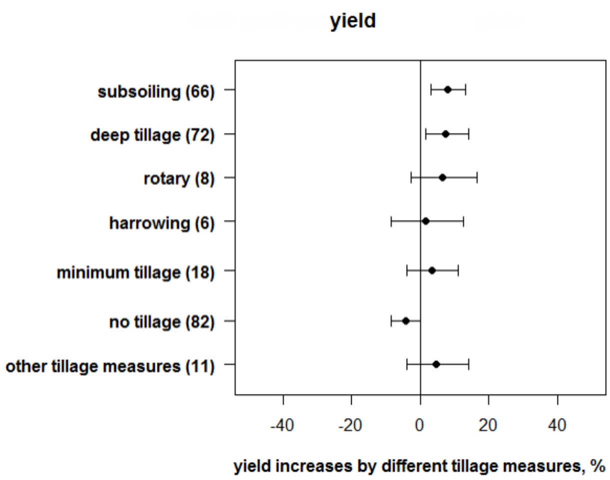


Figure 8.4. *Relative effects of various tillage measures on crop yield. Dots show means, error bars represent 95% confidence intervals. The numbers of observations are displayed in parentheses.*

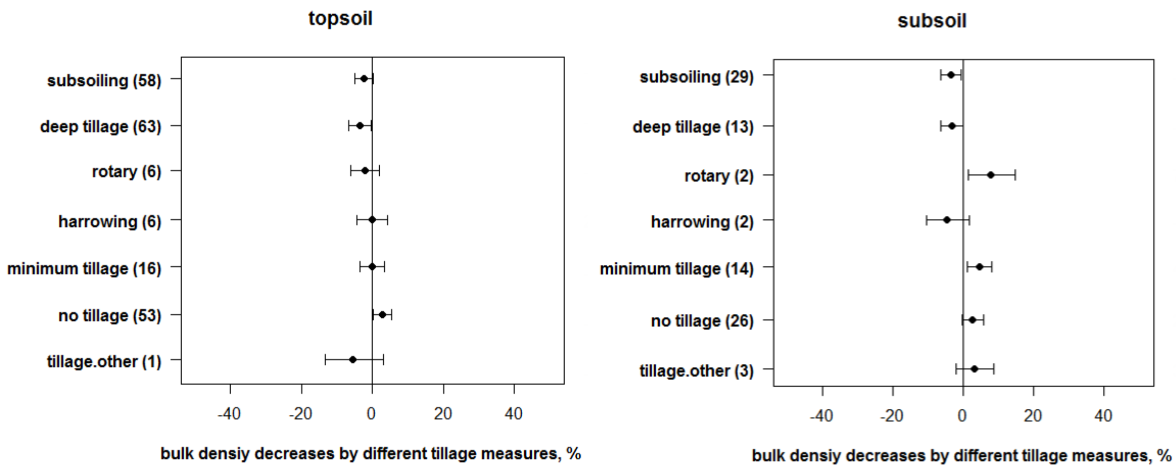


Figure 8.5. *Relative effects of different tillage measures on soil bulk density. Dots show means, error bars represent 95% confidence intervals. The numbers of observations are displayed in parentheses.*

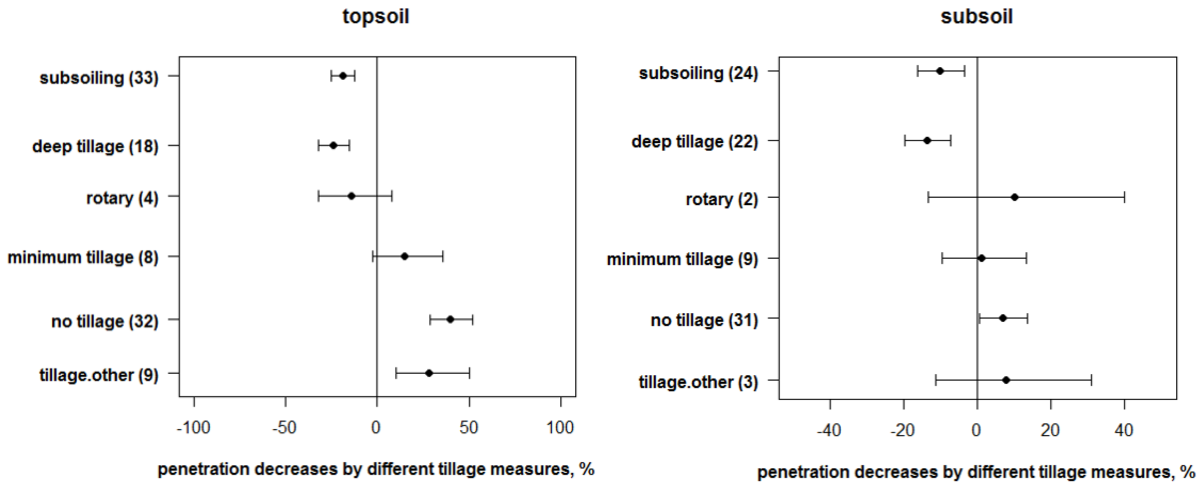


Figure 8.6. *Relative effects of different tillage measures on soil penetration. Dots show means, error bars represent 95% confidence intervals. The numbers of observations are displayed in parentheses.*

d) crop rotation,

Crop rotation tended to decrease subsoil bulk density by 5.2% (Figure 8.2), while the effects on either crop yield or soil penetration were not significant, despite an improvement of the mean of both parameters. It is worth noting that for yield and, in particular, for penetration, the confidence intervals are very wide, indicating that rotation can have locally important effects on soil improvement, but these effects cannot be easily generalised to other soil/climate conditions.

8.4 Conclusions

Compaction refers to the densification of soil and the distortion of soil pores. Soil compaction leads to lower water and air infiltration rates, water logging, risks of anaerobicity, a lower root penetration ability, lower crop yields, poor soil structure, lower biodiversity and biological activity, increased greenhouse gas emissions and erosion and runoff. Compaction of the subsoil is especially a concern because subsoil compaction is difficult to remediate (through natural processes and/or deep ploughing/soil lifting).

Compaction-specific SICS prevent compaction and/or lower the density of the soil, increase the water infiltration rate, lower the penetration resistance, and improve soil structure. Compaction-specific SICS mainly involve substitution and redesign mechanisms. The substitution mechanisms relate to lowering wheel loads and tyre pressures, and to reduced tillage, avoiding driving in the open furrow during ploughing, and working in the field under proper soil and weather conditions. The redesign mechanism relates to controlled trafficking, the growth of deep rooting crops like cereals (in particular summer cereals), alfalfa, some cabbages and trees. Deep soil cultivation and stimulating biological activity through manuring may alleviate the effects of soil compaction. Apart from controlled traffic, the effects of the other mechanisms are highly variable, thus indicating the strong interaction with the environmental factors. Compaction-specific SICS have then to be tailored locally, selecting and combining the best actions depending on soil, climate and available crops.

Most promising compaction-specific SICS (i) prevent further densification of the (sub)soil, and (ii) remediate compacted soils and/or alleviate their effects. They may involve controlled trafficking, adjusting mechanization and the planning of activities, growing deep rooting crops, and stimulating biological activity through addition of organic matter (Table 8.1). It will decrease flooding, overland flow and resource use efficiency.

Table 8.1. *Qualitative assessment of compaction-specific SICS. (+ indicates significant positive effect; - indicates significant negative effect; -/+ indicates variable effect; no scores means no functional effect).*

	Components of cropping systems	Components of compaction-specific SICS	Change in profitability	Changes in soil properties		
				Physical	Chemical	Biologica l
A	Crop rotations	When possible: +deep-rooting crops	-/+	+	+/-	+
B	•Nutrient management	Manuring	+/-	+	+/-	++
C	•Irrigation management	optimal	+	+/-	+/-	+
D	•Drainage management	optimal	+	+	+/-	+
E	•Tillage management	Reduced tillage	+	+	+/-	+
F	•Pest management	Optimal	+	+/-	+/-	+/-
G	•Weed management	Optimal	+	+/-	+/-	+/-
H	•Residue management	Optimal	+/-	+	+/-	+
J	•Mechanization management	Controlled traffic; low-wheel loads, low-inflation tyres	+ [11]	++	+/-	+

8.5 References

Alakukku, L. 1996. Persistence of soil compaction due to high axle load traffic: I. Short-term effects on the properties of clay and organic soils. Soil Till. Res. 37, 211–222.

Alakukku, L., Weisskopf, P., Chamen, W.C.T., Tijink, F.G.J., van der Linden, J.P., Pires, S., Sommer, C., Spoor, G. 2003. Prevention strategies for field traffic-induced subsoil compaction: a review Part 1. Machine/soil interactions. Soil & Tillage Research 73, 145-160.

Alaoui, A. & Helbling, A. (2006). Evaluation of soil compaction using hydrodynamic water content variation: comparison between compacted and non-compacted soil. Geoderma 134, 97–108.

Alaoui, A., Lipiec J. Gerke, H.H. 2011. A review of the changes in the soil pore system due to soil deformation: A hydrodynamic perspective. Soil and Tillage Research, 115-116, 1–15.

Alblas, J., Wanink, F., Van den Akker, J.J.H., Van der Werf, H.M.G., 1994. Impact of traffic-induced compaction of sandy soils on the yield of silage maize in The Netherlands. Soil Tillage Res. 29, 157-165.

Alvarez, R., & Steinbach, H. S. (2009). A review of the effects of tillage systems on some soil physical properties, water content, nitrate availability and crops yield in the Argentine Pampas. Soil and Tillage Research, 104(1), 1–15. doi:10.1016/j.still.2009.02.005

Arvidsson, J., Håkansson, I., C.J., P. 2012. Sensitivity of different crops to soil compaction, Soil compaction - effects on soil functions and strategies for prevention. NJF seminar 448. NJF, Helsinki, Finland, pp. 37-39.

Ball, B.C. (1981). Pore characteristics of soils from two cultivation experiments as shown by gas diffusivities and permeabilities and air-filled porosities. J. Soil Sci. 32, 465–481.

Batey, T. (2009). Soil compaction and soil management - A review. Soil Use and Management, 25(4), 335–345. doi:10.1111/j.1475-2743.2009.00236.x

Battiato, A. (2014). Soil-Tyre Interaction Analysis for Agricultural Tractors: Modelling of Traction Performance and Soil Damage (PhD Thesis, p. 165). Università degli Studi di Padova. Dipartimento di

[11] Controlled traffic has been shown to increase yields significantly, while soil physical properties are improved. However, it requires investments in equipment and machines. On the longer term, benefits seem to outweigh the costs.

Territorio e Sistemi Agro Forestali (T.E.S.A.F.). Scuola di Dottorato di Ricerca in Territorio Ambiente Risorse e Salute (T.A.R.S.) Indirizzo: Tecnologie Meccaniche dei Processi Agricoli e Forestali, Padova, Italy. Ciclo XXVI.

Berisso, F.E., Schjønning, P., Keller, T., Lamande´ , M., Etana, A., de Jonge, L.W., Iversen, B.V., Arvidsson, J., Forkman, J., 2012. Persistent effects of subsoil compaction on pore size distribution and gas transport in a loamy soil. Soil and Tillage Research 122, 42–51.

Blanco-Canqui, H., Claassen, M.M., Stone, L.R. (2010). Controlled traffic impacts on physical and hydraulic properties in an intensively cropped no-till soil. Soil Sci.Soc. Am. J. 74, 2142–2150.

Bochtis, D. D., Sørensen, C. G., & Green, O. 2012. A DSS for planning of soil-sensitive field operations. Decision Support Systems, 53(1), 66-75.

Bochtis, D.D., Vougioukas, S.G. 2008. Minimising the non-working distance travelled by machines operating in a headland field pattern. Biosystems Engineering 101, 1–12.

Boone, F.R. 1988. Weather and other environmental factors influencing crop responses to tillage and traffic. Soil Till. Res. 11, 283-324.

Borggaard, O.K., Gimsing, A.L. 2008. Fate of glyphosate in soil and the possibility of leaching to ground and surface waters: a review. Pest Manag. Sci. 64 (4), 441–456.

Botta, G.F., Jorajuria, D., Draghi, L.M. 2002. Influence of the axle load, tyre size and configuration on the compaction of a freshly tilled clayey soil. Journal of Terramechanics 39, 47-54.

Boussinesq, J., 1885. Application des Potentiels à l'étude de l'équilibre et du Mouvement des Solides élastiques. Gauthier-Villars, Paris, 30 pp.

Busscher, W.J., Lipiec, J., Bauer, P.J., Carter, T.E., 2000. Improved root penetration of soil hard layers by a selected genotype. Commun. Soil Sci. Plant Anal. 31, 3089–3101.

Canarache, A. 1991. Factors and indices regarding excessive compactness of agricultural soils. Soil Till. Res. 19 (1), 145–164.

Capowiez Y, Samartino S., Cadoux S., Bouchant P., Richard G., Boizard H., 2012. Role of earthworms in regenerating soil structure after compaction in reduced tillage systems. Soil Biology & Biochemistry 55: 93-103

Chamen W.C.T, Moxey A.P., Towers W., Balana B., Hallett P.D., 2015. Mitigating arable soil compaction: A review and analysis of available cost and benefit data. Soil & Tillage Research 146: 10–25

Chamen, T., Alakukku, L., Pires, S., Sommer, C., Spoor, G., Tijink, F., Weisskopf, P. 2003. Prevention strategies for field traffic-induced subsoil compaction: a review Part 2. Equipment and field practices. Soil & Tillage Research 73, 161-174.

Chen, G., Weil, R. R. (2010). Penetration of cover crop roots through compacted soils. Plant Soil 331:31–43

Chen, G., Weil, R. R. (2011). Root growth and yield of maize as affected by soil compaction and cover crops. Soil and Tillage Research, 117, 17–27. doi:10.1016/j.still.2011.08.001

Cresswell H.P., Kirkegaard J.A., 1995. Subsoil amelioration by plant-roots - the process and the evidence. Australian Journal of Soil Research 33(2) 221 - 239

da Silva, A.P., Kay, B.D., Perfect, E., 1997. Management versus inherent soil properties effects on bulk density and relative compaction. Soil Till. Res. 44, 81–93.

da Silva, A.P., Kay, B.D., Perfect, E. 1994. Characterization of the least limiting water range of soils. Soil Sci. Soc. Am. J. 58, 1775–1781.

Diserens, E., & Battiato, A. 2013. TASC V3.0 - Prognose Bodengefährdung und Treibstoffverbrauch. ART-Berichte Nr. 766 (p. 8).

Diserens, E., Chanet, M., & Marionneau, A. 2010. Machine Weight and Soil Compaction: TASC V2.0.xls - a Practical Tool for Decision-Making in Farming (p. 10). AgEng 2010, Clermont-Ferrand, France, Sept. 7-9, 2010, REF 239.

Diserens, E., & Spiess, E. (2004). Wechselwirkung zwischen Fahrwerk und Ackerboden. TASC: Eine PC-Anwendung zum Beurteilen und Optimieren der Bodenbeanspruchung. FAT Berichte Nr. 613 (p. 16).

Dexter, A.R., Czyz, E.A., Richard, G., Reszkowska, A. 2008. A user-friendly water retention function that takes account of the textural and structural pores spaces in soil. Geoderma 143, 243–253.

Edwards, G. T., Dybro, N., Munkholm, L. J., & Sørensen, C. G. 2016. Assessing the actions of the farm managers to execute field operations at opportune times. Biosystems Engineering, 144, 38-51
Evans, R. 1996. Soil Erosion and its Impacts in England and Wales. London: Friends of the Earth Trust.
Ferrero, A.F. 1991. Effect of compaction simulating cattle trampling on soil physical characteristics in woodland. Soil and Tillage 19: 319–329.
Flowers, M.D. and Lal, R., 1998. Axle load and tillage effects on soil physical properties and soybean grain yield on a mollic ochraqualf in Northwest Ohio. Soil and Tillage Research, 48(1-2), 21-35.
Fröhlich, O.K. 1934. Druckverteilung im Baugrunde. Springer Verlag, Wien, 178 pp.
Fuller RJ and Gough S.J. 1999. Changes in sheep numbers in Britain: Implications for bird populations. Biological Conservation 91: 73–89.
Gasso, V., Sørensen, C. A. G., Oudshoorn, F. W., & Green, O. (2013). Controlled traffic farming: A review of the environmental impacts. European Journal of Agronomy, 48, 66–73. doi:10.1016/j.eja.2013.02.002
Håkansson, I., Lipiec, J. 2000. A review of the usefulness of relative bulk density values in studies of soil structure and compaction. Soil Till. Res. 53, 71–85.
Håkansson, I. 1990. A method for characterizing the state of compactness of the plough layer. Soil Till. Res. 16, 105–120.
Håkansson, I. 1994. Subsoil compaction caused by heavy vehicles - a long-term threat to soil productivity. Soil & Tillage Research 29, 105-110.
Håkansson, I., Voorhees, W.B., Elonen, P., Raghavan, G.S.V., Lowery, B., Vanwijk, A.L.M., Rasmussen, K., Riley, H. 1987. Effect of high axle-load traffic on subsoil compaction and crop yield in humid regions with annual freezing. Soil & Tillage Research 10, 259-268.
Hamza, M.A., Anderson, S.H., Aylmore, L.A.G., 2001. Studies of water drawdowns by single radish roots at decreasing soil water content using computer assisted tomography. Aust. J. Soil. Res. 39, 1387–1396.
Hamza, M.A., Anderson, W.K. 2005. Soil compaction in cropping systems - A review of the nature, causes and possible solutions. Soil & Tillage Research 82, 121-145.
Hassan, F. U., Ahmad, M., Ahmad, N., & Abbasi, M. K. (2007). Effects of subsoil compaction on yield and yield attributes of wheat in the sub-humid region of Pakistan. Soil and Tillage Research, 96(1-2), 361–366. doi:10.1016/j.still.2007.06.005
Heathwaite AL, Burt TP, and Trudgill ST. 1989. Runoff, sediment, and solute delivery in agricultural drainage basins – a scale dependent approach. International Association of Hydrological Sciences Publication 182: 175–191.
Heathwaite, A.L., Burt, T.P., and Trudgill, S.T. 1990. Land use controls on sediment delivery in lowland agricultural catchments. In: Boardman J, Foster IDL, and Dearing J (eds) Soil Erosion on Agricultural Land. Chichester: Wiley, 69–87.
Hills, R.C. 1971. The influence of land management and soil characteristics on infiltration and the occurrence of overland flow. Journal of Hydrology 13: 163–181.
Holman, I.P., Hollis, J.M., Bramley, M.E., and Thompson, T.R.E. 2003. The contribution of soil structural degradation to catchment flooding: A preliminary investigation of the 2000 floods in England and Wales. Hydrology and Earth System Sciences 7: 754–765.
Horn, R., 1981. Eine Methode zur Ermittlung der Druckbelastung von Bo¨den anhand von Drucksetzungsversuchen. Zeitschrift fu¨ r Kulturtechnik und Flurbereinigung 22 (1), 20–26.
Horn, R., Fleige, H. 2009. Risk assessment of subsoil compaction for arable soils in Northwest Germany at farm scale. Soil & Tillage Research 102, 201-208.
Horn, R., Rostek, J. 2000. Subsoil compaction processes - State of knowledge. Catena Verlag, Reiskirchen.
Ishaq, M., Ibrahim, M., Hassan, A., Saeed, M., Lal, R., 2001. Subsoil compaction effects on crops in Punjab, Pakistan: II. Root growth and nutrient uptake of wheat and sorghum. Soil Tillage Res. 60, 153–161.
Jarvis, N.J. 2007. A review of non-equilibrium water flow and solute transport in soil macropores: principles, controlling factors and consequences for water quality. Eur. J. Soil Sci. 58, 523–546.

Jones, A., P. Panagos, S. Barcelo, F. Bouraoui, C. Bosco, O. Dewitte, C. Gardi, et al. 2012. The State of Soil in Europe. Joint Research Centre Reference Report. Luxembourg: Publications Office of the European Union.

Keller, T., 2005. A model for the prediction of the contact area and the distribution of vertical stress below agricultural tyres from readily available tyre parameters. Biosystems Engineering, 92, 85–96

Keller, T., Arvidsson, J., Dexter, A. R., 2007. Soil structures produced by tillage as affected by soil water content and the physical quality of soil. Soil and Tillage Research 92 45–52

Kirby, J.M. 1991. Strength and deformation of agricultural soil: measurement and practical significance. Soil Use Manage. 7, 223–229.

Koue, P.M., Balstrøm, T., Breuning-Madsen, H. 2008. Update of the European Soil analytical database (SPADE-1) to version SPADE8. Report to the European Soil Bureau, EU-Joint Research Centre, Ispra, Italy

Krümmelbein, J., Wang, Z., Zhao, Y., Peth, S., Horn, R. 2006. Influence of various grazing intensities on soil stability, soil structure and water balance of grassland soils in Inner Mongolia, P.R. China. Adv. GeoEcol. 38, 93–101 (ISBN 3-923381-52-2).

Kulli, B., Gysi, M., Flühler, H. 2003. Visualizing soil compaction based on flow pattern analysis. Soil Till. Res. 70, 29–40.

Lamandé M., Schjønning P., 2008. The ability of agricultural tyres to distribute the wheel load at the soil-tyre interface. Journal of Terramechanics 45 (4), 109-120.

Lamandé, M., Schjønning, P., 2011. Transmission of vertical stress in a real soil profile. Part II: Effect of tyre size, inflation pressure and wheel load. Soil and Tillage Research 114 (2), pp. 71-77

Langmaack, M., Schrader, S., Rapp-Bernhardt, U., Kotzke, K. 2002. Soil structure rehabilitation of arable soil degraded by compaction. Geoderma 105, 141-152.

Lebert, M., Boken, H., Glante, F. 2007. Soil compaction - indicators for the assessment of harmful changes to the soil in the context of the German Federal Soil Protection Act. Journal of Environmental Management 82, 388-397.

Lebert, M., Brunotte, J., Sommer, C. 2004. Ableitung von Kriterien zur Charakterisierung einer schaedlichen Bodenveraenderung. entstanden durch nutzungsbedingte Verdichtung von Boeden/ Regelung zur Gefahrenabwehr, 46-04 ed. Umweltbundesamt.

Lees K.J., McKenzie A.J, Newell Price J.P, Critchley C.N., Rhymer C.M., Chambers B.J., Whittingham M.J., 2016. The effects of soil compaction mitigation on below-ground fauna: How earthworms respond to mechanical

Liu, C.W., Cheng, S.W., Yu, W.S., Chen, S.K., 2003. Water infiltration rate in cracked paddy soil. Geoderma 117, 169-181.

McGarry, D. 2003. Tillage and soil compaction. Conservation Agriculture: Environment, Farmers Experiences, Innovations, Socio-Economy, Policy, 307-316.

Nie, Z.N., Ward, G.N., Michael, A.T., 2001. Impact of pugging by dairy cows on pastures and indicators of pugging damage to pasture soil in south-western Victoria. Aust. J. Agricul. Res. 52 (1), 37–43.

Orr, H.G., and Carling, P.A. 2006. Hydro-climatic and land use changes in the River Lune catchment, NW England: Implications for catchment management. RiverResearch and Applications 22: 239–255.

Osetinsky, A., Shmulevich, I. 2004. Traction performance simulation of a pushed/pulled driven wheel. Trans. ASAE, 47(4), 981-994.

Owens, L.B., Edwards, W., and Van Keuren, R.W. 1997. Runoff and sediment losses resulting from winter feeding on pastures. Journal of Soil and Water Conservation 52: 194–197.

Pagliai, M. 1998. Soil porosity aspects. Int. Agrophys. 4, 215–232.

Pagliai, M., Marsili, A., Servadio, P., Vignozzi, N., Pellegrini, S. 2003. Changes in some physical properties of a clay soil in Central Italy following the passage of rubber tracked and wheeled tractors of medium power. Soil & Tillage Research 73, 119-129.

Pattison, I., Lane, S.N. 2011. The link between land-use management and fluvial flood risk: A chaotic conception. Progress in Physical Geography. 36(1), 72-92. Doi: 10.1177/030913331142538

Peng, X., Horn, R., Smucker, A., 2007. Pore shrinkage dependency of inorganic and organic soils on wetting and drying cycles. Soil Sci. Soc. Am. J. 71, 1095-1104.

Pichlmaier, B.R. 2012. Traktormanagement für Traktoren (PhD Thesis, p. 209). Dissertation der Technischen Universität München, Institut für Maschinen- und Fahrzeugtechnik. Lehrstuhl für Fahrzeugtechnik, München, Germany.

Raper R.L., 2005. Agricultural traffic impacts on soil Journal of Terramechanics, 42, 259–280

Reszkowska, A., Krümmelbein, J., Gan, L., Peth, S., Horn, R. 2011. Influence of grazing on soil water and gas fluxes of two inner Mongolian steppe ecosystems. Soil Till. Res. 111, 180–189.

Rounsevell, M.D.A., Jones, R.J.A, (1993). A soil and agroclimatic model for estimating machinery work-days: the basic model and climatic sensitivity. Soil and Tillage Research, 26 179-191

Sander, D.H., Allaway, W.H., Olson, R.A. 1987. Modification of nutritional quality by environment and production pratices, In: Olson, R.A., Frey, K.J. (Eds.), Nutritional quality of cereal grains. American Society of Agronomy, Inc., Madison, Wisconsin, USA, pp. 45-82.

Schjønning, P., Lamandé , M., Tøgersen, F.A., Arvidsson, J., Keller, T., 2008. Modelling effects of tyre inflation pressure on the stress distribution near the soil–tyre interface. Biosyst. Eng. 99, 119–133.

Schjønning, P., Lamandé, M., Keller, T., Pedersen, J., Stettler, M., 2012. Rules of thumb for minimizing subsoil compaction. Soil Use and Management,28(3), 378-393.

Schjønning, P., Lamandé, M., Tøgersen, F.A., Pedersen, J., Hansen, P.O.M., 2006. Minimering af jordpakning. Størrelse of fordeling af stress i trædefladen mellem hjul og jord (Reduction of soil compaction. Magnitude and distribution of stress in the contact area between wheel and soil). Report No. Markbrug 127, The Danish Institute of Agricultural Sciences, Tjele, Denmark. Available at: http://web.agrsci.dk/djfpublikation/djfpdf/djfma127.pdf ; (accessed 19/12/2011).

Schjønning, P., van den Akker, J.J.H., Keller, T., Greve, M.H., Lamandé, M., Simojoki, A., Stettler, M., Arvidsson, J., Breuning-Madsen, H., 2015. Driver-Pressure-State-Impact-Response (DPSIR) Analysis and Risk Assessment for Soil Compaction–A European Perspective. In: Sparks, D.L. (Ed.), Advances in Agronomy, pp. 183–237

Schwieger, H. 1996. Untersuchung neuartiger Laufwerke und lasergestütze Erfassung der Reifen-/ Bodenverformung (PhD Thesis, p. 166). Forschungsbericht Agrartechnik des Arbeitskreises Forschung und Lehre der Max-Eyth-Gesellschaft Agrartechnik im VDI (VDI-MEG), Kiel, Germany.

Soane, B. D., & van Ouwerkerk, C. (1994). Soil compaction problems in world agriculture. In: Soane, B. D., & van Ouwerkerk, C. (eds.). Soil compaction in crop production. Developments in Agricultural Engineering 11. Elsevier, Amsterdam, The Netherlands, 1 - 21

Soane, B. D., & van Ouwerkerk, C. (1995). Implications of soil compaction in crop production for the quality of the environment. Soil and Tillage Research, 35(1-2), 5–22. doi:10.1016/0167-1987(95)00475-8

Soane, B.D., 1990. The role of organic matter in soil compactibility: a review of some practical aspects. Soil Tillage Res. 16, 179– 201.

Soane, B.D., Dickson, J.W., Campbell, D.J., 1982. Compaction by agricultural vehicles: a review. III. Incidence and control of compaction in crop production. Soil Tillage Res. 2, 3–36.

Sogn, L., Hauge, N.H. 1976. Høstetidsforsøk med sorter av vårhvete, bygg og havre, Scientific reports of the research department - Norwegian grain corporation. The research department- Norwegian grain corporation, p. 39.

Söhne, W., 1953. Druckverteilung im Boden und Bodenverformung unter Schlepperreifen. Grundlagen der Landtechnik 5, 49–63

Söhne, W., 1958. Fundamentals of pressure distribution and soil compaction under tractor tires. Agric. Eng. 39, 276

Spoor, G., 1995. Application of mole drainage in the solution of subsoil management problem. In: Jayawardane, N.S., Stewart, B.A. (Eds.), Subsoil Management Techniques. Advances in Soil Science. pp. 67–109.

Spoor, G., Tijink, F.G.J., Weisskopf, P., 2003. Subsoil compaction: risk avoidance, identification and alleviation. Soil Tillage Research, 73, 175–182.

Steiner, M. 1979. Analyse, Synthese und Berechnungsmethoden der Triebkaft-Schlupf-Kurve von Luftreifen auf nachgiebigem Boden (PhD Thesis, p. 190). Forschungsbericht Agrartechnik des Arbeitskreises Forschung und Lehre der Max-Eyth-Gesellschaft (MEG) 33. München, Germany

Stettler, M., Keller, T., Schjønning, P., Lamande, M., Lassen, P., Pedersen, J., Weisskopf, P. 2010. Terranimo®-a web-based tool for assessment of the risk of soil compaction due to agricultural field traffic. In International Conference on Agricultural Engineering-AgEng 2010: towards environmental technologies, Clermont-Ferrand, France, 6-8 September 2010. Cemagref.

Stolte J., Mehreteab T., Øygarden L., Kværnø S., Keizer J., Verheijen F., Panagos P., Ballabio C., Hessel R (Editors); Soil threats in Europe; EUR 27607 EN; doi:10.2788/488054 (print); doi:10.2788/828742 (online), 2015

Sullivan, L.A. & Montgomery, L.L. 1998. Deep subsoil compaction of two cracking clays used for irrigated cotton production in Australia. Soil Use and Management, 14, 56–57

Taylor, J.H., 1983. Benefits of permanent traffic lanes in a controlled traffic crop production system. Soil & Tillage Research 3, 385–395.

Theng, J.M., Oades, J.M., 1982. Organic matter and water stable aggregates in soil. J. Soil. Sci. 33, 141–163.

Tullberg, J.N., 1990. Hunter, M.N., Paull, C.J., Smith, G.D. (Eds.), Proceedings of Queensland Department of Primary Industries Soil Compaction Workshop. Toowoomba, Australia, Why control field traffic 28, 13–25.

Unger, P.W., Kaspar, T.C. 1994. Soil compaction and root growth- a review. Agronomy Journal 86, 759-766.

van den Akker, J.J.H. 1997. Construction of a wheel–load bearing capacity map of the Netherlands. Proceedings of the 14th ISTRO Conference July 27 - Aug. 1, 1997, Pulawy, Poland. Polish Society of Agrotechnical Sciences. Bibliotheca Fragmenta Agronomica 2A/97, 15-18.

van den Akker, J.J.H. 2004. SOCOMO: a soil compaction model to calculate soil stresses and the subsoil carrying capacity. Soil Till. Res., 79, 113-127. http://dx.doi.org/10.1016/j.still.2004.03.021

Van den Akker, J.J.H. 2008. Soil compaction. In: Huber, S., Prokop, G., Arrouays, D., Banko, G., Bispo, A., Jones, R.J.A., Kibblewhite, M.G., Lexer,W., Möller, A., Rickson, R.J., Shishkov, T., Stephens, M., Toth, G. Van den Akker, J.J.H.,Varallyay, G., Verheijen, F.G.A., Jones, A.R. (Eds.) Environmental Assessment of Soil for Monitoring: Volume I Indicators & Criteria. EUR 23490 EN/1, Office for the Official Publications of the European Communities, Luxembourg, pp 107-124.

Vogeler, I., Horn, R., Wetzel, H., Kruemmelbein, J., 2006. Tillage effects on soil strength and solute transport. Soil Till. Res. 88, 193–204.

Voorhees, W.B. 2000. Long-term effect of subsoil compation on mais yield, In: Horn, R., Van den Akker, J.J.H., Arvidsson, J. (Eds.), Subsoil compaction: Distribution, Processes and Consequences. Catena Verlag, Reiskirchen, Germany, pp. 331-338.

Vzzotto, V.R., Marchezan, E., Segabinazzi, T. 2000. Effects of cattle trampling on lowland soil physical properties. Ciencia Rural. 30, 965-969.

Wiermann, C., Horn, R. 2000. Effect of different tillage systems on the recovery of soil structure following a single compaction event.

Williams S.M, Weil R.R., 2004. Cover crop root channels may alleviate soil compaction effects on soybean crop. Soil Sci. Soc. Am. J. 68:1403–1409.

Zhang, S., Grip, H., Lövdahl, L. 2006. Effect of soil compaction on hydraulic properties of two loess soils in China. Soil Till. Res. 90, 117-125.

Zhang, H., 1994. Organic matter incorporation affects mechanical properties of soil aggregates. Soil Tillage Res. 31, 263–275.

Zhang, H., Hartge, K.H., 1992. Effect of differently humified organic matter on the aggregate stability by reducing aggregate wettability. Z. Pflanzenernahr. Bodenkd. 155, 143–149.

Zhao, Y., Peth, S., Horn, R., Krümmelbein, J., Ketzer, B., Gao, Y., Doerner, J., Bernhofer, C., Peng, X. 2010. Modeling grazing effects on coupled water and heat fluxes in Inner Mongolia grassland. Soil Till. Res. 109, 75–86.

9 Soil-improving cropping systems for soil pollution

X. Yang, V. Silva, E. Huerta and V. Geissen

9.1 Background

Crop and animal production has strongly increased in Europe, especially during the second half of the 20th century, through the increased availability of high-yielding crop and animal breeds, fertilizers, pesticides, animal feed concentrates, mechanization and improved timing of the management activities. A side effect of this increased production is the dispersion of unwanted substances in the environment and sometimes also in food. Several studies on food safety reported mixtures of pesticide residues in food (Jardim and Caldas, 2012; Szpyrka et al., 2015) and even in mother milk (Ennaceur et al., 2007; Honeycutt et al., 2014; Liu et al., 2015). The site effects of intensive pesticide application on water quality is well studied, and international monitoring programs of water quality show that pesticide and antibiotics are present in surface and groundwater bodies with changing concentrations over the years (Larson et al., 1997; Hildebrandt et al., 2008; Folch et al., 2016; Kodešová et al., 2016; Wang et al., 2016). The soil pollution with agro pollutants like pesticides and antibiotics is a not well studied phenomenon in Europe and so far no (inter)national monitoring programs exist with respect to pollutants in agricultural soils (Drevno et al., 2016; Stolte et al., 2016).

Pesticides enter into the soil as a result of plant protection measures (weed control and pest control), while residues of antibiotics and its metabolites enter into soil via the application of animal manures and sewage water and sludge. Although the persistence of prevailing pesticides decreased in the last decades, a number of studies describe the occurrence of mixtures of persistent pesticides in soils as a result of long-term annual applications (e.g., organochlorines like DDT and its metabolites, forbidden in 1973 in Europe) (Oldal et al., 2006; Ferencz and Balog, 2010) or Glyphosate and its metabolite AMPA (Guo et al., 2014). Actually, more than 600 active substances for pest and weed control are on the European market, in more than 2000 pesticides.

The persistence, distribution and transport of pesticides in soil is governed by physico-chemical and biological processes. Sorption and degradation are the main processes influencing pesticide behaviour in soil and therefore its environmental impacts. Photo-, chemical- or bio-degradation are the processes responsible for the breakdown and reduction of pesticide levels in soil. The degradation of pesticides in soil can be partial (resulting in metabolite accumulation) or total (total mineralization of pesticides). Sorption of pesticides to soil particles (organic matter, clay, Fe-oxides) reduces pesticide mobility and the risk of water pollution. Nevertheless, by reducing pesticides bioavailability, the microbial degradation of sorbed pesticides is slower (Arias-Estévez et al., 2008; Shahgholi and Ahangar, 2014).

Pesticides residues in soils affect soil fauna and flora at different scales (as individuals, population and community) in the short and long term, thereby affecting the soil services that soil biota provides, including the decay of pesticides and organic matter (Pélosi, 2014).

Antibiotics are widely used to treat diseases and to protect human and animal health. In agriculture, antibiotics are added to fields through the application of waste-water, manures and biosolids. Due to their increasing use, the presence of antibiotics in the environment has been detected frequently, especially in the regions with intensive animal production, fish farming, horticulture and food preservation (Wang et al., 2015; Widyasari-Mehta et al., 2016), resulting in antibiotic contamination and elevated environmental risks for terrestrial and aquatic environments (Conkle and White, 2012; Folch et al., 2016; Li et al., 2015; Reichel et al., 2014; Thomaidi et al., 2016; Zhang et al., 2016).

Most studies on the environmental fate of antibiotics focus on aquatic environments (Kümmerer, 2009a; Kümmerer, 2009b; Zhang, et al., 2015) or wastewater treatment plants (Tasho and Cho, 2016) while the behaviour of antibiotics at environmentally relevant concentrations in agricultural soil is still limited. Nevertheless, residues of antibiotics entering with the application of organic fertilizers from industrial pig, chicken and cattle farms have been measured in many European soils (Tasho and Cho, 2016; Widyasari-Mehta et al., 2016). In soil matrix, the behaviour of antibiotics depends on its compound structure and also can be affected by soil physiochemical properties, such as soil texture, soil organic matter and pH (Wang et al., 2015) which affect antibiotics mobility in soil. Furthermore, antibiotics addition in soil can also affect soil microbial activities which, in turn, will influence antibiotics degradation processes (Ma et al., 2014; Yim et al., 2013). In addition, antibiotic metabolites may affect soil microbial processes, (Byzov et al., 1999, Kotzerke et al., 2010). Soil bacteria counts are considerably diminished with the presence of some antibiotics, and soil microbial activity is decreased. Also antibiotic resistant strains are formed (Thiele-Bruhn and Beck 2005).

Agricultural soils in Europe face the following problems with respect to these two groups of agro-pollutants:

- No threshold values exist for the majority of residues with respect to soil quality
- There is no knowledge of the actual state of pollution of most European agricultural soils, only fragmented information is available
- No remediation strategies exist for this form of diffuse pollution

For sustainable cropping the following is needed:

- Decrease the current state of soil pollution
- Avoid or reduce future input of pesticides and antibiotics into agricultural soils

9.2 Purpose

We conducted a literature review to study the potential effects of soil improving cropping systems on prevention/reduce future soil pollution.

9.3 Results and Discussion

The main effects of different soil-improving cropping systems are i) reduction/ban the input of new pollutants in the soil; ii) reducing the current (and often unknown) levels of soil contamination and iii) reducing the mobility of pollutants in soil and therefore the risk of water resources contamination, iv) enhancing above-ground diversity for promoting below ground diversity (soil diversity will stabilize the systems).

The most evident effect of the SICS is the increase of organic matter content in soil, enhancing the sorption of pollutants and the microbial populations and activities. By enhancing the sorption capacity, pollutants mobility is lower, decreasing the risk of surface and groundwater contamination. Nevertheless, high organic matter content stimulates tunnel forming by earthworm, facilitating pollutants leaching (Pélosi, 2014; Prado et al., 2016). On the other hand, enhanced microbial degradation reduce soil contamination levels and therefore the pool of contaminants available to be transported to the aquatic resources. Predict the exact efficacy of the practices involving the increase the organic matter content (such as organic amendments or conservational tillage) might not be easy since sorption and biodegradation are interdependent processes: sorbed pollutants are less mobile but also less bioavailable, resulting in slower degradation rates of pollutants and longer persistence in soils.

Considering the relative effects of the different and combined soil-improving cropping systems on the soil pollution, and their associated strengths, the adoption and implementation of SICS should be encouraged in sustainable agricultural strategies avoiding input of pesticides and antibiotics and developing of remediation strategies on a large scale. Below, we summarize the main soil-improving cropping systems for preventing and remediation of soil pollution by pesticides.

Crop rotation and cover crops

- Supressing weed seed germination and development by releasing allelochemicals, direct competition with weeds for resources or by disrupt the development of weed–crop associations → reduced use of herbicides to control weeds (Blackshaw et al., 2007; Campiglia et al., 2010; Teasdale et al., 2007)
- There is as yet no information available about the effects of crops and crop rotation on the degradation of antibiotics

Tillage

- Reduced or no-tillage: reduce soil erosion and the risk of surface water contamination by pesticides (Alletto et al., 2010) and antibiotics (Davis et al., 2006)

- Reduced or no-tillage → increase the use of herbicides to control weeds (During et al., 2002)
- Conservation tillage: decrease surface runoff (and the particle facilitated transport of pesticides) but increase sub-surface flow (Potter et al., 2015; Tebrugge and During, 1999)
- Conservation tillage: increase in organic matter content at the soil surface → increase in pesticides sorption in the topsoil layer (due to a higher sorption capacity) (Alletto et al., 2010; During et al., 2002)
- Sorption of pesticides in the topsoil layer→ lower availability for microbial degradation (Alletto et al., 2010).
- Sorption of antibiotics in soils: tillage activities strongly decreases Ofloxacin sorption (Zhou et al., 2014)
- Conservation tillage: supports the formation of macropores, and allows preferential flow → increased risk of leaching of pesticides (Okada et al., 2014)
- Reduced or no-tillage: decrease the possibility of antibiotics leaching (Stoddard et al., 1998) due to higher sorption, but increase through preferential flow in macropores.

Organic amendments (such as humic acid and green manure):

- Enhance biodegradation of pesticide residues due to enhancing microbial populations and activities (Fenoll et al., 2011b; Fenoll et al., 2014)
- Affect on the half-lives of antibiotics: no significant change (Lertpaitoonpan et al., 2015) or shorten the half-life time of antibiotics in soils (Žižek et al., 2015)
- Enhance adsorption of pesticides (higher adsorptive capacity of the insoluble organic matter added to the soil) → latent source pollution (Fenoll et al., 2011b; Fenoll et al., 2014; Okada et al., 2016; Si et al., 2006)
- Promote accumulation of antibiotics residues in soil: long-term treatment of manure soil contains higher antibiotics than open crop lands (Zhang et al., 2016)
- Reduction leaching of pesticide residues (by higher retention in topsoil layers and/or degradation) → lower risk of groundwater pollution (Fenoll et al., 2011b)
- Leaching risks: liquid manure in soil will increase the risk of leaching of antibiotics (Aust et al., 2010)

Mulches (plastic mulches, cover crops converted to mulches, crop wastes):

- Solarization and biosolarization → stimulate microbial degradation (due to the increased temperatures) (Fenoll et al., 2011a)
- The release of allelochemicals during the decomposition of the organic mulches → lower herbicide use to control weeds (Campiglia et al., 2010)
- The use of vegetative mulches/furrows reduces soil erosion and consequently pesticide concentrations in the surface runoff (Rice et al., 2007)
- Plastic residues fragmentation led to an accumulation of microplastics in soil → enhance adsorption of pesticides (Steinmetz et al., 2016)
- There is as yet no information about mulches and antibiotics.

Organic farming

- Pesticide use banned→ protection of soil (trace levels of persistent pesticide residues may still be detected), surface and groundwater quality (Puech et al., 2014)
- Control antibiotics input: nonessential antibiotics of medicine halted will reduce the concentration of antibiotics in the discharge/ faecel. Then the risks for surface water and soil contamination can be reduced (Oliver and Gregory, 2015).
- Forbidden waste water irrigation: the slurry of pig/chicken will be supervised. The risk of antibiotics from these sources can be reduced (Yonggang et al., 2015).
- Organic farming enhance a more long term-stable- system

Heavy mental pollution

The main sources for heavy metal entering environment include expanding industries, mining, disposal of high metal wastes (e-waste), leaded gasoline and paints, animal manures, sewage sludge, pesticides, wastewater irrigation, coal combustion residues (Lekfeldt et al., 2017; Li et al., 2014; Wuana and Okieimen, 2011). Heavy metals (Zn, Cu, Cr, Cd, Pb, Ni, Hg) accumulated in soil which can be adsorbed, accumulated and also be poisoned in plants, threatening human health via food chain (Roya and Akram, 2016). It is reported that 16% of the soil samples, 19% for the Chinese agricultural soils, are contaminated by heavy metals and metalloids (Zhao et al., 2015). With the risks of heavy mental in soil, a summary of the evidence from laboratory ecotoxicological studies of the effect of heavy metals on soil base respiration reveals an enormous disparity (Figure 9.1). Two hypothetical models of the effects of stress resulting from heavy metal toxicity on soil biodiversity (and consequently function) are presented in Figure 9.2. One model suggest that a modest concentration of heavy metals yields the highest level of biodiversity, while the other just shows a decrease with an increase of stress from heavy metal accumulation.

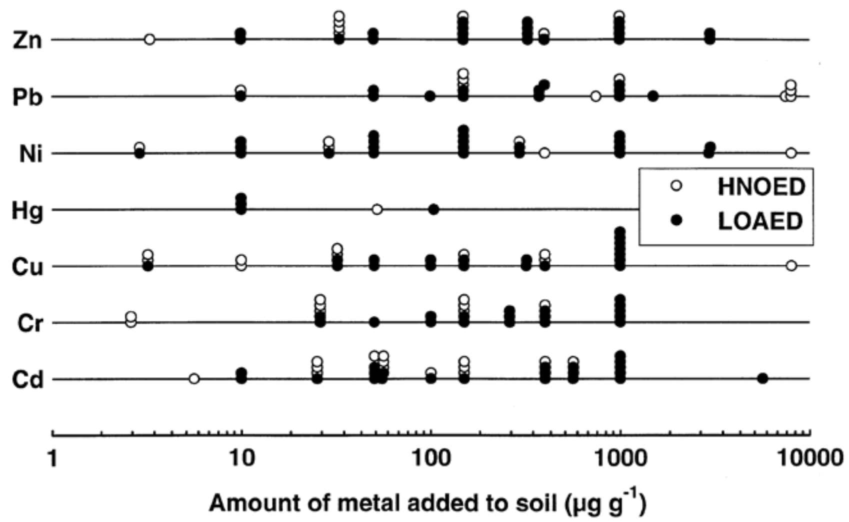


Figure 9.1. *Effect of addition of heavy metals on base respiration in different soils in laboratory ecotoxicological studies (Ken et al., 1998).*

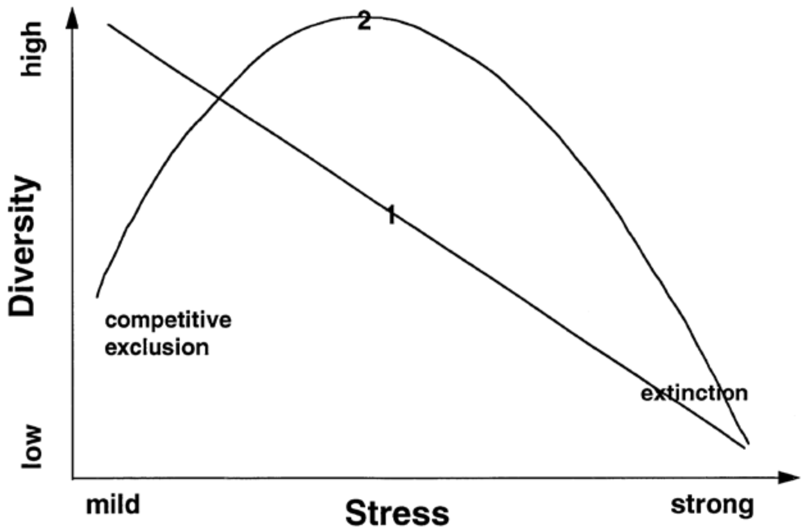


Figure 9.2. *Two hypothetical models of the effects of stress resulting from heavy metal toxicity on diversity (and consequently function) of a community of microorganisms, or a population of a microbial species (Ken et al., 1998).*

9.4 Conclusions

Pollution (or contamination) is the accumulation and occurrence of contaminants in soil, including heavy metals, pharmaceuticals, pesticides, disinfection by-products, and wood preservation and industrial chemicals. The origin of pollutants may be natural (genetic), industrial (deposition via air or dumping wastes) and/or agricultural (through contaminated inputs, including those by reusing waste-water). Soil contamination affects crop yield and quality, human health, biodiversity and biological activity, and may cause malnutrition and nutrient imbalances.

Pollution-specific SICS are directed towards (i) preventing pollution, (ii) minimizing the mobility and toxicity and/or stimulating the breakdown of pollutants, and (iii) lowering pollutant concentrations in soil through phytoremediation. In serious cases, contaminated soils may have to be treated chemically or physically (through heating). Pollution-specific SICS may involve the following three mechanisms, i.e., (i) changes in inputs, (ii) substitution, and (iii) redesign. The first mechanism relates to a drastic lowering of pollutant inputs (and to withdrawal of pollutants with harvested crops through phytoremediation, where possible). The second mechanism involves soil amendments which stimulate the biological breakdown of organic pollutants, and/or the lock-up of pollutants in soil in a less mobile and less toxic form. The third mechanism involves the growth of crops that are less sensitive to pollutants and/or the change of food and feed crops to bio-energy crops and set-aside land. Certain crops are called hyperaccumulators, i.e. these crops accumulate pollutants in the plant tissue, or degrade or render pollutants in less harmful contaminants.

Most promising pollution-specific SICS (i) prevent further pollution, and (ii) remediate polluted soils through phytoremediation (Table 9.1). They will improve resource use efficiency and the quality and safety of the crop products.

Table 9.1. *Qualitative assessment of pollution-specific SICS.*

	Components of cropping systems	Components of pollution-specific SICS	Change in profitability	Changes in soil properties		
				Physical	Chemical	Biological
A	Crop rotations	When possible/needed: +hyper-accumulating crops	-	+/-	+	+/-
B	•Nutrient management	Manuring, low in pollutants, Soil pH adjustment	+/-	+/-	++	+
C	•Irrigation management	optimal				
D	•Drainage management	optimal				
E	•Tillage management	optimal				
F	•Pest management	Low pesticide use	-/+	+/-	+/-	+/-
G	•Weed management	Low herbicide use	-/+	+/-	+/-	+/-
H	•Residue management	Optimal				
J	•Mechanization management	optimal				
K	•Landscape management	optimal				

9.5 References:

Lekfeldt JDS, Holm PE, Kjaergaard C, Magid J. Heavy metal leaching as affected by long-time organic waste fertilizer application. Journal of Environmental Quality 2017; 46: 871-878.

Li Z, Ma Z, van der Kuijp TJ, Yuan Z, Huang L. A review of soil heavy metal pollution from mines in China: Pollution and health risk assessment. Science of the Total Environment 2014; 468-469: 843-853.

Roya AQ, Akram A. Impact of toxic heavy metals in food systems: A systemic review. International Journal of ChemTech Research 2016; 9: 422-429.

Wuana RA, Okieimen FE. Heavy Metals in Contaminated Soils: A Review of Sources, Chemistry, Risks and Best Available Strategies for Remediation. ISRN Ecology 2011; 2011: 1-20.

Zhao FJ, Ma Y, Zhu YG, Tang Z, McGrath SP. Soil contamination in China: Current status and mitigation strategies. Environmental Science and Technology 2015; 49: 750-759.

10 Soil-improving cropping systems for soil organic matter decline

M.A. Bolinder, F. Crotty, A. Elsen, M. Frac, T. Kismanyoky, T. Kätterer, J. Lipiec, M. Tits and Z. Toth

10.1 Background

Knowledge about the importance of soil organic matter (SOM) for soil fertility and the impact of management techniques and cropping systems on SOM has a long history (Manlay et al., 2007). For instance, the beneficial effect of green manure on SOM and crop production goes back to the golden age (800 to 200 BC) of Greeks (Fageria, 2007), and soil preparation through some kind of tillage in agriculture originated several milleniums ago (Lal et al., 2007).

Over the past 300 years, concepts related to SOM evolved from a humic period (prior to 1840s), to a mineralist period (up until 1940s), moving towards the contemporary ecological period with an emphasis of more sustainable agro-ecosystems and global concerns such as GHGs emissions (Manley et al., 2007). SOM is now viewed as an assemblage and complex mixture of different compounds (organic molecules including soluble components), in which carbon (C) is the main constituent and that is derived from the microbial decomposition (at different degrees) of plant and animal residues, faunal and microbial biomass (e.g., Stockmann et al., 2013). The amount of SOM is determined by soil forming factors, i.e., parent material, climate, vegetation and land use, topography and hydrology, management and time.

Total soil organic C (SOC) is a measure of the SOM content without considering its origin or degree of decomposition. Approximately half of the SOM is C (Pribyl et al., 2010). Many soils contain also inorganic C (e.g., carbonates and bicarbonates), particularly in arid and semi-arid regions, which release CO_2 following dissolution of the carbonates, or sequester CO_2 through the formation of secondary carbonates (Izaurralde et al., 2001). Inorganic C in soils is not considered here further.

SOM is highly heterogenuous. Its quality spectrum varies from easily decomposable to very recalcitrant. Frequently, SOM is divided into a number of conceptual pools which are considered to be homogenuous. In the past, these pools of SOM have mostly been characterized using a number of chemical extractions. It is now more commonly defined with either functional approaches (kinetically based distinction) and/or by means of different fractionation procedures (physical size and density fractionation) such as particulate- and light-fraction organic matter (Feller and Beare, 1997; Stockmann et al., 2013; Diochon et al., 2016). For example, based on their mean residence time (MRT), there is commonly considered that there are three pools, one with a MRT less than a few years (labile), a more stable (MRT between 15 to 100-years) and a passive pool with a MRT ranging from centuries to millennia (Stockmann et al., 2013). The latter pool can include charcoal, which is preferably quantified separately.

A large proportion of the SOM is found in association with the silt- (20 to 40%) and clay-size (35 to 70%) fractions, a smaller amount (10 to 30%) is found in the sand-size fraction, and there is a trend towards higher MRTs as the fraction size decreases (Feller and Beare, 1997). However, recent work has shown that SOM in the clay-size fraction is more dynamic than previously believed, and have been found to cycle within very short time-scales (Diochon et al., 2016). The soil biodiversity also plays an important role in the decomposition process through soil microbiological activity, which is largely affected by soil meso- and macro-invertebrates (Dungait et al., 2012).

The total amount of SOM is considered the best indicator for soil quality because it influences a large number of physical, biological and chemical soil properties and conditions. With a characteristic C:N:P ratio of 100:10:1, decomposition of SOM is an important source of major plant nutriens. Although it is not always clear-cut (Oelofse et al., 2015), there is sometimes a positive relationship between SOM content and net primary producivity (NPP) in agro-ecosystems (Körschens et al., 2013; Lal, 2013). However, there is no agreement on a critical minimum SOM content (Merante et al., 2014) and desirable targets are often not quantified (Sparling et al., 2003).

For agro-ecosystems in temperate climatic regions, it was underlined by Loveland and Webb (2003) that a SOM level of 4% (i.e., about 2% soil organic C) was often deemed by soil scientists as a universal threshold level, below which there is a decline in soil quality. Recognizing that large uncertainties remain and lower levels (1% soil organic C) has been suggested (e.g., Lal, 2013). Since the SOM content of soil is related to particle size distribution and closely linked to soil physical properties, a ratio of clay or clay plus silt to SOM has been considered a more appropriate measure related to soil quality (e.g., Dexter et al., 2008; Morari et al., 2014), and potentially a better indicator relating crop yields to SOM dynamics (Olesen et al., 2014). The relation between clay and SOM was recently suggested as an indicator to guide the adoption of soil improving management techniques in Europe (Merante et al., 2017).

From a large-scale perspective, the pool of SOM present in the biosphere is important, and higher than the amount of C that is present in the atmosphere (800 Pg) and vegetation (600 Pg) combined (Lal, 2013; Stockmann et al., 2013). Indeed, there is about 2400 Pg of C down to a 2-m depth, 1500 to 1-m depth, with as much as around 700 Pg only in the arable layer (Kätterer et al., 2012; Paustian et al., 2016). Most agro-ecosystems have been subject to significant losses of SOM, with a depletion ranging from 25 to 75% of their initial stocks. Consequently, it should be possible to store more C in soils. Since SOM is important for productivity and also provides several other ecosystem services, it is considered one of the most cost-effective alternatives to counteract climatic change (Freibauer et al., 2004).

It is apparent that management practices favoring SOM accumulation in agricultual soils are predominately beneficial. However, “more” or “as much as possible” is not necessarily always best under all conditions and for all types of soils, and the influence of management practices

on SOM stocks are also scale-dependent. For example, Sojka and Upchurch (1999) raised the concern of a conceptual contradiction, where e.g., higher SOM contents may in fact increase the application rates of soil-incorporated pesticides and favor the transport of some elements (such as P and heavy-metals) through complexing with dissolved SOM.

Janzen (2015) suggested that it is more appropriate to focus on the energy carried by C, rather than attempting to maximise stocks, and thereby emphasize the importance of C flows in relation to ecosystem functionnality. Indeed, the decomposer system transforms plant debris, where richer (better quality) litter is degraded faster, increasing the speed of nutrient cycling through the soil trophic network (Ponge, 2013).

10.2 Purpose

We assessed the impacts of soil-improving cropping systems (SICS) on SOM contents and stocks of mineral soils using data from long-term (mainly) field experiments. Results from soil-inventories and modelling were not considered here; these latter sources were covered in other EU-funded projects such as RECARE and SmartSOIL (Morari et al., 2014; Merante et al., 2014).

The number of reviews (and meta-analysis) on this topic seems to be never-ending, enrichening the understanding and stimulating the debate on particular issues that have still to reach a concensus. Our starting point was partly based on the information available from a Systematic Map (Haddaway et al., 2015), where some of the existing reviews on crop rotations and management techniques had been retrieved.

10.3 Results and Discussion

Here we summarize the results of an extensive analysis made on 25 published reviews for the effect of some selected SICSs on SOM (results from research articles were also included in the discussion). It address the effect of recycled organic materials, N fertilization, aboveground crop residue handling, cover crops and the effect of no-tillage. A full overview of the results can be found in Bolinder et al. (Review in preparation for Soil Use and Management).

10.3.1 Concept

The SOM balance for agro-ecosystems is dynamic and determined by inputs and outputs. This concept has been developed via cumulative knowledge about SOM dynamics; the different sub-components and related driving factors in this system are continuously up-dated and refined accordingly with new experimental results. Soils are considered to be in steady-state (or equilibrium) when the inputs equals outputs. Carbon is being sequestered when the inputs are greater than the outputs and the net removal of CO_2 from the atmosphere through photosynthesis is transformed into SOM pools with long turnover times (long-lived SOM) (Kätterer et al., 2012).

The annual plant C inputs to soil are driven by NPP that determines the amount of photosynthetically-fixed C that can potentially be sequestered in SOM. NPP is defined as the total amount of C fixed by photosynthesis that is not respired by plants, and converted into

the above- and below-ground (roots and rhizodeposits) plant biomass through a given period of time. Total NPP varies among crop types and is mainly driven by climate, soil quality and management techniques (e.g., fertilisation, irrigation), and the share of total NPP that is returned to soil also depends on the handling of aboveground NPP (e.g., small-grain cereal straw left or removed from the field). In terms of the actual amount of C from crop residues and rhizodeposits to soil, both forage crops in the last year of a rotation and grain-maize are known to contribute the most, while root crops such as potato and sugar beet contributes much less (Bolinder et al., 2007; 2015); the other major crop types used in agro-ecosystems have an intermediate contribution. Additional inputs derive from recycled organic materials such as manures, composts, sludge and biochars (e.g., as a C-enriched coproduct from pyrolysis of biomass).

The proportion of annual C inputs to soil that enters the stable SOM-pool is often termed "humification coefficient" and it mostly follows the meaning given by Henin and Dupuis (1945), i.e., the fraction of organic material converted to more resistant SOM. It varies with the quality of the inputs, with typical values for above-ground plant tissues of 0-15%, higher for root-derived materials (15-35%), and are generally greater for e.g., manures, composts and sewage sludge (Kätterer et al., 2011, and references cited therein). However, it is also debated in the literature to which extent input of fresh organic compounds (e.g., from certain type of manures) would also stimulate decomposition (i.e., eventually even lead to a negative effect on the SOM balance) of SOM present in soil: the priming hypothesis. It is recognized that this effect still needs further evaluation and that it is possibly better perceived as an short-term effect (e.g., see discussion in Fontaine et al., 2003; Stockmann et al., 2013). The magnitude of the annual C inputs to soil, in particular from post-harvest crop residues and rhizodeposition, is the most crucial factor with respect to both potential soil organic C sequestration rates and SOM steady-state levels.

In terms of output, the decay of SOM as a result of soil biological activity is the major component, and the main route by which C returns as CO_2 back to the atmosphere. Generally, the decompositon rate is in the order of magnitude between 0.5 to 2.0%, annually, with the lower rates in fine-textured soils. It further varies with soil temperature and moisture and is therefore driven by climatic variables. Disregarding extreme conditions, SOM decomposition rates tend to increase with higher soil temperature and moisture levels (Bolinder et al., 2013). Moreover, it can be influenced by management techniques. Soil disturbance can enhance decomposition of SOM compounds because of reduced physical protection (Six et al., 2004), and bare-fallow used in semi-arid regions to store soil moisture may also stimulate decomposition (Boehm et al., 2004).

Another pathway for C losses from soil is the leaching of dissolved organic carbon (DOC) to groundwater and surface waters. Although a large proportion of leached DOC can be mineralized and lost as CO_2, along its way through the landscape some may also be precipitated and sequestered (Izaurralde et al., 2001). Usually, DOC represent a fairly small

share of the total SOM and has mostly been studied in temperate forest ecosystems. Concentrations are often greater in forest soils, followed by grassland and arable soils (Chantigny, 2003), and more important in organic (compared to mineral) soils (Smith et al., 2010).

A third output component is related to the removal of soil C attached to harvested products. Soil C losses via crop harvesting can be viewed as a form of erosion but is often overlooked as such in conventional erosion and sediment budget studies, albeit in the same order of magnitude as water and tillage erosion (Ruysschaert et al., 2006). It is mostly important for root crops, in particular sugar beet, potato and carrots.

Fourth, soil erosion (water, wind and tillage) can also affect the SOM content and budgets. However, some of the surface soil particles are only subject to a redistribution across the landscape, others are entering depressions or transported into aquatic ecosystems where a certain fraction can be lost as CO_2 through mineralization or methanogenesis (Lal, 2004). Therefore, this later component is more scale-dependent, i.e., it can be important in a farmers field but not necessarily from a global perspective, compared to the three former pathways of SOM outputs. In fact, although still under examination, soil erosion may eventually even be a sink for atmospheric CO_2 in a large-scale perspective (Sommer et al., 2016).

Since the decomposition of SOM is slow and there is a high background-level of stocks already present in soils, the changes in SOM are usually difficult to quantify in the short-term, and treatment effects have to accumulate over decades before they become measurable. As a consequence, several alternatives to study the effect of SICSs have been developed. The most useful and indispensable source of information are long-term (≥ 10-yrs) field experiments (LTEs). Debreczeni and Körschens (2003) was recently estimating there are more than 600 in the world, many of them located in Europe. In the past fifteen years, there are a number of reviews and meta-analysis published using results from these LTEs and other medium-term (≤ 10-yrs to about 5-yrs) experiments.

In the following, we briefly present an overview of results from 25 reviews and meta-analysis. Unless stated otherwise, the assessments made in these publications for each management technique were for various common crop types and rotations, and all considered the arable soil layer (0-20 and/or 0-30 cm depth). Some reviews covered more than one management technique.

10.3.2 The effect of recycled organic material (ROM).

The application of ROM is one of the most efficient components of SICS to mitigate or prevent SOM losses (Diacono and Montemurro, 2010). Increases obviously vary depending upon the quantity applied, but also with the quality of the materials that is driving the proportions of organic material converted to more resistant SOM. It is therefore predominately an input driven effect, although losses may attenuate the impacts (e.g., DOC in liquid manures).

In four reviews where the focus was on solid ROM and most clearly defined by the authors as manure (or animal- and farmyard manure), the relative effect ranged from 26.0 to 43.4%. Three studies that also allowed a reporting of data as soil organic C (SOC) sequestration rates showed it ranged from 250 to 311 kg C ha^{-1} yr^{-1} (Figure 10.1). The earliest review by Smith et al. (1997) considers data that allowed seventeen paired comparisons (*n*=17) from fourteen long-term field experiments (LTEs) in Europe with a mean duration (MD) of 72-yrs. Körschens et al. (2013) also reviewed results from several European LTEs (MD=40-yrs). The most extensive studies were the meta-analysis by Ladha et al. (2011), and Maillard and Angers (2014) with a much larger number of paired observations from several regions of the world, covering different climatic zones. The MD of the observations used to derive the relative effect by Maillard and Angers (2014) were at least 20-yrs, and those used for SOC sequestration rates was 18-yrs. The duration of experiments reviewed by Ladha et al. (2011) ranged from 6 to 158-yrs.

In a research paper on another European long-term field experiment (MD=30-yrs) in Poland, Rutkowska and Pikula (2013) reported a mean relative increase in SOM due to solid manure applications of 12% (range of values from 3 to 21%). In two other reviews, one with data (*n*=17) from LTEs specific for Canada (VandenBygaart et al., 2003) and data (*n*=37) specific for a Mediterranean climate (Aigulera et al., 2013), the relative increase were 28.2 and 23.5%, respectively. However, the solid ROM in these studies were not so well defined and may have included e.g., wood chips and sewage sludge. In particular, the SOC sequestration rate of as much as 1310 kg C ha^{-1} yr^{-1} reported by Aigulera et al. (2013) may be due to the fact that the studies they reviewed for solid ROMs also included composted materials.

The highest effects occured for municipal solid ROMs and sewage sludge, with relative increases and SOC sequestration rates ranging from 98 to 117% and 1650 to 5290 kg C ha^{-1} yr^{-1}, respectively (Smith et al., 1997; Aigulera et al., 2013). Aigulera et al. (2013) also assessed the effect of liquid animal manure in their study, but it was not significant. However, as pointed out in the meta-analysis by Maillard and Angers (2014), there is a lack of studies allowing realistic comparisons between the effects of liquid versus solid manures on SOC stocks.

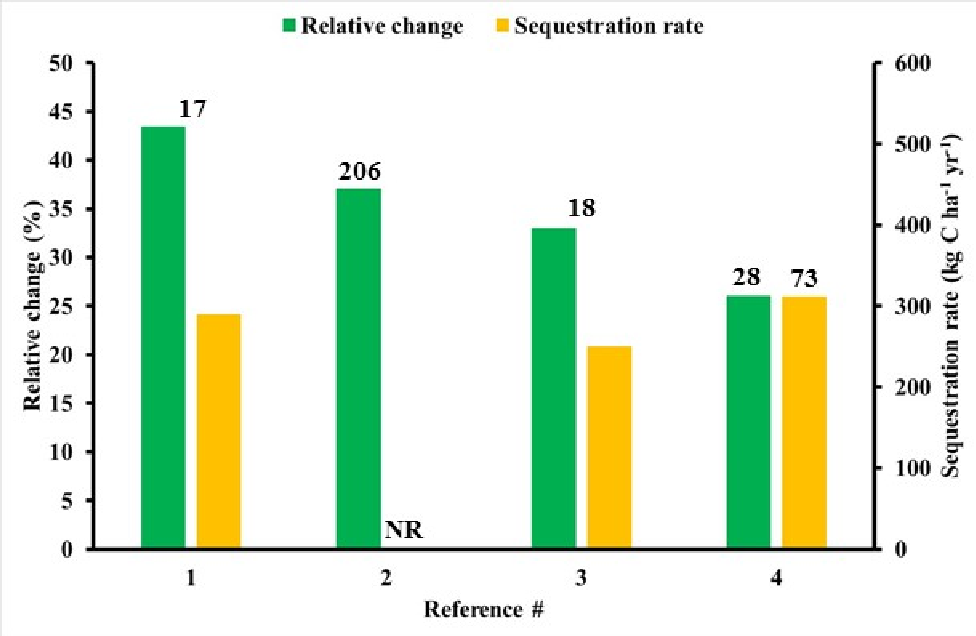


Figure 10.1. *The effect of solid recycled organic material (manure) on the relative change and soil organic C (SOC) sequestration rates, where numbers indicate paired observations (n) made against a control treatment (from Bolinder et al. in preparation). Reference #1 Smith et al. (1997), #2 Ladha et al. (2011), #3 Körschens et al. (2013) and #4 Maillard and Angers (2014). NR = Not reported.*

The data for the effect of manure, in terms of SOC sequestration rates, are facing some difficulties because the amounts applied in LTEs are not necessarily reflecting today's agronomic practices. A lowering of application rates is occurring since they are subject to regional agro-ecosystem N (and P) balance-based regulations. Sewage sludge applications are also subject to other type of regulations. For instance, in Sweden, regulations limit SOC sequestration rates to around 80 kg C ha^{-1} yr^{-1} (Kirchmann et al., 2017), which is much lower than the above-mentioned literature data.

With respect to the influence of pedo-climatic conditions, Ladha et al. (2011) found no noteworthy difference between climatic zones. However, Maillard and Angers (2014) mentioned a trend towards lower SOC sequestration rates for the tropical zone, as compared to the warm- and cool-temperate zones they considered in their analysis. Maillard and Angers (2014) found no effect of soil texture, but Körschens et al. (2013) noted a trend for higher SOC sequestration rates with increasing clay content. The latter study also found that the yield increase associated with manure applications was 6%.

10.3.3 The effect of N fertilization (NF).

It is well established that there is a positive effect of NF on SOM in agro-ecosystems, which is mainly an input driven consequence, where the increase in NPP results in higher amounts of annual C inputs to soil from aboveground post-harvest crop residues and rhizodeposition (e.g.,

Christopher and Lal, 2007). However, this effect is usually expected to be limited if the aboveground crop residues are removed.

In five reviews on the effect of N fertilization, using paired comparisons between fertilized versus unfertilized treatments, the relative effect ranged from 3.5 to 10.0%. Two of the studies allowed a reporting of data as SOC sequestration rates, and showed a lowest and highest value of 73 and 230 kg C ha^{-1} yr^{-1}, respectively (Figure 10.2). The data from Körschens et al. (2013) are from the same experiments as in their review on the effect of solid ROM, also allowing a calculation on the effect of mineral NPK (reference=no NPK, i.e., N effect was confounded with PK effect). Similarly, some of the data in the analysis on the effect of ROM made by Aigulera et al. (2013) for a Mediterranean climate also allowed a comparison between fertilized versus unfertilized treatments (MD=6-yrs), as well as those considered in VandenBygaart et al. (2003) for Canada (MD=23-yrs). The meta-analysis of Ladha et al. (2011), mentioned in the previous section for ROM, actually focused on the NF effect (at various rates) against a reference receiving no N (but most often receiving PK) and is by far the largest data set. Lu et al. (2011) also constitutes an extensive review with as much as 340 paired comparisons from experiments mainly located in Europe and North America (duration between 1 to 45-yrs).

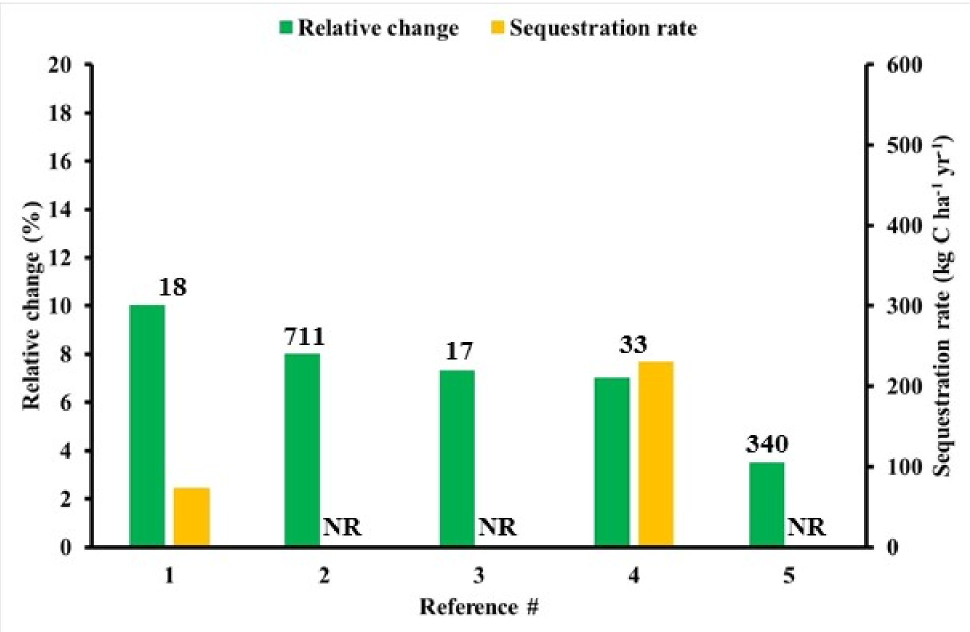


Figure 10.2. *The effect of N fertilization on the relative change and soil organic C (SOC) sequestration rates, where numbers indicate paired observations (n) made against a control treatment (from Bolinder et al. in preparation). Reference #1 Körschens et al. (2013), #2 Ladha et al. (2011), #3 Aigulera et al. (2013), #4 VandenBygaart et al. (2003) and #5 Lu et al. (2011). NR = Not reported.*

In another review on long-term (about > 5 to 20-yrs) field experiments mostly located in North America (*n*=111), it was found that SOC storage increased by 2 Mg C ha^{-1} for each cumulative 1 Mg of N ha^{-1} applied (Alvarez, 2005). However, this assessment showed the effect was only present when aboveground crop residues were returned. Similarly, under Nordic conditions, in a review on 10 long-term (about 50-yrs) field experiments with above-ground crop residues left in the field, Kätterer et al. (2012) found SOC stocks were increasing with 1 to 2 kg C ha^{-1} yr^{-1} for each kg of N applied. Similarly, Körschens et al. (2014) found that for commonly recommended application rates of combined mineral-organic fertilization, results in 15 central and east-European pedo-climatic conditions generally showed increasing SOC trends.

The effect of NF assessed within the study by Ladha et al. (2011), showed the mean increase in SOC was 8% for all climatic zones (Figure 10.2), but the values were higher (11 to 16%) in humid subtropical and tropical areas as compared to the values (3%) for temperate areas (data not shown). Alvarez (2005) also found that the effect of NF varied with climate, increasing with higher rainfall but decreasing with higher mean temperatures. The latter study also indicate a lower effect in fine textured soils.

10.3.4 The effect of aboveground crop residues (AGCR) handling.

The effect of aboveground crop residue handling on SOM is also an input driven consequence, although it can significantly reduce losses occurring from soil erosion. The subject has been highly debated in recent decades because aboveground biomass also is a source for producing biofuels. It is mostly an issue for residues from maize (grain) or small-grain cereal crops. The use of aboveground biomass from forage crops may have less effect on SOM due to its perennial growing cycle and more significant contribution of rhizodeposits. Depending on growing conditions and harvesting techniques, only about 50% of the straw may actually leave the field, a large proportion is left behind as stubble, chaff and uncollected straw (e.g., Powlson et al., 2011). The impacts on the annual C inputs to soil are generally expected to be greater for grain-maize because the potential aboveground crop residues represent approximately twice the amount that of small-grain cereals (Wilhelm et al., 2004).

We summarized data from eight reviews and meta-analyses on the effects of AGCR handling (Figure 10.3). The earliest with European studies (MD=21-yrs) by Smith et al. (1997) and more recently within the CATCH-C project (majority of studies were between > 5 to > 20-yrs) by Lehtinen et al. (2014). Powlson et al. (2011) analyzed a combination of European and North American LTEs, where the soil depth considered was usually the arable layer (but in many cases it was only for the 0 to 10 cm depth) and the duration varied from 6- to 56-yrs. The assessment made by Liu et al. (2014) was global with sites around the world of varying lengths, from short-term (1- to 3-yrs) to long-term (> 15-yrs). The other studies were country specific. Where Luo et al. (2010) considered only Australian data, with a duration between > 1- to 25-yrs, while VandenBygaart et al. (2003) data were Canadian specific (MD=14-yrs). Two studies considered the major agricultural zones in China, that by Lu (2015) using LTEs (of varying lengths), and that by Wang et al. (2015), with a MD=18-yrs.

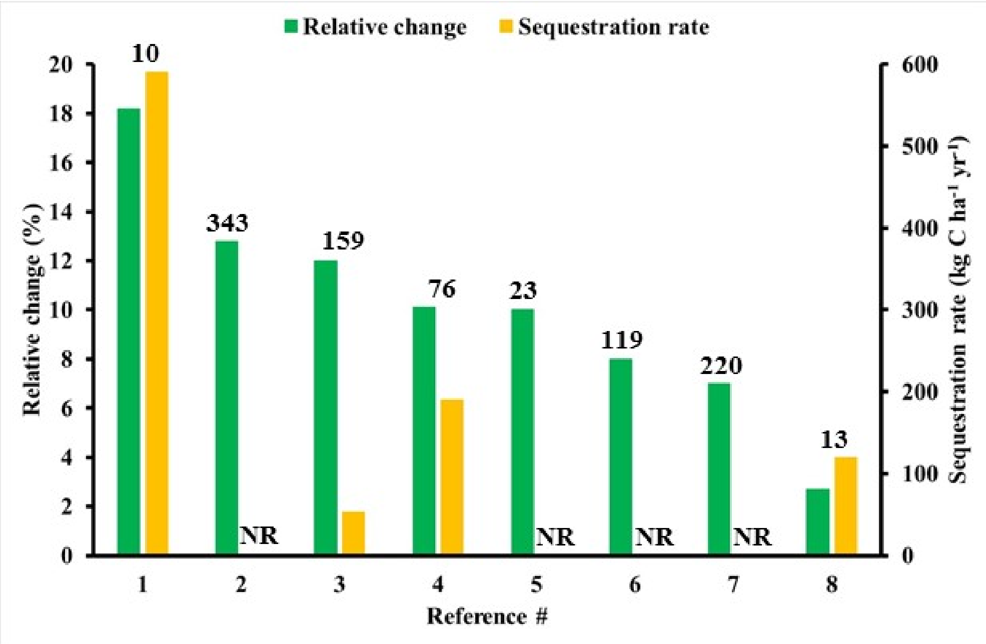


Figure 10.3. *The effect of aboveground crop residue handling on the relative change and soil organic C (SOC) sequestration rates, where numbers indicate paired observations (n) made against a control treatment (from Bolinder et al. in preparation). Reference #1 Smith et al. (1997), #2 Liu et al. (2014), #3 Lu (2015), #4 Wang et al. (2015), #5 Powlson et al. (2011), #6 Luo et al. (2010), #7 Lehtinen et al. (2014) and #8 VandenBygaart et al. (2003). NR = Not reported.*

Compared to the control treatment with straw removal, the mean relative increase in SOC with aboveground residue incorporation ranged from a low of 2.7 to a high of 18.2%. Four of the studies that allowed a reporting of data as SOC sequestration rates, showed a range of values from 53 and 590 kg C ha^{-1} yr^{-1} (Figure 10.3). The effect of different crop types and rotations was generally not significant, except in one study that observed a trend for a lower relative effect (9 to 10%) when rice was present, compared with maize- and wheat-based (13 to 14%) rotations (Lu, 2015). Lu (2015) also found a greater relative effect (13%) for chopped AGCR, compared to unchopped (9%).

Another summary assessment for the effect of AGCR handling, but specific for grain-maize by Anderson-Texeira et al. (2009) and only considering North American studies (*n*=15, MD=5-yrs), shows that the mean relative increase in SOC stocks varies with the proportion of aboveground biomass removed. Indeed, when all (100%) or only a lower proportion (25%) of residues are removed for grain-maize, then the mean relative increase in SOC stocks varied from as much as 800 to 300 kg C ha^{-1} yr^{-1}, respectively.

The effect of climate was only significant in the study specific for Australian sites (Luo et al., 2010), where it varied with mean annual air temperature and rainfall. The influence of soil texture was detectable in some of the studies. Luo et al. (2010) found variable effects; while Liu et al (2014) and Anderson-Texeira et al. (2009) found that the relative effect of straw removal on SOC for clayey soils was lower than for sandy soils, which is contrary to the findings by Lehtinen et al. (2014) where the effect was greater for soils with a clay content exceeding 35%. When data were available, it was also shown that the consequence of straw retention on crop yields were positive, increasing by 6% (Lehtinen et al. (2014), 7% Wang et al. (2015) and by 12.3% Liu et al. (2014).

10.3.5 The effect of cover (and catch-) crops (CC).

The use of CC have a positive effect on SOM, and it is frequently a management option promoted to counteract the negative impact of AGCR removal. This effect is mainly input-driven since the CC provides an additional source of annual C inputs to soil. The yield of the main-crop can also increase, further increasing the annual C inputs to soil through an overall higher NPP. For example, in a meta-analysis for spring cereal rotations, Valkama et al. (2015) found that legumes and mixed cover- and catch crops increased the main-crop yields by 6%, however, non-legumes decreased yields by 3%. Moreover, the overall positive effect of CC on SOM is related to decreased soil erosion, for periods when there is often no ground cover in annual cropping systems. The associated reduction in soil erosion can also be very high when CC are used in permanent woody cropping systems (Aigulera et al., 2013).

Compared with the previous management techniques (e.g., the effect of ROM, NF, AGCR), the mean relative increase in SOM with the presence of CC (as compared to the reference treatment with no CC) in five reviews and meta-analysis's we retrieved was more constant. The relative effect ranged from 8.5 to 13% and SOC sequestration rates showed a variation of values from 279 to 410 kg C ha^{-1} yr^{-1} (Figure 10.4). The Poeplau et al. (2015) study was specific for Sweden, analyzing the effect of a ryegrass CC (undersown in cereal-based crop rotations) in three LTEs (MD=20-yrs). Blanco-Canqui (2013) summarized data (mostly U.S. studies) from 10 short- and long-term experiments (MD=8-yrs), where crop rotations considered was mainly cereal-based but the different CC treatments varied considerably. Aigulera et al. (2013) studied CC treatments specific for situations where they substituted bare soils, either in studies with herbaceous or woody cropping systems under a Mediterranean climate (MD=10-yrs). McDaniel et al. (2014) and Poeplau and Don (2015) represents the most extensive meta-analysis allowing comparisons of CC vs. no CC. McDaniel et al. (2014) considers mostly observations from North America, but includes also European, South American Australian and African data (MD=18-yrs). The observed mean relative increase in SOC for that study of 8.5% was obtained with CCs that were mainly (i.e., 97% of the observations) leguminous. The data analyzed (MD=54-yrs) by Poeplau and Don (2015) had about 25 and 75% of the observations representing a tropical and temperate climate, respectively.

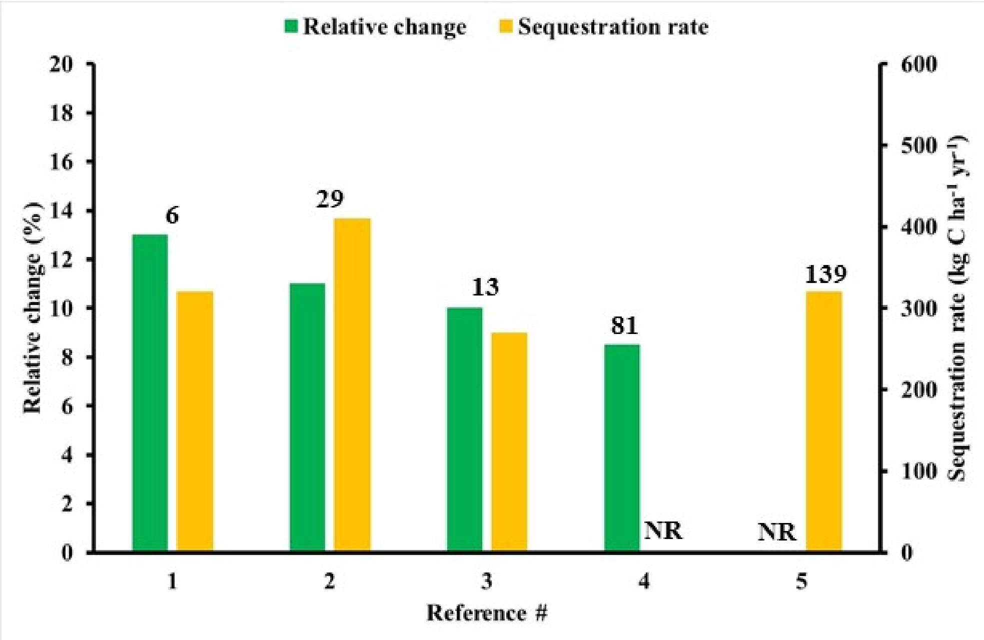


Figure 10.4. *The effect of cover (and catch-) crops on the relative change and soil organic C (SOC) sequestration rates, where numbers indicate paired observations (n) made against a control treatment (from Bolinder et al. in preparation). Reference #1 Poeplau et al. (2015), #2 Blanco-Canqui (2013), #3 Aigulera et al. (2013), #4 McDaniel et al. (2014) and #5 Poeplau and Don (2015). NR = Not reported.*

The effect of using of CC in perennial tree plantations are often higher than that observed for agro-ecosystems. For example, Gonzalez-Sanchez et al. (2012) studied the effect of CC implemented in between perennial tree (Olives) rows in Spain (*n*=13). Most of the comparisons came from short-term experiments (4-yrs) but three of the observations represented a long-term (28-yrs) effect, resulting in a mean increase in SOC storage of 1590 kg C ha^{-1} yr^{-1}. The increase was higher with native species CC (1780 kg C ha^{-1} yr^{-1}) compared to using sowed barley and ryegrass species (1160 kg C ha^{-1} yr^{-1}). Furthermore, in a research paper by Palese et al. (2014) for an olive orchard in a Mediterranean climate (Italy), a CC (and no tillage) consisting of spontaneous grasses and pruning's left was compared with the traditional management using mechanical tillage (2 to 3 times per year) and pruning's removed. The relative increase in SOC with the CC was 27% in the upper 10 cm of the soil after 7 years. However, the effect was only significant in the flat position, and less pronounced in the deeper soil layers (10 to 30 cm). In another research paper for a vineyard located in the hot and warm Mediterranean area of Italy, it was found that the short-term (3-yrs) effect of a CC (subterranean clover as mulching (and no tillage) compared with a conventional tilled and un-mulched treatment) resulted in a relative increase of as much as 55% (Favretto et al., 1992).

Poeplau et al. (2015), and Poeplau and Don (2015) was the only studies addressing the influence of climate and soil texture on the effect of CC, and the effect of CC on the main-crop yields. There was no significant relationship between the SOC storage rates with soil texture or climatic variables, and the presence of CC was generally not influencing the yield of the main-crop at the Swedish sites (Poeplau et al. (2015). Likewise, there was no interaction with climate (i.e., temperate vs. tropical) or with plant functional types (i.e., legumes vs. non-legumes) with the much larger data set (Poeplau and Don, 2015).

10.3.6 The effect of tillage

Studies for the effect of tillage techniques usually considers three main categories. No-tillage (NT; also often referred to as zero-tillage or direct drilling, where the soil is left undisturbed from harvest to seeding), conventional-tillage (CT; a practice that substantially mixes the soil and often implies full soil inversion through ploughing in cool and humid temperate regions) and reduced-tillage (RT; with an intermediate degree of tillage operations excluding ploughing).

The main factors that favor accumulation or minimize losses of SOC for NT systems in the arable soil layer are reduced soil erosion and improved soil structure (enhanced protection of SOC compounds in aggregates), while reduced yields (leading to less annual crop residue C inputs to soil) can be an unfavorable factor. The common belief is that systems with RT, and in particular CT practices, are subject to higher losses of SOC through increased decomposition of SOC (disruption of soil aggregate structure).

The outcomes for RT treatments usually are in-between, or sometimes quite similar to one of the other two categories. For example, in three reviews considering worldwide data, West and Post (2002) found no significant difference in SOC contents when they compared CT versus RT treatments, Alvarez (2005) indicated that although there was a trend for higher SOC levels under NT it was not significantly different from those in RT treatments. Ogle et al. (2005) showed that compared to CT, the positive effect of RT on SOC was approximately half that of NT. Furthermore, these three categories of tillage systems are not necessarily continuous through time, they can also be periodic (discontinuous), which may results in altered consequences (e.g., Conant et al., 2007).

We summarized results from ten reviews considering NT versus CT comparisons. The mean duration of the LTEs in those assessments are similar, being between 11 to 13-yrs in most reviews (Ogle et al., 2005; Aigulera et al., 2013; VandenBygaart et al., 2003; Luo et al., 2010; Gonzalez-Sanchez et al., 2012; Alvarez, 2005), however, the Ogle et. (2005) estimate is representing a SOC change occurring after 20-yrs. A slightly longer mean duration (i.e., 15-yrs) occurring for the studies by Virto et al. (2012) and West and Post (2002). While the longest mean duration of LTEs (16-yrs) was that in the study by Angers and Eriksen-Hamel (2008), Manley et al. (2005) did not report mean duration. The mean relative increase in SOC and the

SOC sequestration rate for NT (as compared to the CT reference treatment) in seven references ranged from 3.0 to 17% and from 50 to 580 kg C ha^{-1} yr^{-1}, respectively (Figure 10.5).

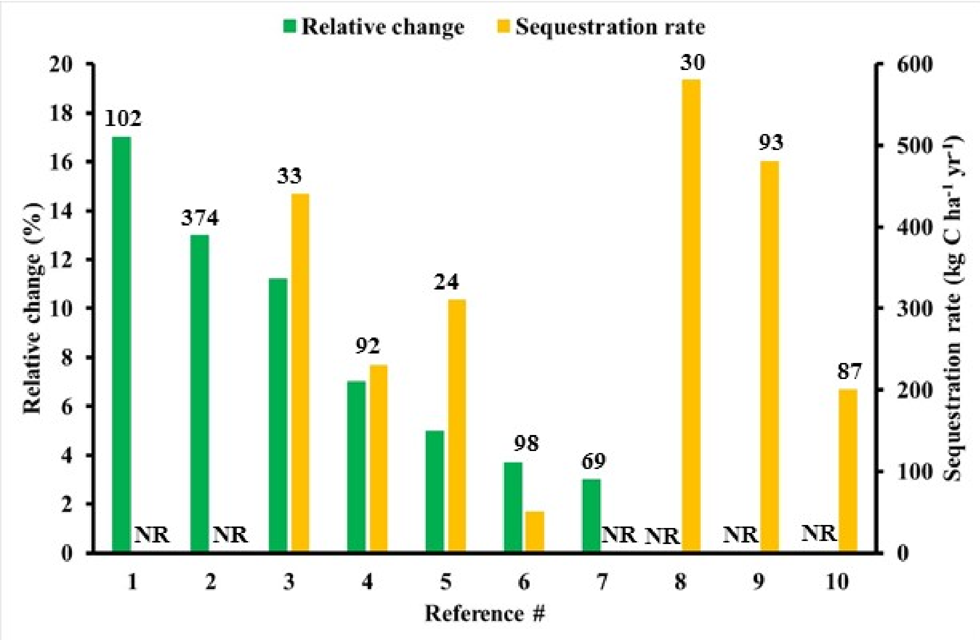


Figure 10.5. *The effect of no-tillage on the relative change and soil organic C (SOC) sequestration rates, where numbers indicate paired observations (n) made against a control treatment (from Bolinder et al. in preparation). Reference #1 Ogle et al. (2005), #2 Manley et al. (2005), #3 Aigulera et al. (2013), #4 Virto et al. (2012), #5 Angers and Eriksen-Hamel (2008), #6 VandenBygaart et al. (2003), #7 Luo et al. (2010), #8 Gonzalez-Sanchez et al. (2012), #9 West and Post (2002) and #10 Alvarez (2005). NR = Not reported.*

However, the effects are not always overall in favor of NT and large variation exists, e.g., in Virto et al. (2012) about 25% of the observations had a response ratio NT/CT less than unit. The outcome converts to even more variability when a large part of the observations includes more subsoil depths. For instance, in another review where almost half of the studies had measured SOC for depths greater than 30 cm, Govaerts et al. (2009) reported that the effect of NT was significantly lower for 7 paired comparisons, higher in 40, and not significantly dissimilar in 31 comparisons. It is often shown that most of the changes occurs in the 0-10 cm layer, with as much as 85% of the differences being accounted for in the top 7 cm (West and Post, 2002), and that they thereafter asymptotically tends to reach a no-difference around 30 cm (Manley et al., 2005). Angers and Eriksen-Hamel (2008) and Luo et al. (2010), having 70 to 100% of their pairwise comparisons involving depths greater than 30 cm, found the effect of NT was not significant in intermediate soil layers (10-20 cm); the gain in SOC for NT only occurring in the 0-10 cm and the reverse effect found below the plough layer (20-35 cm). As highlighted by

Franzluebbers (2011), there is often a higher stratification ratio of SOC (0-5/12.5-20 or 15-30 cm) in soils under long-term conservation tillage, compared to conventionally tilled systems.

The mean sampling depth (MSD) of the references we summarized (Figure 10.5) was quite shallow. MSD was 23 cm in West and Post (2002) but relatively similar in many of the other studies ranging from 26 to 33 cm (Gonzalez-Sanchez et al., 2012; Alvarez, 2005; Aigulera et al., 2013; Virto et al., 2012; VandenBygaart et al., 2003). Ogle et al. (2005) was using a dataset with deeper MSD (36 cm) but it is representing a SOC change integrated for the upper 30 cm of the soil, while MSD was 49 cm in Angers and Eriksen-Hamel (2008).

There is no obvious trend between MSD and the relative change or SOC sequestration rates (Figure 10.5). However, the study by Luo et al. (2010) with the deepest MSD (61 cm) presenting the lowest relative change (i.e., 3%). Furthermore, when comparing SOC stock changes for tillage treatments, the mass of soil for pre-defined depth increments may differ because of loosening or compaction (i.e., different dry soil bulk densities). Therefore, it is preferable comparing NT versus CT by accounting for differences in bulk densities using an equivalent soil mass approach (Ellert and Bettany, 1995). Among the above-mentioned studies, only Virto et al. (2012) were using such an approach, and results showing SOC sequestration rates in the lower rage of values (Figure 10.5). Another study (*n*=72, MD=18-yrs, MSD=35 cm) by Meurer et al. (submitted) also used an equivalent soil mass approach, this systematic review for boreo-temperate regions resulting in a SOC sequestration rate for NT less than 100 kg C ha^{-1} yr^{-1} (data not shown).

In the regional datasets, differences in the effect of NT on SOC stock changes were found between maritime and continental climates in Spain (Gonzalez-Sanchez et al., 2012), between eastern and western Canada (VandenBygaart et al., 2003) and within North American regions (Manley et al., 2005). However, in the studies with worldwide databases, only Ogle et al. (2005) found an interaction with climate, where variations occurred between tropical and temperate (both divided into moist and dry) regions. Three of the reviews made no such assessment (West and Post, 2002; Govaerts et al., 2009; Aigulera et al., 2013). VandenByggart et al. (2003) found that the effect of NT varied with soil classification (Great Groups), but for the other reviews, when it was considered in the analysis, no interactions was detected with soil texture (Alvarez, 2005; Angers and Eriksen-Hamel, 2008; Virto et al., 2012).

With only a few exceptions, there is apparently still difficultly (because of data availability) to address interactions with other management techniques. However, Luo et al. (2010) was able to observe that the effect of NT was only detectable for systems with two crops per year, while West and Post (2002) pointed out that the effect of NT was higher for crop rotations (as compared with continuous cropping systems). Furthermore, when excluding fallow treatments in the comparisons, the latter study were indicating a lower effect of NT, i.e., 480 versus 570 kg C ha^{-1} yr^{-1} including the comparisons with fallow. Aigulera et al. (2012) were also pointing out a negative effect of NT on SOC occurring for Olive orchards, as opposed to the treatment

comparisons made for cropland, attributing this to a lower input of crop residues in NT treatments for the former because of herbicide use.

Indeed, differences in NPP and thereby in annual C inputs to soil from crop residues are influencing the effect of tillage systems on SOC. Ogle et al. (2012) raised concerns about the negative effect NT might have on yields in North America, particularly under cooler and/or wetter conditions. In a review for European agro-ecosystems, Van den Putte et al. (2010) revealed an average potential yield reduction with NT (compared to CT) of 8.5%. Rasmussen (1999) reviewed results for Scandinavian countries and found a yield decrease of about 5% with NT and RT for small-grain cereals. Using a worldwide database, Pittelkow et al. (2015) showed an average potential yield reduction of 9.9% with NT. However, it is not always possible to include this factor in the assessments because of data limitations. Only the review by Virto et al. (2012) included this factor in their analysis, showing that it explained as much as 35% of the observed difference in SOC stocks between NT and CT.

10.4 Conclusions

On average over the reviews we considered for each management technique (Figure 10.1 to 10.5), the relative change is highest for manure, more or less identical for the effect of NT, AGCR handling and cover crops, and lowest for NF (Table 10.1). The SOC sequestration rates being fairly similar for manure, NT and cover crops, and lowest for AGCR handling and NF. As a comparison, the highest SOC sequestration rates still remains much lower than that typically observed for the effect of perennial forage crops (when the reference is annual crops), with values generally ranging from 500 to 600 kg C ha^{-1} yr^{-1} (e.g., VandenBygaart et al., 2010; Kätterer et al., 2013).

However, there is a large variation between the reviews for a given management technique. In particular the effect of AGCR handling and NT (Figure 10.3 and 10.5); the effect of manure and cover crops are relatively consistent between studies (Figure 10.1 and 10.4). The effect of AGCR handling is highly variable with crop types (e.g., maize vs. small-grain cereals), influencing the actual amount of annual crop residue C inputs that are affected. Furthermore, although repeated removal of small-grain cereal straw can eventually decrease soil quality and productivity in a given farmers field, the large-scale effect (e.g., regional, national) related to C cycling in the biosphere could be negligible because the straw is often returned somewhere with the manure.

Table 10.1. *The average effect of different management techniques calculated from review references on the relative change and soil organic C (SOC) sequestration rates, where numbers in parenthesis indicate total number of paired observations (n) made against a control treatment (from Bolinder et al. in preparation).*

Management technique	Relative change (%)	SOC sequestration rate (kg C ha^{-1} yr^{-1})
Recycled organic materials (manure)	36.0 (*269*)	298 (*108*)
No-tillage	10.5 (*792*)	279 (*457*)
Aboveground crop residue handling	10.4 (*963*)	118 (*258*)
Cover crops	9.4 (*129*)	330 (*187*)
Nitrogen fertilization	6.6 (*1119*)	175 (*51*)

The effect of NT, on average across a large number of reviews, shows a high relative change and SOC sequestration rate. Although some yield reduction is a reasonable expectation with NT and RT practices (e.g., 5 to 10%), unless NPP would decline exceedingly, a mean relative increase of approximately 10% more carbon in the topsoil with NT is quite well established. However, studies including a large proportion of deeper soil layers (Luo et al., 2010), and in combination with analysing the data on an equivalent depth basis such as Virto et al. (2012) and Meurer et al. (submitted), are suggesting a much lower SOC sequestration potential for this management technique. None of the reviews for the other management techniques considered an equivalent depth approach.

Accordingly with the reviews we summarized, there is a trend indicating a potentially larger number of observations available for studying the effect of no-tillage and aboveground crop residues handling, compared to that for manure and cover crops (Table 10.1). Recognizing that the data included are overlapping between many of the reviews, depending (among many other factors) on the selection criteria used. Since many LTEs are also including a zero N treatment as a reference, the number of observations allowing to assess the effect of N fertilization is also large, albeit it is not implicitly an objective of the LTEs as such. Besides, no N fertillization is not a common management practice but it provides useful baseline information in agro-ecosystem analysis. With the exception of cover crops, there is apparently and not so surprisingly, less observations allowing to express results as SOC sequestration rates, than for making relative change based assessments.

Despite the fact that some of the reviews and meta-analysis were made on very large databases, there was generally only limited interactions found between changes in SOC stocks and pedo-climatic conditions or crop types and rotations. The effect of climate was mentioned in only a few studies, while the effect of soil texture mostly was considered as “trends”. For the management techniques we considered here, there was also no differences detected between crop types or rotations. With the exception for AGCR handling, and for tillage practices where some studies were able to observe differences between very contrasting cropping systems,

such as two crops per year or fallow-based rotations versus single (one crop per year) cropping systems.

10.5 References

Aigulera E., Lassaletta L., GattingerA., Gimeno B.S. 2013. Managing soil carbon for climate change mitigation and adaptation in Mediterranean cropping systems: A meta-analysis. Agric. Ecosyst. Environ. 168: 25-36.

Alvarez R. 2005. A review of nitrogen fertilizer and conservation tillage effects on soil organic carbon storage. Soil Use and Management. 21: 38-52.

Anderson-Texeira K.J., Davis S.C., Masters M.D., Delucia E.H. 2009. Changes in soil organic carbon under biofuel crops. Global Change Biology and Bioenergy. 1: 75-96.

Angers D.A., Eriksen-Hamel N.S. 2008. Full-inversion tillage and organic carbon distribution in soil profiles: A meta-analysis. Soil Sci. Soc. Am. J. 72: 1370-1374.

Blanco-Canqui H. 2013. Crop residue removal for bioenergy reduces soil carbon pools: How can we offset carbon losses? Bioenerg. Res. 6: 358-371.

Bolinder M.A., Kätterer T., Poeplau C., Börjesson G., Parent L.E. 2015. Net primary productivity and below-ground crop residue inputs for root crops: Potato (*Solanum tuberosum* L.) and sugar beet (*Beta vulgaris* L.). Can. J. Soil Sci. 95: 87-93.

Bolinder M.A., Fortin J.G., Anctil F., Andren O., Kätterer T., de Jong R., Parent L.E. 2013. Spatial and temporal variability of soil biological activity in the Province of Quebec, Canada (45-58 °N, 1960-2009) – calculations based on climate records. Climatic Change. 117: 739-755.

Bolinder M.A., Janzen H.H., Gregorich E.G., Angers D.A., VandenBygaart A.J. 2007. An approach for estimating net primary productivity and annual carbon inputs to soil for common agricultural crops in Canada. Agric. Ecosyst. Environ. 118: 29-42.

Boehm M., Junkins B., Desjardins R., Kulshrestha S., Lindwall W. 2004. Sink potential of Canadian agricultural soils. Climatic Change. 65: 297-314.

Chantigny M.H. 2003. Dissolved and water-extractable organic matter in soils: a review on the influence of land use and management practices. Geoderma. 113: 357-380.

Christopher S.F., Lal R. 2007. Nitrogen management affects carbon sequestration in North American cropland soils. Critical Reviews in Plant Sciences. 26: 45-64.

Conant R.T., Easter M., Paustian K., Swan A., Williams S. 2007. Impacts of periodic tillage on soil C stocks: A synthesis. Soil Tillage Res. 95: 1-10.

Debreczeni K., Körschens M. 2003. Long-term field experiments of the world. Archives of Agronomy and Soil Science. 49: 465-483.

Dexter A.R., Richard G., Arrouays D., Czyz E.A., Jolivet C., Duval O. 2008. Complexed organic matter controls soil physical properties. Geoderma. 144: 620-627.

Diacono M., Montemurro F. 2010. Long-term effects of organic amendments on soil fertility: A review. Agron. Sustain. Dev. 30: 401-422.

Diochon A., Gillespie A.W., Ellert B.H., Janzen H.H., Gregorich E.G. 2016. Recovery and dynamics of decomposing plant residue in soil: an evaluation of three fractionation methods. Europ. J. Soil Sci. 67: 196-205.

Dungait J.A.J., Hopkins D.W., Gregory A.S., Whitmore A.P. 2012. Soil organic matter turnover is governed by accessibility not recalcitrance. Global Change Biology. 18: 1781-1796.

Ellert B.H., Bettany J.R. 1995. Calculation of organic matter and nutrients stored in soils under contrasting management regimes. Can. J. Soil Sci. 75: 529-538.

Fageria N.K. 2007. Green manuring in crop production. Journal of Plant Nutrition. 30: 691-719.

Favretto M.R., Paoletti M.G., Caporali F., Nannipieri P. et al. 1992. Invertebrates and nutrients in a Mediterranean vineyard mulched with subterranean clover (Trifolium subterraneum L.). Biology and Fertility of Soils. 14: 151-158.

Feller C., Beare M.H. 1997. Physical control of soil organic matter dynamics in the tropics. Geoderma. 79: 66-116.

Fontaine S., Mariotti A., Abbadie L. 2003. The priming effect of organic matter: a question of microbial competition?. Soil Biol. Biochem. 35: 837-843.

Franzluebbers A.J. 2011. Stratification of Soil Porosity and Organic Matter. In: Glinski J., Horabik J., Lipiec J. (Eds.). Encyclopedia of Agrophysics, Springer, Dordrecht, Heidelberg, London, New York. pp. 858-861.

Freibauer A., Rounsevell M.D.A., Smith P., Verhagen J. 2004. Carbon sequestration in the agricultural soils of Europe. Geoderma. 122: 1-23.

Gonzalez-Sanchez E.J., Ordonez-Fernandez R., Carbonell-Bojollo R., Veroz-Gonzalez O., Gil-Ribes J.A. 2012. Meta-analysis on atmospheric carbon capture in Spain through the use of conservation agriculture. Soil Tillage Res. 122: 52-60.

Govaerts B., Verhulst N., Castellanos-Navarrete A., Sayre K.D., Dixon J., Dendooven L. 2009. Conservation agriculture and soil carbon sequestration: Between myth and farmer reality. Critical Reviews in Plant Science. 28: 97-112.

Haddaway N.R., Hedlund K., Jackson L.E., Kätterer T., Lugato E., Thomsen I.K., Jorgensen H.B., Söderström B. 2015. Systematic Map – What are the effects of agricultural management on soil organic carbon in boreo-temperate systems?. Environmental Evidence. 4:23. 1-29.

Henin S., Dupuis M. 1945. Essai de bilan de la matiere organique du sol. Ann. Agron. 15: 17-29.

Izaurralde R.C., Rosenberg N.J., Lal R. 2001. Mitigation of climatic change by soil carbon sequestration: Issues of science, monitoring, and degraded lands. Advances in Agronomy. 70: 1-75.

Janzen H.H. 2015. Beyond carbon squestration: soil as conduit of solar energy. Eur. J. Soil Sci. 66: 19-32.

Kirchmann H., Börjesson G., Kätterer T., Cohen Y. 2017. From agricultural use of sewage sludge to nutrient extraction: A soil science outlook. Ambio. 46: 143-154.

Körschens M., Albert E., Baumecker M. et al. 2014. Humus und Klimaänderung – Ergebnisse aus 15 langjährigen Dauerfeldversuchen. Archives of Agronomy and Soil Science. 60: 1485-1517.

Körschens M., Albert E., Armbruster M. et al. 2013. Effect of mineral and organic fertilization on crop yield, nitrogen uptake, carbon and nitrogen balances, as well as soil organic carbon content and dynamics: results from 20 European long-term field experiments of the twenty-first century. Archives of Agronomy and Soil Science. 59: 1017-1040.

Kätterer T., Bolinder M.A., Thorvaldsson G., Kirchmann, H. 2013. Influence of ley-arable systems on soil carbon stocks in Northern Europe and Eastern Canada. EGF Series 'Grassland Science in Europe'. Vol 18: 47-56.

Kätterer T., Bolinder M.A., Berglund K., Kirchmann H. 2012. Strategies for carbon sequestration in agricultural soils in northern Europe. Acta Agriculturae Scand Section A. 1-18.

Kätterer T., Bolinder M.A., Andren O., Kirchmann H., Menichetti L. 2011. Roots contribute more to refractory soil organic matter than above-ground crop residues, as revealed by a long-term field experiment. Agric. Ecosyst. Environ. 141: 184-192.

Ladha J.K., Reddy C.K., Padre A.T., van Kessel C. 2011. Role of nitrogen fertilization in sustaining organic matter in cultivated soils. J. Environ. Qual. 40: 1756-1766.

Lal., R. 2013. Intensive agriculture and the soil carbon pool. Journal of Crop Improvement. 27: 735-751.

Lal R., Reicosky D.C., Hanson J.D. 2007. Evolution of the plow over 10,000 years and the rationale for no-till farming. Soil Tillage Res. 93: 1-12.

Lal R. 2004. Soil carbon sequestration impacts on global climate change and food security. Science. 304: 1623-1627.

Lehtinen T., Schlatter N., Baumgarten A., Bechini L. et al. 2014. Effect of crop residue incorpororation on soil organic carbon and greenhouse gas emissions in European agricultural soils. Soil Use and Management. 30: 524-538.

Liu C., Lu M., Cui J., Li B., Fang C. 2014. Effects of straw carbon input on carbon dynamics in agricultural soils: a meta-analysis. Global Change Biology. 20: 1366-1381.

Loveland P., Webb J. 2003. Is there a critical level of organic matter in the agricultural soils of temperate regions: a review. Soil Tillage Res. 70: 1-18.

Lu F. 2015. How can straw incorporation management impact on soil carbon storage? A meta-analysis. Mitig. Adapt. Strateg. Glob. Change. 20: 1545-1568.

Lu M., Zhou X., Luo Y., Yang Y., Fang C., Chen J., Li B. 2011. Minor stimulation of soil carbon storage by nitrogen addition: A meta-analysis. Agric. Ecosyst. Environ. 140: 234-244.

Luo Z., Wang E., Sun O.J. 2010. Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. Agric. Ecosyst. Environ. 139: 224-231.

Luo Z., Wang E., Sun O.J. 2010. Soil carbon change and its responses to agricultural practices in Australian agro-ecosystems: A review and synthesis. Geoderma. 155: 211-223.

Maillard E., Angers D.A. 2014. Animal manure application and soil organic carbon stocks: a meta-analysis. Global Change Biology. 20: 666-679.

Manlay J.R., Feller C., Swift M.J. 2007. Historical evolution of soil organic matter concepts and their relationships with the fertility and sustainability of cropping systems. Agric. Ecosyst. Environ. 119: 217-233.

Manley J., Van Kooten G.C., Moeltner K., Johnson D.W. 2005. Creating carbon offsets in agriculture through no-till cultivation: A meta-analysis of costs and carbon benefits. Climatic Change. 68. 41-65.

Meurer K.E., Haddaway N.R., Bolinder M.A., Kätterer T. submitted. Conservation tillage affects total SOC stocks in boreo-temperate regions only in the topsoil – conclusions from a systematic review using an equivalent soil mass approach.

McDaniel M.D., Tiemann L.K., Grandy A.S. 2014. Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics? A meta-analysis. Ecological Applications. 24: 560-570.

Merante P., Dibari C., Ferrise R., Sanchez B. et al. 2017. Adopting soil organic carbon management practices in soils of varying quality: Implications and perspectives in Europe. Soil Tillage Res. 165: 95-106.

Merante P., Dibari C., Ferrise R., Bindi M., Lesschen J.P., Kuikman P., sanchez B., Iglesias A. 2014. Report on critical low soil organic matter contents, which jeopardise good functioning of farming systems. Deliverable 2.1 for EC-funded Project SmartSOIL.

Morari F., Panagos P., Bampa F. 2014. Decline in organic matter in mineral soils. In Stolte J., Tesfai M., Oygarden L. (eds.). Soil in Europe – threats, functions and ecosystem services. A review report, deliverable 2.1 of the RECARE project.

Oelofse M., Markussen B., Knudsen L., Schelde K., Olesen J., Jenses L.S., Bruun S. 2015. Do soil organic carbon levels affect potential yields and nitrogen use efficiency? An analysis of winter wheat and spring barley field trials. Europ. J. Agronomy. 66: 62-73.

Ogle S.M., Breidt F.J., Paustian K. 2005. Agricultural management impacts on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. Biogeochemistry. 72: 87-121.

Olesen J.E., et al. 2014. A simplified model of management on SOC flows and stocks and crop yield. Deliverable 1.3 for EC-funded Project SmartSOIL.

Palese A.M., Vignozzi N., Celano G., Agnelli A.E. et al. 2014. Influence of soil management on soil physical characteristics and water storage in a mature rainfed olive orchard. Soil Tillage Res. 144: 96-109.

Paustian K., Lehmann J., Ogle S., Reay D., Robertson G.P., Smith P. 2016. Climate – smart soils. Nature. 532: 49-57.

Pittelkow C.M., Liang X., Linquist B.A., van Groenigen K.J. et al. 2015. Productivity limits and potentials of the principles of conservation agriculture. Nature. 527: 365-368.

Poeplau C., Don A. 2015. Carbon sequestration in agricultural soils via cultivation of cover crops – A meta-analysis. Agric. Ecosyst. Environ. 200: 33-41.

Poeplau C., Aronsson H., Myrbeck Å., Kätterer T. 2015. Effect of perennial ryegrass cover crop on soil organic carbon stocks in southern Sweden. Geoderma Regional. 4: 126-133.

Ponge J.F. 2013. Plant-soil feedbacks mediated by humus forms: A review. Soil Biol. Biochem. 57: 1048-1060.

Powlson D.S., Glendining M.J., Coleman K., Whitmore A.P. 2011. Implications for soil properties of removing cereal straw: Results from long-term studies. Agron. J. 103: 279-287.
Pribyl D.W. 2010. A critical review of the conventional SOC to SOM conversion factor. Geoderma. 156: 75-83.
Rasmussen K.J. 1999. Impact of ploughless soil tillage on yield and soil quality: A Scandinavian review. Soil Tillage. Res. 53: 3-14.
Rutkowska A., Pikula D. 2013. Effect of crop rotation and nitrogen fertilization on the quality and quantity of soil organic matter. http://dx.doi.org/10.5772/53229
Ruysschaert G., Poesen J., Verstraeten G., Govers G. 2006. Soil losses due to crop harvesting in Europé. In Boardman J., Poesen J. (eds.).Soil Erosion in Europe, John Wiley & Sons, Ltd.
Six J., Bossuyt H., Degryze S., Denef K. 2004. A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. Soil Tillage Res. 79: 7-31.
Smith P., Lanigan G., Kutsch W.L. et al. 2010. Measurements necessary for assessing the net ecosystem carbon budget of croplands. Agric. Ecosyst. Environ. 139: 302-315.
Smith P., Powlson D.S., Glendining M.J., Smith J.U. 1997. Potential for carbon sequestration in European soils: preliminary estimates for five scenarios using results from long-term experiments. Global Change Biology. 3: 67-79.
Sojka R.E., Upchurch D.R. 1999. Reservations regarding the soil quality concept. Soil Sci. Soc. Am. J. 63: 1039-1054.
Sommer M., Augistin J., Kleber M. 2016. Feedbacks of soil erosion on SOC patterns and carbon dynamics in agricultural landscapes – The CarboZALF experiment. Soil Tillage Res. 156: 182-184.
Sparling G., Parfitt R.L., Hewitt A.E., Schipper L.A. 2003. Three approaches to define desired soil organic matter contents. J. Environ. Qual. 32: 760-766.
Stockmann U., Adams M.A., Crawford J.W. et al. 2013. The knowns, known unknowns and unknowns of sequestration of soil organic carbon. Agric. Ecosyst. Environ. 164: 80-99.
Valkama E., Lemola R., Känkänen H., Turtola E. 2015. Meta-analysis of the effects of undersown catch crops on nitrogen leaching loss and grain yields in the Nordic countries. Agric. Ecosyst. Environ. 203: 93-101.
VandenBygaart A.J., Bremer E., McConkey B.G., Janzen H.H., Angers D.A., Carter M.R., Drury C.F., Lafond G.P., McKenzie R.H. 2010. Soil organic carbon stocks on long-term agroecosystem experiments in Canada. Can. J. Soil Sci. 90: 543-550.
VandenByggart A.J., Gregorich E.G., Angers D.A. 2003. Influence of agricultural management on soil organic carbon: A compendium and assessment of Canadian studies. Can. J. Soil Sci. 83: 363-380.
Van den Putte A., Govers G., Diels J., Gillijns K., Demuzere M. 2010. Assessing the effect of soil tillage on crop growth. A meta-regression analysis on European crop yields under conservation agriculture. Europ. J. Agronomy. 33: 231-241.
Virto I., Barre P., Burlot A., Chenu C. 2012. Carbon input differences as the main factor explaining the variability in soil organic C storage in no-tilled compared to inversion tilled agroecosystems. Biogeochemistry. 108: 17-26.
Wang J., Wang X., Xu M., Feng G. et al. 2015. Crop yield and soil organic matter after long-term straw return to soil in China. Nutr. Cycl. Agroecosyst. 102: 371-381.
West T.O., Post .M. 2002. Soil organic carbon sequestration rates by tillage and crop rotation: A global data analysis. Soil Sci. Soc. Am. J. 66: 1930-1946.
Wilhelm W.W., Johnson J.M.F., Hatfield J.L., Voorhees W.B., Linden D.R. 2004. Crop and soil productivity response to corn residue removal: A literature review. Agron. J. 96: 1-17.

11 Soil-improving cropping systems for reducing loss of soil biodiversity

F. Crotty, A. Termorshuizen, E. Hannula and E. Kandeler

11.1 Background

Soil is a critically important habitat to thousands of millions of organisms, it is also the basis for most food production. Yet, the agricultural revolution and intensification of land use are considered to be the main cause of soil biodiversity loss (Orgiazzi et al., 2016).

Soil quality refers to "*the continued capacity of soil to function as a vital living system, within ecosystems and land-use boundaries, to sustain biological productivity, maintain the quality of air and water environments, and promote plant, animal, and human health*" (Doran and Zeiss, 2000). The response of diversity, abundance and function to soil management constitutes an important aspect of soil quality (Mbuthia et al., 2015), and life within the soil (i.e. soil biodiversity) in essence represents the soil's health (Paoletti, 1999). However, only something that is considered alive can have health, thereby using this term we are (unconsciously) acknowledging that we regard soil as a living ecosystem and not just an inert base for agriculture.

Soils are among the most diverse habitats on earth (Bender et al., 2016) with millions of species inhabiting one gram of soil. Together these organisms participate in functions important to the maintenance of soil health, nutrient cycling and plant growth. The feeding relationships among them form the soil food web (Hunt et al., 1987) which ranges over multiple scales (micro, meso and macro) (Swift et al., 1979). Conversely, due to the large diversity of species there is also a considerable amount of functional redundancy (i.e. same function is being performed by multiple distinct groups of organisms), being referred to as the enigma of soil animal diversity (Anderson, 1975). This functional redundancy creates an inbuilt resilience of soil biodiversity to perturbations (Bengtsson, 2002).

Soil-improving cropping systems (SICS) are defined in this review as any agricultural management strategy that modifies the soil environment to improve the "health" or quality of the soil. A healthy soil is a sustainable soil that continues to produce profitable crop yields within agriculture, whilst also reducing biodiversity loss. Agroecosystems based on crop monoculture usually reduce the size and diversity of soil biota and consequently associated ecosystem functions. The recovery of soil biota through different management strategies has been postulated and this review chapter will focus on comparative studies of management systems that successfully modified the soil biota composition or functional diversity whilst remaining agriculturally viable at the field-scale. Soil improving cropping systems are any

agricultural management method that reduces the risk of soil biodiversity loss and have the potential to increase soil biodiversity over time to more "natural" levels (Figure 11.1).

Figure 11.1. *Visualisation of the effect of land management on soil biodiversity (A) and the effect soil improving cropping systems would have if implemented (B) (from Crotty et al. in preparation).*

If the landscape is severely modified through intensive agriculture, SICS implementation is going to have a greater effect on reducing soil biodiversity loss than if the landscape is in a more natural state e.g. organic farming. However, even in a more extensive system, SICS can still have an effect on reducing soil biodiversity loss (Figure 11.1).

There are many different types of SICS for soil biodiversity (Table 11.1) and these can be broadly grouped into five main categories – crop choice, physical/environmental change and water within the soil, chemicals, biological amendments and technology. Each SICS will affect biodiversity differently and different components of biodiversity will respond to different SICS. Alternative crop management practices, such as cover crops, addition of composted manures, and reducing or eliminating mineral fertilizers use, have emerged as integrated and ecologically sound approaches to increase soil organic matter levels (Kong et al., 2011). The choice of cover crops also can have an impact on biodiversity. For example, using deep rooting cover crops like chicory (*Cichorium intybus*) that 'mine' deep soil resources via their tap roots (Belesky et al., 2001), provide a greater habitat for earthworms in the following crop (Kautz et al., 2014), whilst also allowing subsequent crops to utilise deeper soil layers increasing access to nutrients and water from the subsoil (Perkons et al., 2014). Other cover crop species have been found to affect other soil organisms e.g. the beneficial arbuscular mycorrhizal fungi (AMF) (Benitez et al., 2016), nitrogen fixing bacteria, and the harmful soil-borne plant pathogens (Vukicevich et al., 2016). Several studies have also found increased biodiversity of fauna after cover crops in comparison to bare fallow (DuPont et al., 2009).

Measuring biodiversity is incredibly challenging (Tibbett, 2015) and compounded by the opacity of the medium the biota reside in, limited insight into feeding specificity and function (Crotty et al., 2011); as well as a general lack of taxonomic expertise. Blindly enhancing soil biodiversity infers random inclusion of many species (Bender et al., 2016), however maintaining or increasing everything may lead to the inclusion of a greater diversity of undesirable organisms, e.g. pests (Simon et al., 2010) and weeds (Sanyal and Shrestha, 2008). The use of species number as an indicator of an ecosystem's diversity suggests that all species are considered equal with respect to function (Bengtsson, 1998); however, this is unlikely to be true. Focusing on the increase in abundance of one group or species over time, dependent on cropping system (e.g. Pelosi et al., 2015); although an important consideration, is different to an increase in overall biodiversity. However, focusing on a group of organisms like earthworms' abundance and diversity may prove useful; earthworms are often referred to as ecosystem engineers because they affect the physical soil environment for other species living within it. A soil with a healthy earthworm population is likely to have abundant soil biodiversity throughout the soil food web.

Within two previous EU research funded projects investigating soil threats included soil biodiversity loss assessments – RECARE (2013-2018) and ENVASSO (2006-2008), three

biodiversity indicators were highlighted as the most important i) microbial respiration, ii) Collembola diversity and abundance, and iii) earthworm diversity, abundance and biomass. These three groups are representative of organisms across three different scales (micro, meso and macro) and, it is vital to ascertain an understanding of soil biodiversity across these scales; this chapter does this and also breaks down some of the organisms into the different functional groups (e.g. AMF vs plant pathogens).

Table 11.1. *Overview of potential soil improving cropping systems (from Crotty et al. in preparation). Cropping systems in italics are discussed more fully within main text.*

Key types	Cropping systems	
Crop selection and rotation	- *Cover crops* - Cash crops - Catch crops - Mixed cropping - Strip cropping - Companion cropping - *Intercropping* - Monoculture - Double cropping - Crop rotation - Deep rooting crops - Vegetable crops - Brassicacea inclusion - Nitrogen fixing plants	- Landraces (natives) - Permanent cropping (fruit) - Biodiverse mixes - Wild flower mixes - Pollinator mixes - *Headland alternatives* - *Grass or grass/clover leys* - Fallow - Set aside - Buffer strip - *Agroforestry* - *Energy crops/forests*
Physical soil environment changes and water	- *Non inversion tillage* - *Reduced tillage* - *Minimum tillage* - *Spike aeration* - Soil conservation - Terracing	- Soil stripping - Subsoiling - Contouring - *Drainage management* - Irrigation management - Drip irrigation
Chemicals	- *Plant protection products* - Pesticide use - Soil amendments - *BCSR amendment* - *Liming* - *Gypsum*	- Biofumigation - *Organic farming* - Solarisation
Biological amendments	- *Fertiliser* - *Manure / slurry addition* - *Biodigestate* - *Mulching / living mulch* - Composting - Residues - Compost tea	- *Biochar* - Woodchip - Bioaugmentation - Mycorrhizal amendments - Biostimulants - Paludiculture - *Biocontrol*
Technology	- *Precision farming* - *Controlled traffic* - Drones	- Low pressure tyres - Smaller machines

Microbiological criteria are often used as indicators of management driven changes in diversity, abundance and function of soil microorganisms (Kandeler 2015). Microbial biomass carbon and nitrogen characterise soil microorganisms as pools of easily available elements. The pattern of phospholipid fatty acids (PLFAs) gives evidence of microbial community composition (e.g. Bossio et al., 1998), the activities of enzymes describe the potential of soils to decompose organic compounds of different complexity (Mbuthia et al., 2015). Composition of the AMF community is generally determined by plant species present, plant diversity and soil nutrient status (Smith and Read 2002). Furthermore, hyphae can be physically damaged by mechanical actions such as ploughing. Earthworms represent the largest component of animal biomass within the soil and are commonly considered to be ecosystem engineers (Blouin et al., 2013). Ecosystem engineers are organisms that affect the whole environment and either directly or indirectly have an impact on the other species inhabiting the same space (Jones et al., 1994).

11.2 Purpose

The aim of this review of literature is to understand the impact of soil-improving cropping systems on reducing soil biodiversity losses. We focus on a selection of cropping systems highlighted in Table 11.1 that cover five main categories – crop choice, physical/environmental change and water within the soil, chemicals, biological amendments and technology. Each SICS will affect biodiversity differently and different components of biodiversity will respond to different SICS. A full overview of the results can be found in Crotty et al. (Review in preparation for Soil Biology and Biochemistry).

11.3 Results and discussions

11.3.1 Concept

Soil biotic activity is governed by the presence of nutrients (mainly nitrogen (N) and phosphorus (P)), carbon, and oxygen. Carbon is provided from root exudates and from organic matter; N from organic matter and from free-living and symbiotic N-binding bacteria. Oxygen is available through diffusion of air into the soil. Soils low in organic matter will likely sustain less soil life (particularly those species relying on SOM for food requirements (Ponge et al., 2013). Conditions leading to poor oxygen diffusion in soil will also lower soil life (fauna and aerobic microorganisms, although could lead to an increase in anaerobic microorganisms). From these two conditions, organic matter content and oxygen diffusion, many generalized predictions can be made about biological soil activity and diversity. All actions that increase soil organic matter content (e.g. diverse crop rotation, green manure cropping, intercropping, mulching, organic matter amendments) can be regarded as positive and vice versa (e.g. no crop rotation, tillage). Likewise, measures that lead to poor oxygen diffusion are negative (e.g. inundation, poor soil structure, low organic matter content) and consequently all measures that affect soil structure negatively (e.g. no crop rotation, heavy machinery, low organic matter content). Symbiotic organisms are generally performing better if they function under

conditions of low N (for N-binding bacteria) and low P (AMF). All SICS that affect soil structure positively will reduce the threat of soil biodiversity loss, whilst also increasing the disease suppressiveness of the soil, reducing opportunities for opportunistic plant pathogens such as *Pythium* spp to flourish.

Increasing the above ground diversity, maintaining an active rhizosphere throughout the year and by including a biological nitrogen source (legume crop) are strategies to maintain and improve soil microbial biodiversity and to sequester more C and N in agricultural ecosystems (Frasier et al., 2016a; Frasier et al., 2016b). Soil invertebrates are also strategically located forming a continuum of structures and processes linking basic microbial processes to the scale of fields and landscapes where ecosystem services are produced (Lavelle et al., 2006).

It is commonly acknowledged that maintaining soil biodiversity is key to improving and maintaining soil health, nutrient cycling and decomposition within the soil habitat (e.g. Crotty et al., 2015, Firbank et al., 2008 and Handa et al., 2014). However, the agriculture system usually prioritises current yields over other ecosystem (supporting) services provided by soil such as nutrient cycling or soil formation (Bender et al., 2016), functions largely performed by the soil organisms. Generally, all the different components of biodiversity in the soil are directly or indirectly affected by food and fibre production and cropping system (Table 11.2; at the end of this chapter). However, each group of organisms may be affected differently and it is thus important to understand how the mechanisms of agronomic practices affect these organisms and the functions they fulfil.

Conservation agriculture might mitigate biodiversity loss by different mechanisms. For example, any strategy in conservation agriculture which leads to the addition or re-distribution of crop residues will impact soil microorganisms. Studies have found more labile residues favour bacterial dominance in microbial community structure, whilst more recalcitrant residues favour fungi (Bossuyt et al., 2001; Kramer et al., 2012). Nevertheless, changes in microbial community composition and function after changes in soil management have to be ranked according to their relative importance of various environmental variables. Bossio et al. (1998) gave a useful synopsis of the order of importance in evaluation of SICS could follow: soil type > season > specific farming operation (e.g. cover crops, crop incorporation or side-dressing) > management system > spatial variation. Consequently, recommendations to mitigate high levels of biodiversity in agro-ecosystems have to consider specific pedoclimatic conditions.

Soil-borne plant pathogens are defined as plant pathogens that can survive a prolonged time within the soil. These organisms from various taxa including fungi, nematodes, chromists, protists and parasitic plants, are omnipresent, both in agricultural and natural systems. While in nature they are known to play essential regulatory roles related to plant biodiversity, in agriculture they are known as primary yield reducing factors, and the goal is to avoid their occurrence; once they appear at damaging levels it is necessary to control them as quickly and completely as possible; without destroying the rest of the soil biodiversity. Topics on the

management of soil-borne plant pathogens fall into two sections: (1) avoidance measures and (2) control measures. Although it is known that certain crops have a higher risk of encountering certain soil-borne plant pathogens, their presence is typically erratic, because their ability to disperse is quite limited. Also, a given crop may deal with a range of soil-borne pathogens simultaneously, each requiring specific attention from the farmer. It is not the scope of this paper to detail here the intricacies of the many host/pathogen combinations that exist but rather to evaluate at the level of crop system where the general tendencies occur.

There is often a discrepancy in the literature on the effects of soil biodiversity on plant productivity as soil biologists tend to measure the effects of soil biodiversity on other ecosystem services than crop yield. Whilst agronomists measure yield and monetary gain for a farmer without considering the role of biodiversity in maintaining soil and plant related functions. It is thus important to combine the available knowledge from experiments where these different focuses have been taken into account.

Modern plant breeding has reduced the dependency of plants to AMF, by selecting for plants better able to take up plant-available P and N (Duhamel and Vandenkoornhuyse 2013). Modern crop varieties have been selected to be adapted to modern agricultural practices and the soil conditions they lead to (Noguera et al., 2011); however, these same varieties may not grow well with changing agricultural practices as they won't be adapted to these new conditions. Commercial cultivars have been bred to be more resistant to soil-borne pathogens, therefore returning to historic varieties could increase soil-borne pathogen problems.

11.3.2 Crop selection and rotation

A number of studies have shown that when the intensity of agricultural practices increases the abundance and biodiversity of soil biota decreases (e.g. Tscharntke et al., 2005; Ponge et al., 2013; Bedano et al., 2016). Agricultural intensification, for example soil tillage, increased mineral fertiliser usage and crop diversity reductions (Postma-Blaauw et al., 2012), has been shown to affect abundances of taxonomic groups with larger body sizes (earthworms, enchytraeids, microarthropods and nematodes) more negatively than smaller-sized taxonomic groups (protozoans, bacteria and fungi) (Postma-Blaauw et al., 2010). Detrimental effects were found on taxonomic richness and diversity across taxonomic groups in the short term, following agricultural intensification; conversion from grassland to arable affected both functional group structure, abundance and taxonomic diversity of predatory mites (Postma-Blaauw et al., 2012), earthworms and nematodes (Crotty et al., 2016). Oehl et al. (2003) found that increased land use intensity was correlated with a decrease in AMF species number and the selection of species which sporulate rapidly. Ferrari et al. (2015) found PLFA and neutral lipid fatty acid (NLFA) profiles provided useful and complementary information of the footprints of different soil use and management. Conversion from grassland to arable lowers the SOM content and stability of the environment, introducing grass leys into a rotation will likely lead to positive SICS effects on biodiversity.

Crop rotation is the alternation of different crops in time and is the oldest and most important measure to avoid a build-up of populations of soil-borne pathogens at damaging levels. Replanting the same perennial crops can lead to phenomena often referred to as replant disease (Hoestra, 1967), and are generally caused by specific pathogens (e.g. asparagus: *Fusarium oxysporum* f. sp. asparagi; Blok and Bollen, 1996) or by consortia of pathogens (e.g. apple, Mazzola and Manici, 2012). By growing crops too frequently, pathogenic populations are likely to pop up in many crops (e.g. Bollen et al., 1989; Hwang et al., 2009). In proper rotations, populations decline to acceptable levels. For instance, populations of potato cyst nematode decline in a 3-4 year period during which less susceptible crops or resistant hosts are grown. Since many soil-borne pathogens have an erratic appearance, it is not always possible to say which rotation is the best. Designing optimal rotations is rather a function of the soil-borne pathogens that appear. Hence, the importance of monitoring of these pathogens. Leaving fields fallow (without crops; also referred to as black fallow – without crops but still applying herbicide) is another control measure to reduce the build-up of pathogens and weeds within the rotation. However, in the review by Vukicevich et al. (2016), showed that increasing plant diversity, increases soil microbial diversity and reduces population build-up of soil-borne pathogens, whilst also increasing beneficial microbes.

Crops differ in their ability to form arbuscular mycorrhizae (AM) associations and in the benefits they gain from them (Kiers et al., 2011; Higo et al., 2016). Most agricultural crop species, except members of the Cruciferae, Polygonaceae and the Chenopodiaceae, are able to form AM fungal associations (Sylvia and Chellemi, 2001). The degree to which these benefits are manifest is dependent on many factors, both biotic and abiotic. Since AM fungi are biotrophic, viability of AM hyphae gradually decreases in the absence of host plants such as during a fallow period. Harinikumar and Bagyaraj (1988) in India reported 40% reduction of AM inoculum in field soil after leaving the land fallow for one season. Long-fallow periods (>1 year) in northern Australia were associated with a decline in mycorrhizal colonization and AM sporulation in various crops (Thompson 1987). Studies have shown that the longer the time the soil is left bare for, the larger decrease in SOM; and also, the larger detrimental effect on beneficial organisms like earthworms and other soil biota (Bertrand et al., 2015).

11.3.3 Cover crops, catch crops, cash crops

Cover and catch crops refer to farming practices where plants are grown to help maintain soil productivity and fertility (rather than to be harvested) (Orgiazzi et al., 2016). In general, the use of cover and catch crops increase soil organic matter content and reduce soil compaction; and will therefore likely have a positive effect on maintaining soil biodiversity. Earthworm abundance and diversity is affected by cover crop species, for example Valckx et al. (2011) found ryegrass to be preferred over mustard, and phacelia and rapeseed residues were preferred over oats (Valckx et al., 2011). Nematode fauna was found to have twice the abundance in cover cropped fields compared to fallowed (DuPont et al., 2009).

Many studies consider the choice of cover crops that sustain the most diverse / most colonised AMF populations and how this subsequently affects the main crop (Hallama et al., *in prep*), particularly the response of P cycling. The selection of the best cover crop seems to be dependent on soil type and the species of main crop. However, using a non-mycorrhizal (or weakly mycorrhizal) cover crop, such as plants from the family Brassicaceae (Black and Tinker 1979); create a similar situation, from an AMF perspective, to leaving the ground fallow as there is no AMF host (Miller, 2000); and potentially having negative consequences on the following crop yields. The effect of a non-mycorrhizal cover crop (mustard) on the proceeding maize yields was found to be larger than the effect of P-fertilization and tillage (Gavito and Miller, 1998). Root and AMF hyphal fragments, which are important for early colonisation of the host plant, only survive for around 6 months in the soil (Tommerup and Abbot, 1981). Many studies have also found that *Trifolium* (clover) species are good hosts during off-season for AMF (Benitez et al., 2016). Given that the cover crops alter the AMF community composition (Heberle et al., 2015; Betidez et al., 2016; Garcia-Gonzalez et al., 2016), they are likely to affect the growth and P-uptake of subsequent crops.

Depending on their host status, crops grown additional to the main target crop (i.e. cover, catch or cash crops) are actually part of the crop rotation and thus may affect soil-borne pathogens. Some cover crops could act as multipliers of pathogens, such as certain *Meloidogyne* species (the causal agents of root knot disease). Similarly, certain weeds may act as sources for survival or multiplication of soil-borne plant pathogens (e.g. Freeman, 2008). Utilising a fallow period suppresses most populations of soil-borne pathogens, but less if weeds are allowed to develop. If the main risk factor for a farmer is soil-borne pathogens, keeping the ground black fallow is a preferred option, although this would have to fit in with environmental regulations for that country. There are exceptions however, firstly, if the farmer knows about the pathogens that are occurring, they can take this into account when choosing the crop species/cultivar (however, such soil assays are usually quite costly). Secondly, if the cover crop affects soil structure positively, this may act to suppress pathogens thriving on compacted soils (e.g. *Pythium* spp.). Thirdly, if crops are grown that specifically suppress pathogens (e.g. *Tagetes* if grown during summer is known to suppress the root lesion nematode *Pratylenchus penetrans*), this will reduce soil-borne pathogen risks.

11.3.4 Intercropping

Similar to cover crops, the selection of species of plants in intercropping will affect the outcome. In general, higher plant diversity leads to higher AMF diversity (Maherali and Klironomos 2007). For example, tomato intercropped with leek showed 20 % higher AMF colonisation rate (for tomato) than tomato intercropped with tomato (Hage-Ahmed et al., 2013). Fennel cropped with tomato on the other hand decreased the colonisation level of tomato by 13 %. Neither one of the treatments affected the biomass of tomato though. Intercropping has also been found to support large earthworm populations through increasing the food supply throughout the year (Schmidt and Curry, 2001; Schmidt et al., 2003), and

communities have even been found to be comparable to pasture and grass-legume leys (Schmidt et al., 2001). However, mixing two crops together could lead to the build-up of multiple soil-borne pathogens which may complicate setting up a proper rotation.

11.3.5 Grass/clover leys

Perennial fodder crops (e.g. grass/clover or lucerne) have been found to increase biomass and abundance of deep-burrowing (anecic) earthworms, improve soil structure and increase following crop yields (Kautz et al., 2010). Planting forages that are more favoured by certain soil fauna will lead to increases in abundance and biodiversity (e.g. white clover instead of ryegrass (Crotty et al., 2015)). Whilst perennial ley crops in general have been found to increase earthworm numbers (Kautz et al., 2014). Reductions in host species number has been found to reduce AMF diversity (Burrows and Pfleger, 2002). When AMF diversity was measured under different crops and adjacent semi-natural grasslands, it was found that native grasslands and plots where clover was grown had greater number of AMF compared to plots with continuous wheat or barley (Menéndez et al., 2001). Cultivating clover after wheat restored the AMF diversity and increased spore numbers over three years to resemble numbers in semi-natural grasslands (Oehl et al., 2003).

11.3.6 Agroforestry / agro-sylviculture

There is limited research on the effects of agroforestry on soil biodiversity within European climates and soil types, with most research focusing on (sub)tropical regions. However, it is well known that the perennial nature of most trees will have a profound impact on soil properties and hence, soil biodiversity, abundance and function (Barrios et al., 2013). There is a strong link between above and below-ground organisms creating both positive and negative feedback between the two (Wardle et al., 2004), for example trees will affect soil temperature, moisture, erosion, and nutrient cycling (Barrios et al., 2013). Biodiversity conservation is also one of the main ecosystem services / environmental benefits of agroforestry often reported (Jose, 2009). Generally, trees share limited soil-borne pathogens with annual crops, though exceptions to this rule exist (e.g. *Verticillium dahliae*, the causal agent of Verticillium wilt), which is able to infect a manifold of hosts including olive tree, cotton and potatoes. Most trees form symbiosis with ectomycorrhizal fungi but some have also AMF as a partner. For example, poplars and sugar maple have been shown to benefit greatly from the symbiotic relationship with the fungus (da Silva Sousa, 2013). In studies done in (sub)tropics it has been shown that presence of trees in plots increased sporulation, mycorrhizal colonization of the crop species and number of AMF propagules in the plant roots (e.g. da Silva Sousa, 2013).

11.3.7 Energy forests / Biofuel crops

In areas with limited choice of rotation crops, energy crops may be a good addition for widening crop rotation. Where energy crops are perennial, negative effects of soil tillage are alleviated. Abundance and diversity of earthworms found in energy forests was greater than neighbouring arable fields, due to the absence of tillage, increase in organic matter layer and environmental buffering (Lagerlof et al., 2012). Earthworms were also found to increase in

perennial energy crops in comparison to silage maize (Emmerling, 2014), and even *Miscanthus* had positive effects on earthworm communities (Felten and Emmerling, 2011). Most of the plants used as biofuel crops are fast growing and benefit from forming AMF especially in low nutrient conditions or during drought. A study conducted in Canada identified that abundance of the AMF was significantly higher in the herbaceous perennial grasses (switchgrass and *Miscanthus*) than in woody species (poplar and willow) used for biofuel production (Mafa-Attoye 2015). In this study the addition of chemical fertilizers did not affect the colonization of AMF. However, where natural grasslands have been converted to bioenergy crops, the impact of land-use change was the main driver of biodiversity change (Desiree et al., 2014).

11.3.8 Tillage

Tillage is performed to prepare the seed bed for the next crop, whilst also removing weeds, and in some cases soil-borne pathogens, from the top-soil layer. During tillage, the bulk density of the soil is reduced, with greater amount of air spaces being created. This leads to a faster decline of organic matter (OM) compared to no-tillage systems. This fast decline in OM provides a temporary boost to the bacteria and fungi that decompose OM, followed by a dip in activity. The heavy equipment used for tillage may lead to subsoil compaction, which in turn can affect rooting ability and water infiltration. Potentially leading to inundated soils during times of heavy rainfall which could be detrimental to crop yields and soil biodiversity. There is a general agreement that tillage intensity influences microbial abundance and function (Ahl et al., 1999, Kandeler et al., 1999a, b, Kraus et al., 2017). The major outcome of different studies showed that reduced or zero tillage changed spatial distribution of residues leading to a re-distribution of soil microorganisms within the upper 40 cm of the soil profile. Changes of spatial pattern of microorganisms might also have large consequences for the decay of organic pollutants (like pesticides in the top soil).

The impact of tillage practises on AMF has been reviewed (Kabir 2005). In short, most of the studies have found reduction in the number of AMF taxa colonizing roots in systems with conventional tillage compared to reduced or no-tillage systems. This is likely to be caused by dilution effect when AMF is abundant in topsoil and mixed with large volumes of soil. Under no-till, AM fungi survive better, particularly when they are close to the host crop on which they developed (although this is dependent on crop rotation). There is also evidence on the tillage changing community composition of AMF (Kabir 2005; Jansa et al., 2003). Mechanisms for this include: (1) the differences in tolerance to the tillage-induced disruption of the hyphae among the different AMF species, (2) changes in nutrient content of the soil, (3) changes in microbial activity, or (4) changes in weed populations in response to soil tillage (Jansa et al., 2003). The timing of tillage also seems to have a large effect on AMF diversity; autumn tillage has been shown to cause reduced AM hyphal viability, whereas spring tillage had little effect on AM hyphal viability (Kabir et al., 1997). This is caused by the hyphae being detached from the host plant.

All soil fauna impacted by ploughing will benefit from no-till or reduced till management (Orgiazzi et al., 2016). The detrimental effect of tillage on earthworm community composition and abundance is often dependent on the intensity (Emmerling, 2001) and frequency – the less intensively the soil is disturbed, the less harmful tillage is for earthworms (Bertrand et al., 2015). Tillage is known to be detrimental to other soil fauna including Collembola (Bedano et al., 2006a), Acari (Bedano et al., 2006b) and to a lesser extent nematodes (Fiscus and Neher, 2002). Spike aeration or the use of different tine options to alleviate compaction in wheelings of arable crops (Niziolomski et al., 2016) or within pasture (Cournane et al., 2011) has potential to improve soil structure, whilst also reducing runoff, phosphorus and nitrogen losses (DeLaune et al., 2013). However, to date the impact of this soil improving cropping system on soil biodiversity loss has not been investigated in any form.

Since inoculum of soil-borne pathogens is usually in the top soil layer, soil tillage moves the inoculum to deeper soil layers, thus enlarging the distance to the stem base and lowering disease risk. Deep tillage has even been applied to manage soil-borne pathogens specifically (Katan, 2010) to enable the cultivation of cash crops on certain fields. No-tillage is the reverse of deep tillage and thus, if they are present, soil-borne pathogens remain in the top soil where they can do more damage than when they are moved to deeper soil layers. Another negative side-effect of no-tillage is that pathogens residing on crop residues (e.g. on stubbles) survive better than when they are incorporated into the soil (Bockus and Shroyer, 1998).

11.3.9 Drainage management

Drought decreases soil water content and has been found to decrease microarthropod species richness (Tsiafouli et al., 2005). Drainage and irrigation may encourage multiplication of the more robust species such as the Prostigmata and Astigmata within the Acari (Behan-Pelletier, 2003). Soil drainage has also been found to impact the community structures of actinomycetes and pseudomonads (Clegg et al., 2003). Some soil-borne pathogens thrive well under wet soil conditions, such as species of *Pythium* and *Phytophthora*, as well as black-grass (*Alopecurus myosuroides*); therefore, proper drainage management will alleviate this. Also, poor soil drainage decreases tolerance of crops to many opportunistic plant pathogens.

11.3.10 Plant protection products (pesticides)

All slug pellets (of different formulations / active ingredients) have been found to be deleterious to non-target soil invertebrates (e.g. metaldehyde (Santos et al., 2010) although relatively less toxic compared to alternatives (iron phosphate) for earthworm survival (Langan and Shaw, 2006; Edwards et al., 2009). Reducing the number and amount of slug pellet applications, and following manufacturers instructions, will reduce the risks of soil biodiversity loss. Using pesticides formulated as bait pellets (rather than a broad spray), have been found to have no effect on the density and diversity of soil meso and macrofauna (Salvio et al., 2011).

Nematicides and fumigants are often used to control the soil-borne nematode pests, however brassicaceae-based management strategies as a soil biodiversity-friendly alternative have been

shown to also be effective at managing the top-three economically important nematode pests (root-knot (*Meloidogyne*), cyst (*Heterodera* and *Globodera* and lesion (*Pratylenchus*) nematodes) (Fourie et al., 2016). Addition of nematode biocontrol agents have been shown not to affect earthworm or mesofauna abundance (Iglesias et al., 2003). Nematicides have been shown to impact all nematode trophic groups (Timper et al., 2012), also including those nematodes that predate on plant pathogens (Hofman and Jongebloed, 1988) and may have a lingering effect due to the changes in the soil food web of other invertebrates over the same time period (niche filling). Specific crops can be grown that actively control soil-borne nematodes. In this context, three mechanisms can be applied: (1) growing *Tagetes* (marigold) which causes *Pratylenchus* to intoxicate after they attempt to penetrate the roots, (2) growing resistant crops that hatch nematodes, and (3) biofumigation, i.e. incorporating brassicaceous crops which form toxic compounds (isothiocyanates) during their decomposition (Larkin and Griffin, 2007). Currently the use of *Tagetes* is used in practice, although growers will lose a growing season as the crop has to be grown during summer. The same applies for the cultivation of hatch crops. The most common application of a hatch crop in Europe is the use of *Solanum sisymbriifolium* against potato cyst nematode. AMF can also be considered to act as a type of biocontrol agent, as well as exchanging carbon with the host plant it can also help to defend the plant from attack (Johansson et al., 2004).

In general, nematodes can be controlled using nematicides and the other methods described above, but fungicides effective against soil fungal pests are much less available. Soil fungicides are inactive against most fungal soil-borne pathogens except *Rhizoctonia solani* and the fungal-like *Pythium* and *Phytophthora* spp. (causing damping-off and root rot). For instance, against Fusarium diseases (causing root rot and wilt in many crops) no fungicides are available. There is a significant body of literature about negative side-effects of applying soil pesticides (Siepel, 1996, Firbank et al., 2008). Therefore, it is important to create an integrated pest management strategy, applying pesticides through "spot application" or when an outbreak is greater than the "economic threshold" for damage occurring.

Schreiner and Bethlenfalvay (1997) examined the effect of fungicides (e.g. captan) on spore germination and colonisation of pea by two species of AMF. All fungicides were capable of reducing spore germination, AMF root colonisation or spore production, but the interactions were highly variable and depended on AMF species, fungicide combinations and environmental factors. Furthermore, studies have found an increase in colonization rates after application of fungicides in reduced rates (Sreenivasa and Bagyaraj 1989). Similarly, the same study found that three nematicides used reduced AMF colonisation and spore production at recommended application rates. Yet, when applied at half of the recommended rate, all three nematicides increased spore production and had a neutral or positive effect on root colonisation.

Miller and Jackson (1998) showed that weeds are important hosts maintaining AMF when growing non-mycorrhizal crops, forming a mycorrhizal bridge between mycorrhizal crops.

However, intensive agricultural practice aims for the elimination of all weeds, removing this mycorrhizal bridge thus exacerbating the effect of non-mycorrhizal crops, especially where non-mycorrhizal crops are grown consecutively. Therefore, a SICS strategy could be to mix planting a non-mycorrhizal crop with something that encourages AMF (e.g. clover).

11.3.11 Soil chemical amendments

Reducing the acidity of soil through liming has been found to promote earthworm abundance in the laboratory (Davidson and Grieve, 2006) and in the field over time (Hirth et al., 2009). However, there is the potential for other groups with lower optimum pH requirements (e.g. fungi) to be detrimentally effected (Murray et al., 2006) although this has been disproven in some studies (Treonis et al., 2004). Furthermore, liming has been shown to reduce Collembola species diversity (Chagnon et al., 2001) and can also reduce the abundance of mites (Hagvar and Amundsen, 1981). Liming is an appropriate measure to control *Plasmodiophora brassicae*, the causal agent of club root on brassicas. These conflicting results on the effects of liming on different forms of soil biodiversity, indicate that more research is needed on the impact of liming on soil biodiversity loss. The addition of gypsum (mined or flue gas desulphurised) has been found to reduce earthworm abundance and biomass in some instances (Chen et al., 2014) but also in this case, more research is needed. Currently although a lot of historic research has occurred into the Basic Cation Saturation Ratio (BCSR amendment), the data to date do not support the claims of the BCSR (Kopittke and Menzies, 2007) and thus is not a useful method of reducing the threat of soil biodiversity loss.

11.3.12 Organic farming

Organic farming, replaces the use of synthetic inputs (fertilisers, herbicides, pesticides) with more natural ones (slurry, farm yard manure, larger crop rotation, intercropping and tillage (to terminate crops)). Organic farming is often promoted as a way to enhance sustainability of agriculture whilst decreasing environmental impacts (Bedoussac et al., 2015). The effect of organic farming practises as SICS to improve the soil AMF diversity, was evaluated in a study in Netherlands across 26 arable sandy soil sites (Verbruggen et al., 2010). The average number of AMF taxa in the plant roots was highest in natural grasslands, intermediate in organically managed fields and significantly lower in conventionally managed fields. Moreover, AMF richness increased significantly with the time since conversion to organic agriculture. AMF communities of organically managed fields were also more similar to those of natural grasslands when compared with those under conventional management, and were less uniform than their conventional counterparts (Verbruggen et al., 2010). There are multiple explanations why organic farm management has positive effect on AMF richness and diversity. Firstly, organic farms often utilise a wider crop rotation, often with a grass–clover mixture as a forage crop. The inclusion of legumes in crop rotations can have positive effects on AMF diversity. Secondly, organic farming appears to select for AMF species with long life cycles (Oehl et al., 2003).

Organic farming in comparison to synthetic fertilisers, pesticides and fumigation has been found to increase the abundance and diversity of soil biota particularly nematodes and earthworms, whilst synthetic mineral fertilisers have been found to detrimentally affect enchytraeids and Diptera larvae (Birkhofer et al., 2008). However, the use of some "natural chemicals" in organic farming recommended for use against plant pathogens are toxic to various soil organisms e.g. copper at recommended concentrations is toxic to earthworms and many microorganisms (van Bruggen et al., 2016). It is likely that it is the combination of organic farming principles (and extensive management) – reduced tillage, wider crop rotations, that reduce soil biodiversity loss, rather than each individual factor working on its own (Overstreet et al., 2010). For organic farming systems, maintaining wide rotation is by far the most commonly used method to keep soil-borne pathogens at acceptable levels. Wide rotations of 1:6 or more are regularly occurring, including 2-3 years of grass, are commonly used in these systems.

11.3.13 Physical methods to inactivate soil-borne pathogens

Physical methods to inactivate soil-borne pathogens include treatments by heat (soil solarisation (normally done in the field), or soil steaming (normally occurring in greenhouses)); by induced anaerobiosis (biological soil disinfestation (Blok et al., 2000)); and by inundation; or combinations of these three methods. Physical methods to control soil-borne pathogens can be quite effective against a wide spectrum of pathogens. However, these methods are expensive and therefore applied only in capital-intensive crops. Physical methods to inactivate pathogens are not selective, thus also affecting many non-target organisms including AMF and earthworms. Some studies of biological soil disinfestation have been shown to have only a moderate impact on native beneficial microorganisms (Momma et al., 2010). Whilst steam disinfestation of soil has an extremely detrimental effect on soil microarthropods immediately after steaming (Fenoglio et al., 2006). The effect of soil steaming on earthworm populations has currently not been investigated to date. Solarisation was found to have no effect on arthropod communities – with abundance and diversity similar to non-solarised samples (da Silva et al., 2009). Again, an integrated strategy needs to occur, to reduce the effects of these methods on non-target soil biodiversity.

11.3.14 Fertiliser applications

Fertilisation may affect AMF growth and root colonization ability (Liu et al., 2016) by altering the concentration of soil mineral nutrients and shifting the N:P ratio of plant tissues, which in turn may stimulate the growth of AMF populations more adapted to the new nutritional conditions. Although AMF have been described as natural biofertilisers (Berruti et al., 2016) most farmers would not rely on this. AMF is usually less diverse in agricultural systems due to the over-supply of nutrients artificially, reducing the need for crop plants to invest in their relationship with the fungus. Organic fertilisers are thought to be more favourable for AMF (see organic farming).

The use of inorganic fertiliser has been found to reduce the abundance of Collembola, Oribatid mites, Enchytraeidae, and earthworms (Siepel, 1996; Yeates et al., 1997). Earthworm abundance and diversity increases with organic matter input and content of soils, therefore when an using an organic fertiliser source should increase numbers (Lapied et al., 2009). Nematode abundance was also found to increase within organic production with cumulative benefits when this was combined with minimum tillage compared to inorganic fertiliser and pesticide inputs (Overstreet et al., 2010). Long term application of swine slurry did influence AMF and their products (glomalin) in the soil environment (Balota et al., 2014) potentially causing a decrease in soil aggregation. However, the application of animal manures as nutrient sources generally increases the abundance and activity of other soil biota (particularly nematodes, Collembola, Acari and earthworms) (Altieri, 1999; Wu et al., 2013; Orgiazzi et al., 2016). Organic amendments will stimulate soil microbial activity, thereby potentially increasing the disease suppressiveness of the soil as soil-borne pathogens are out-competed.

Anaerobic digestate has a decreased pathogen load in comparison to manure/slurry (Insam et al., 2015), and has been found to increase microbial biomass within the soil (Nkoa, 2014). Digestate does not appear to have a negative effect on earthworms exposed directly to it at low concentrations (Pivato et al., 2016).

11.3.15 Biochar amendments

There have been very few studies on the effect of biochar on soil fauna and little in temperate soils. Those that have investigated the addition of biochar have shown that microbial biomass is increased (Lehmann et al., 2011). However, AMF abundance did not increase if there was an already abundant nutrient supply (i.e. in agricultural environment). Experiments manipulating the number of earthworms and biochar showed that rice yields increased the most when both were added together (Noguera et al., 2011); however little is known about what occurs at the field scale.

11.3.16 Mulching / living mulch / green residues

Mulch is usually plant material that is partially decomposed left on the soil surface to form a cover, it can however also be plastic that covers the soil, reducing the amount of water infiltration and warming the soil up. Organic mulch biomass is a source of carbon and nutrients required for soil biological activity (Orgiazzi et al., 2016). Mulching allows for the maintenance of a greater organic layer on the soil, reducing soil erosion, stabilising temperatures and reducing water losses. A variety of mulches can be used in-row in perennial horticultural cropping systems, these mulches have been found to increase the abundance of protozoa, bacterivorous nematodes and enrichment opportunist nematodes in comparison to bare ground or polyethylene covering (Forge et al., 2003). Long term use of living mulch and organic fertiliser have been found to increase earthworm populations by between 1.5-2.3 times greater than conventionally fertilised populations (Pelosi et al., 2015). Incorporation of green residues of crops can pose a risk for some opportunistic pathogens such as *Pythium* spp. and *Rhizoctonia solani* able to multiply on these residues, especially if the main crop is sown shortly

after residue incorporation (Manici et al., 2004). Thus, in presence of such pathogens, the timing of incorporation of green residues relative to sowing time is of importance.

Mulching with organic residues and green residues are unlikely to have direct effects on AMF but will affect the saprotrophic fungal community by providing a nutrient source. Increase in saprotrophic fungi and fungal to bacterial ratios have been observed in multiple studies following an addition of green manure or mulch (i.e. Miura et al., 2013; Ramos et al., 2015; Frac et al., 2009). This is likely to help build up the soil organic carbon pool through actions of saprotrophic fungi (e.g. Miura et al., 2015).

11.3.17 Mycorrhizal amendments

The use of commercial (laboratory grown) inoculants containing non-resident AMF is an emerging technology in field crop production. Adding AMF in soils has been thought to enhance crop yield and protect plants from biotic and abiotic stresses. However, carrying out an open-field, extensive inoculation treatment is often technically impractical and economically prohibitive and is advised only if native AMF population is not present. The current perspective in literature is to rather use other AMF-friendly management systems (such as cover cropping and conservation tillage) to conserve the native AMF-population (Berruti et al., 2016).

The most commonly commercially used fungus is *Rhizophagus intraradices* (formerly known as *Glomus irregulare*) but the trend now is to use a mixed inoculum containing spores from multiple species imitating the natural community. A study on pea showed that using a mixed inoculum from three species of AMF increased the plant biomass and N and P uptake compared to inoculation with single species of AMF (Lin et al., 2013). However, the natural community in the fields performed almost as well as the added mixed community indicating that in this case the inoculation did not yield extra benefit. Indeed, the effects of inoculants on native soil populations is largely unknown and needs to be understood before large scale amendments are done (Rodriquez and Sanders 2015). The added AMF inoculum has been shown to be an effective and an economical option to restore degraded soils with little of its native biodiversity left (Gulati and Cummings, 2008; Berruti et al., 2016). The overall success of mycorrhizal inoculation has been reviewed recently by Berruti et al. (2016). However, very little research has occurred on the effect of mycorrhizal amendments on soil biodiversity in general.

11.3.18 Precision agriculture

Precision farming (basing the application of fertilisers on pesticides on historic field data, and planting at variable seed rates) potentially can affect AMF abundance and diversity. For example, through the precision application of fungicides (only directly affecting specific areas, leaving the wider AMF community intact). The application of fertiliser using precision farming practice (fertilising areas differently based on the average yield of the previous three years and whether they had previously been high or low yielding) has been shown to change the proteolytic activity of microbes within a low yielding system (Schloter et al., 2003). However, there is limited research on how precision farming affects soil biodiversity in general. Remote

sensing to detect plant stress could be used to pinpoint plant problems e.g. plant parasitic nematodes or fungal pathogens (Hillnhutter et al., 2011), these could then be treated at "site specific" locations rather than the "whole field" approach that has been historically used (Liu et al., 2014) – saving money and reducing the potential threat to soil biodiversity loss.

11.3.19 Controlled traffic

Heavy trafficked areas have been shown to have a detrimental impact on soil structure and hence on density and biomass of all three earthworm functional groups (Bottinelli et al., 2014); utilising a controlled traffic system would reduce this impact across the whole farm. Reduction in Collembola has been shown to be due to mechanical perturbations produced by conventional agricultural practice (Bedano et al., 2006a), utilising controlled traffic as a SICS would reduce the impact of this. This effect would be similar for AMF populations, with reduction in perturbations increasing population sizes.

Table 11.2. *Effect of organisms on food and fibre production overview (red text indicates negative effects) (from Crotty et al. in preparation).*

Organism	Effect on food and fibre production	How cropping system can effect organism?
Bacteria (and Archae)	- Increase nutrient availability - Promote plant growth - Reduce pathogens - Process/modify agrochemicals and xenobiotics - Change soil composition - Enhance soil structure - Soil-borne pathogens	- Increase food source (OM / amendments) - Flush of nutrients - Change environment of soil (water/temperature) - Kill organism directly (pesticides)
Fungi	- Cycling of essential nutrients - Promote plant growth - Increase N and P availability (through symbiosis) - Reduce pathogens - Process/modify agrochemicals and xenobiotics - Change soil composition - Enhance C allocation and build up OM - Decreased seedling mortality - Biocontrol against pests - Enhance soil structure (through hyphal growth and glomalins secreted by AMF) - Decomposition of plant residues (and subsequent release of nutrients). - Soil-borne pathogens	- Increase food source (OM / amendments / maintain plant residues / covercrops) - Flush of nutrients to provide burst of growth - Change structure of soil (destroy hyphal network) - Kill organism directly (pesticides) - Remove host plant - Removal of residues of plant (food source)
Protozoa	- Enhance microbial growth - Increase nutrient availability - Soil-borne pathogens	- Flush of nutrients - Increase food source (OM / amendments) - Change environment of soil (water/temperature) - Kill organism directly (pesticides)
Rotifers	- Contribute to nutrient cycling - Flocculation of bacteria	- Flush of nutrients - Change environment of soil (water/temperature) - Kill organism directly (pesticides)

Organism	Effect on food and fibre production	How cropping system can effect organism?
Tardigrades	- Enhance microbial growth	- Flush of nutrients - Change environment of soil (water/temperature) - Kill organism directly (pesticides)
Nematodes	- Increase nutrient availability - Disperse bacteria and fungi - Reduce pathogens - Soil-borne pathogens	- Flush of nutrients - Change structure of soil - Increase food source (OM / amendments) - Change environment of soil (water/temperature) - Kill organism directly (pesticides)
Collembola	- Increase nutrient availability - Breakdown plant material, animal carcases - Faecal pellets contribute to soil microstructure and fertilisation. - Disperse microorganisms and nematodes, - Micro-ecosystem engineer (Brussaard et al., 1997) - Consumer of pathogens - Host for parasites	- Flush of nutrients - Increase food source (OM / amendments) - Change environment of soil (water/temperature) - Change structure of soil - Kill organism directly (pesticides)
Mites	- Increase nutrient availability - Breakdown plant material, animal carcases - Faecal pellets contribute to soil microstructure and fertilisation. - Disperse microorganisms and nematodes, - "Micro-ecosystem engineer" (Brussaard et al., 1997) - Consumer of pathogens - Host for parasites/parasitoids	- Flush of nutrients - Increase food source (OM / amendments) - Change environment of soil (water/temperature) - Change structure of soil - Kill organism directly (pesticides)
Soil dwelling immature invertebrates e.g. beetle larvae, fly larvae	- Fragmentation and decomposition of organic material - Change pH of soil passing through gut	- Flush of nutrients - Increase food source (OM / amendments) - Change environment of soil (water/temperature) - Change structure of soil - Remove food source (plant species) - Kill organism directly (pesticides)

Organism	Effect on food and fibre production	How cropping system can effect organism?
Other mesofauna (body width less than 2mm) e.g. protura, dipluran, pseudoscorpions, beetles, spiders, thysanoptera,	- Increase organic matter through burial of dung or carcasses - Predators of pests (pseudoscorpions, spiders etc)	- Flush of nutrients - Increase food source (OM / amendments) - Change environment of soil (water/temperature) - Change structure of soil - Remove food source (plant species) - Kill organism directly (pesticides)
Myriapoda (centipedes and millipedes mainly and pauropoda, symphylan)	- Excreta contribute to coprogenic humus	- Flush of nutrients - Increase food source (OM / amendments) - Change environment of soil (water/temperature) - Change structure of soil - Remove food source (plant species) - Kill organism directly (pesticides)
Enchytraeids	- Fragmentation and breakdown of plant litter - Enhance microbial growth, - Change soil structure (bioturbation) - Disperse of microorganisms	- Flush of nutrients - Increase food source (OM / amendments) - Change environment of soil (water/temperature) - Change structure of soil - Remove food source (plant species) - Kill organism directly (pesticides)
Root herbivorous insects / pests	- Modifies plant performance - Yield losses - Changes plant physiology - Transmits diseases	- Remove food source (plant species) - Introduce host food source - Change structure of soil - Kill organism directly (pesticides)
Earthworms	- Enhance microbial growth, - Change soil structure (bioturbation) - Disperse microorganisms - Aids sporulation / germination of fungal spores. - Improves water infiltration - Ecosystem engineer	- Flush of nutrients - Increase food source (OM / amendments) - Change environment of soil (water/temperature) - Change structure of soil - Remove food source (plant species) - Kill organism directly (pesticides)

Organism	Effect on food and fibre production	How cropping system can effect organism?
Ants	- Enhance microbial growth - Disperse plant propagules - Change soil structure (bioturbation) - Increase porosity and drainage - Reduce bulk density - Ecosystem engineer	- Flush of nutrients - Change environment of soil (water/temperature) - Change structure of soil - Remove food source (plant species) - Kill organism directly (pesticides)
Termites	- Enhance macroporosity and infiltration - Change soil structure (bioturbation) - Enhance microbial growth, - Ecosystem engineer	- Flush of nutrients - Change environment of soil (water/temperature) - Change structure of soil - Remove food source (plant species) - Kill organism directly (pesticides)

11.4 Conclusions

The European Commission defined the soil as having seven basic functions: 1) Biomass production (including agriculture and forestry); 2) Storing, filtering and transforming nutrients, substances and water. 3) Biodiversity pool, (habitats, species and genes); 4) Physical and cultural environment for humans and human activities; 5) Source of raw material; 6) Carbon pool; 7) Archive of geological and archaeological heritage; (EC, 2006). Agricultural production is only one of these seven soil functions, biodiversity pool is actually one of the others; it is up to SICS to prevent the loss of one function at the expensive of another. Soil biodiversity, abundance and function are important aspects of soil quality, and acknowledge that soil is a living ecosystem. Decline of soil biodiversity relates to a loss of diversity of living organisms in soil and their inter-relationships; it may occur as a result of poor soil management. The decline may relate to i) species diversity, ii) genetic diversity, and/or iii) functional diversity.

Biodiversity-specific SICS may involve three mechanisms, i.e., (i) changes in inputs, (ii) substitution, and (iii) redesign. The first mechanism relates to inputs of energy – increasing organic matter as substrate, changing the available nitrogen source used. The second mechanism relates to possible substitution of chemical (pesticides), physical (tillage) and/or biological measures (mycorrhizal amendments). Thirdly, the redesign mechanism relates to the diversification of crop rotations, i.e., various crop types in sequence and/or in mixtures (intercropping), cover crops, fallow crops, set-aside, and the inclusion of hedges and other landscape elements (Table 11.3).

Numerous studies have shown that agricultural intensification decreases the abundance and biodiversity of soil biota. However, there are examples of measures and practices that combine high crop yields with promoting soil biodiversity.

Most promising biodiversity-specific SICS relate to the diversification of crop rotation by providing a greater range of food sources, increasing soil organic matter, and reducing the build-up of soil-borne pathogens. Reducing the intensity of tillage will also reduce soil biodiversity loss (conventional tillage is known to have a detrimental effect on many groups of organisms from AMF to earthworms). Reducing pesticide use also helps, as well as controlled traffic (less compaction).

Table 11.3. *Qualitative assessment of biodiversity-specific SICS. Scores are quantified as positive 1 to 3 (3 is most positive), neutral = 0, and negative -1 to -3 (with -3 most negative) (from Crotty et al., in preparation).*

	Components of cropping systems	Components of biodiversity-specific SICS	Change in profitability	Changes in soil properties		
				Physical	Chemical	Biological
A	Crop rotations	Wide (1:6) crop rotations +intercropping +cover crops, green manures	0	+2	0	+3
B	•Nutrient management	Manuring	+1	0	+2	+3
C	•Irrigation management	Optimal	+2	0	+1	0
D	•Drainage management	Optimal	+3	+3	+1	0
E	•Tillage management	Reduced tillage	0	+3	0	+3
F	•Pest management	Integrated pest management	+1	0	0	+1
G	•Weed management	Mechanical weeding	-2	-1	0	+1
H	•Residue management	Residue return	0	+1	+1	+2
J	•Mechanization management	Controlled trafficking	+1	+2	0	+2
K	•Landscape management	Treelines, hedges, fringes	+2	+1	+1	+2

11.5 References

Ahl C., Joergensen R.G., Kandeler E., Meyer B., Woehler V. (1999) Microbial biomass and activity in silt and sand loams after long-term reduction in tillage using the "Horsch" system. Soil and Tillage Research 49, 93-104.

Altieri, M.A., 1999. The ecological role of biodiversity in agroecosystems. Agriculture, Ecosystems and Environment 74, 19-31.

Anderson, J.M., 1975. The Enigma of Soil Animal Species, In: Vanek, J. (Ed.), Progress in Soil Zoology: Proceedings of the Fifth International Colloquium on Progress in Soil Zoology. W Junk / Prague Academia, The Hague, pp. 51-58.

Balota E.L., Machineski O., Honda C., Yada I.F.U., Barbosa G.M.C., Nakatani A.S., Coyne M.S., 2014. Response of arbuscular mycorrhizal fungi in different soil tillage systems to long-term swine slurry application. Land Degradation and Development,

Barrios, E., Sileshi, G.W., Shepherd, K., Sinclair, F., 2013. Agroforestry and Soil Health: Linking Trees, Soil Biota, and Ecosystem Services.

Bedano, J.C., Cantu, M.P., Doucet, M.E., 2006a. Soil springtails (Hexapoda : Collembola), symphylans and pauropods (Arthropoda : Myriapoda) under different management systems in agroecosystems of the subhumid Pampa (Argentina). European Journal of Soil Biology 42, 107-119.

Bedano, J.C., Cantú, M.P., Doucet, M.E., 2006b. Influence of three different land management practices on soil mite (Arachnida: Acari) densities in relation to a natural soil. Applied Soil Ecology 32, 293-304.

Bedano, J.C., Dominguez, A., Arolfo, R., Wall, L.G., 2016. Effect of Good Agricultural Practices under no-till on litter and soil invertebrates in areas with different soil types. Soil and Tillage Research 158, 100-109.

Bedoussac, L., Journet, E.P., Hauggaard-Nielsen, H., Naudin, C., Corre-Hellou, G., Jensen, E., Prieur, L., Justes, E., 2015. Ecological principles underlying the increase of productivity achieved by cereal-grain legume intercrops in organic farming. A review. Agronomy for Sustainable Development 35, 911-935.

Behan-Pelletier, V.M., 2003. Acari and Collembola biodiversity in Canadian agricultural soils. Canadian Journal of Soil Science 83, 279-288.

Belesky, D.P., Turner, K.E., Fedders, J.M., Ruckle, J.M., 2001. Mineral composition of swards containing forage chicory. Agronomy Journal 93, 468-475.

Bender, S.F., Wagg, C., van der Heijden, M.G.A., 2016. An Underground Revolution: Biodiversity and Soil Ecological Engineering for Agricultural Sustainability. Trends in Ecology and Evolution 31, 440-452.

Bengtsson, J., 1998. Which species? What kind of diversity? Which ecosystem function? Some problems in studies of relations between biodiversity and ecosystem function. Applied Soil Ecology 10, 191-199.

Bengtsson, J., 2002. Disturbance and resilience in soil animal communities. European Journal of Soil Biology 38, 119-125.

Benitez, M.-S., Taheri, W.I., Lehman, R.M., 2016. Selection of fungi by candidate cover crops. Applied Soil Ecology 103, 72-82.

Berruti, A., Lumini, E., Balestrini, R., Bianciotto, V., 2016. Arbuscular Mycorrhizal Fungi as Natural Biofertilizers: Let's Benefit from Past Successes. Frontiers in Microbiology 6. 1559

Bertrand, M., Barot, S., Blouin, M., Whalen, J., de Oliveira, T., Roger-Estrade, J., 2015. Earthworm services for cropping systems. A review. Agronomy for Sustainable Development 35, 553-567.

Birkhofer, K., Bezemer, T.M., Bloem, J., Bonkowski, M., Christensen, S., Dubois, D., Ekelund, F., Fliessbach, A., Gunst, L., Hedlund, K., Maeder, P., Mikola, J., Robin, C., Setala, H., Tatin-Froux, F., Van der Putten, W.H., Scheu, S., 2008. Long-term organic farming fosters below and aboveground biota: Implications for soil quality, biological control and productivity. Soil Biology and Biochemistry 40, 2297-2308.

Black, R., Tinker, P.B., 1979. The development of endomycorrhizal root systems. II. Effect of agronomic factors and soil conditions on the development of vesicular-arbuscular mycorrhizal infection in barley and on the endophyte spore density. New Phytologist 83: 401-413

Blok, W.J., Lamers, J.G., Termorshuizen, A.J., Bollen, G.J., 2000. Control of soilborne plant pathogens by incorporating fresh organic amendments followed by tarping. Phytopathology 90, 253-259.

Blouin, M., Hodson, M.E., Delgado, E.A., Baker, G., Brussaard, L., Butt, K.R., Dai, J., Dendooven, L., Peres, G., Tondoh, J.E., Cluzeau, D., Brun, J.J., 2013. A review of earthworm impact on soil function and ecosystem services. European Journal of Soil Science 64, 161-182.

Bossio, D.A., Scow, K.M., Gunapala, N., Graham, K.J., 1998. Determinants of Soil Microbial Communities: Effects of Agricultural Management, Season, and Soil Type on Phospholipid Fatty Acid Profiles. Microbial Ecology 36, 1-12.

Bossuyt, H., Denef, K., Six, J., Frey, S.D., Merckx, R., Paustian, K., 2001. Influence of microbial populations and residue quality on aggregate stability. Applied Soil Ecology 16, 195-208.

Bottinelli, N., Capowiez, Y., Ranger, J., 2014. Slow recovery of earthworm populations after heavy traffic in two forest soils in northern France. Applied Soil Ecology 73, 130-133.

Brussaard, L., Behan-Pelletier, V.M., Bignell, D.E., Brown, V.K., Didden, W., Folgarait, P., Fragoso, C., Freckman, D.W., Gupta, V., Hattori, T., Hawksworth, D.L., Klopatek, C., Lavelle, P., Malloch, D.W., Rusek, J., Soderstrom, B., Tiedje, J.M., Virginia, R.A., 1997. Biodiversity and ecosystem functioning in soil. Ambio 26, 563-570.

Burrows, R.L., Pfleger, F.L., 2002. Arbuscular mycorrhizal fungi respond to increasing plant diversity. Canadian Journal of Botany 80: 120-130

Chagnon, M., Pare, D., Hebert, C., Camire, C., 2001. Effects of experimental liming on collembolan communities and soil microbial biomass in a southern Quebec sugar maple (Acer saccharum Marsh.) stand. Applied Soil Ecology 17, 81-90.

Chen, L., Kost, D., Tian, Y., Guo, X., Watts, D., Norton, D., Wolkowski, R.P., Dick, W.A., 2014. Effects of Gypsum on Trace Metals in Soils and Earthworms. Journal of Environmental Quality 43, 263-272.

Clegg, C.D., Lovell, R.D.L., Hobbs, P.J., 2003. The impact of grassland management regime on the community structure of selected bacterial groups in soils. Fems Microbiology Ecology 43, 263-270.

Cournane, F.C., McDowell, R.W., Littlejohn, R.P., Houlbrooke, D.J., Condron, L.M., 2011. Is mechanical soil aeration a strategy to alleviate soil compaction and decrease phosphorus and suspended sediment losses from irrigated and rain-fed cattle-grazed pastures? Soil Use and Management 27, 376-384.
Crotty, F.V., Blackshaw, R.P., Murray, P.J., 2011. Tracking the flow of bacterially derived ^{13}C and ^{15}N through soil faunal feeding channels. Rapid Communications in Mass Spectrometry 25, 1503-1513.
Crotty, F.V., Fychan, R., Sanderson, R., Rhymes, J.R., Bourdin, F., Scullion, J., Marley, C.L., 2016. Understanding the legacy effect of previous forage crop and tillage management on soil biology, after conversion to an arable crop rotation. Soil Biology and Biochemistry 103, 241-252.
Crotty, F.V., Fychan, R., Scullion, J., Sanderson, R., Marley, C.L., 2015. Assessing the impact of agricultural forage crops on soil biodiversity and abundance. Soil Biology and Biochemistry 91, 119-126.
da Silva Sousa, C., Cezar Menezes, R.S., de Sá Barreto Sampaio, E.V., de Sousa Lima, F., Oehl, F., Costa Maia, L., 2013. Arbuscular mycorrhizal fungi within agroforestry and traditional land use systems in semi-arid Northeast Brazil. Acta Scientiarum: Agronomy 35: 307
da Silva, M.G., de Oliveira, C.M., Junqueira, A.M.R., 2009. Solarization, organic and chemical fertilization combined effects on arthropods community in soil cultivated with lettuce. Horticultura Brasileira 27, 465-472.
Davidson, D.A., Grieve, I.C., 2006. Relationships between biodiversity and soil structure and function: Evidence from laboratory and field experiments. Applied Soil Ecology 33, 176-185.
DeLaune, P.B., Sij, J.W., Krutz, L.J., 2013. Impact of soil aeration on runoff characteristics in dual-purpose no-till wheat systems. Journal of Soil and Water Conservation 68, 315-324.
Desiree, J.I., Pita, A.V., Floor, V.D.H., Andre, P.C.F., 2014. Biodiversity impacts of bioenergy crop production: a state-of-the-art review. Global Change Biology Bioenergy 6, 183-209.
Doran, J.W., Zeiss, M.R., 2000. Soil health and sustainability: managing the biotic component of soil quality. Applied Soil Ecology 15, 3-11.
Duhamel, M., Vandenkoornhuyse, P., 2013. Sustainable agriculture: possible trajectories from mutualistic symbiosis and plant neodomestication. Trends in Plant Science 18: 597-600
DuPont, S.T., Ferris, H., Van Horn, M., 2009. Effects of cover crop quality and quantity on nematode-based soil food webs and nutrient cycling. Applied Soil Ecology 41, 157-167.
Edwards, C.A., Arancon, N.Q., Vasko-Bennett, M., Little, B., Askar, A., 2009. The relative toxicity of metaldehyde and iron phosphate-based molluscicides to earthworms. Crop Protection 28, 289-294.
Emmerling, C., 2001. Response of earthworm communities to different types of soil tillage. Applied Soil Ecology 17, 91-96.
Emmerling, C., 2014. Impact of land-use change towards perennial energy crops on earthworm population. Applied Soil Ecology 84, 12-15.
European Commission (EC), 2006. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions. Thematic Strategy for Soil Protection, COM 231 Final, Brussels.
Felten, D., Emmerling, C., 2011. Effects of bioenergy crop cultivation on earthworm communities—A comparative study of perennial (Miscanthus) and annual crops with consideration of graded land-use intensity. Applied Soil Ecology 49, 167-177.
Fenoglio, S., Gay, P., Malacarne, G., Cucco, M., 2006. Rapid recolonization of agricultural soil by microarthropods after steam disinfestation. Journal of Sustainable Agriculture 27, 125-135.
Ferrari A.E., Ravnskov S., Larsen J., Tonnersen T., Maronna R.A., Wall L.G., 2015. Crop rotation and seasonal effects on fatty acid profiles of neutral and phospholipids extracted from no-till agricultural soils. Soil Use and Management 31, 165-175.
Firbank, L.G., Petit, S., Smart, S., Blain, A., Fuller, R.J., 2008. Assessing the impacts of agricultural intensification on biodiversity: a British perspective. Philosophical Transactions of the Royal Society B: Biological Sciences 363, 777-787.
Fiscus, D.A., Neher, D.A., 2002. Distinguishing sensitivity of free-living soil nematode genera to physical and chemical disturbances. Ecological Applications 12, 565-575.

Forge, T.A., Hogue, E., Neilsen, G., Neilsen, D., 2003. Effects of organic mulches on soil microfauna in the root zone of apple: implications for nutrient fluxes and functional diversity of the soil food web. Applied Soil Ecology 22, 39-54.

Fourie, H., Ahuja, P., Lammers, J., Daneel, M., 2016. Brassicacea-based management strategies as an alternative to combat nematode pests: A synopsis. Crop Protection 80, 21-41.

Frac, M., Siczek, A., Lipiec, J., 2009. The effect of mulching and soil compaction on fungi composition and microbial communities in the rhizosphere of soybean. EGU General Assembly Conference Abstracts 11: 2009 EGUGA 11.2596F

Frasier, I., Noellemeyer, E., Figuerola, E., Erijman, L., Permingeat, H., Quiroga, A., 2016a. High quality residues from cover crops favor changes in microbial community and enhance C and N sequestration. Global Ecology and Conservation 6, 242-256.

Frasier, I., Quiroga, A., Noellemeyer, E., 2016b. Effect of different cover crops on C and N cycling in sorghum NT systems. Science of the Total Environment 562, 628-639.

Gavito, M.E., Miller, M.H., 1998. Changes in mycorrhiza development in maize induced by crop management practices. Plant and Soil 198: 185-192.

Gulati, A., Cummings, J. R. (2008). Mycorrhiza, a fungal solution for the farm economy. Econ. Times. Available online at: http://articles.economictimes.indiatimes.com/2008-03-12/news/27728495_1_fertiliser-subsidy-fertiliser-consumption-nitrogenous

Hage-Ahmed, K., Krammer, J., Steinkellner, S., 2013. The intercropping partner affects arbuscular mycorrhizal fungi and Fusarium oxysporum f. sp. lycopersici interactions in tomato. Mycorrhiza 23: 543-550

Hagvar, S., Amundsen, T., 1981. Effects of liming and artificial acid-rain on the mite (acari) fauna in coniferous forest. Oikos 37, 7-20.

Handa, I.T., Aerts, R., Berendse, F., Berg, M.P., Bruder, A., Butenschoen, O., Chauvet, E., Gessner, M.O., Jabiol, J., Makkonen, M., McKie, B.G., Malmqvist, B., Peeters, E.T.H.M., Scheu, S., Schmid, B., van Ruijven, J., Vos, V.C.A., Hattenschwiler, S., 2014. Consequences of biodiversity loss for litter decomposition across biomes. Nature 509, 218-221.

Harinikumar, D.M. and Bagyaraj, D.J. (1988). Effect of crop rotation on native vesicular-arbuscular mycorrhizal propagules in soil. Plant Soil. 110: 77–80.

Heberle, E.S., de Armas, R.D., Heberle, D.A., Stürmer, S.L., Peruch, L.A.M., Lovato, P.E., Sousa Soares, C.R.F., 2015. Occurrence and structure of arbuscular mycorrhizal fungal communities in cassava after cultivation of cover crops as observed by the "PCR-DGGE" technique. Revista Brasileira de Ciencia do Solo 39, 1292-1301

Higo, M., Isobe, K., Miyazawa, Y., Matsuda, Y., Rhae A. Drijber, R.A., Torigo, Y., 2016. Molecular diversity and distribution of indigenous arbuscular mycorrhizal communities colonizing roots of two different winter cover crops in response to their root proliferation. The Journal of Microbiology 54, 86-97

Hillnhutter, C., Mahlein, A.K., Sikora, R.A., Oerke, E.C., 2011. Remote sensing to detect plant stress induced by Heterodera schachtii and Rhizoctonia solani in sugar beet fields. Field Crops Research 122, 70-77.

Hirth, J.R., Li, G.D., Chan, K.Y., Cullis, B.R., 2009. Long-term effects of lime on earthworm abundance and biomass in an acidic soil on the south-western slopes of New South Wales, Australia. Applied Soil Ecology 43, 106-114.

Hunt, H.W., Coleman, D.C., Ingham, E.R., Ingham, R.E., Elliott, E.T., Moore, J.C., Rose, S.L., Reid, C.P.P., Morley, C.R., 1987. The Detrital Food Web in a Shortgrass Prairie. Biology and Fertility of Soils 3, 57-68.

Iglesias, J., Castillejo, J., Castro, R., 2003. The effects of repeated applications of the molluscicide metaldehyde and the biocontrol nematode Phasmarhabditis hermaphrodita on molluscs, earthworms, nematodes, acarids and collembolans: a two-year study in north-west Spain. Pest Management Science 59, 1217-1224.

Insam, H., Gomez-Brandon, M., Ascher, J., 2015. Manure-based biogas fermentation residues - Friend or foe of soil fertility? Soil Biology and Biochemistry 84, 1-14.

Jansa, J., Mozafar, A., Kuhn, G., Anken, T., Ruh, R., Sanders, I.R., Frossard, E. (2003) Soil Tillage Affects the Community Structure of Mycorrhizal Fungi in Maize Roots. Ecological Applications 13: 1164-1176

Johansson, J.F., Paul, L.R., Finlay, R.D., 2004. Microbial interactions in the mycorrhizosphere and their significance for sustainable agriculture. FEMS Microbiology Ecology 48, 1-13.

Jones, C.G., Lawton, J.H., Shachak, M., 1994. Organisms as ecosystem engineers. Oikos 69, 373-386.

Jose, S., 2009. Agroforestry for ecosystem services and environmental benefits: an overview. Agroforestry systems 76, 1-10.

Kabir, Z., 2005. Tillage or no-tillage: Impact on mycorrhizae. Canadian Journal of Plant Science

Kandeler E., 2015. Physiological and biochemical methods for studying soil biota and their function. In: Soil Microbiology, Ecology, and Biochemistry (editor: E.A. Paul). Fourth edition, Elsevier, pp 187-222.

Kandeler E., Palli S., Stemmer M. and Gerzabek M.H., 1999a. Tillage changes microbial biomass and enzyme activities in particle-size fractions of a Haplic Chernozem. Soil Biology and Biochemistry 31, 1253 – 1264.

Kandeler E., Tscherko D. and Spiegel H., 1999b. Long-term monitoring of microbial biomass, N-mineralisation and enzyme activities of a Chernozem under different tillage management. Biology and Fertility of Soils 28, 343-351.

Kautz, T., Luesebrink, M., Paetzold, S., Vetterlein, D., Pude, R., Athmann, M., Kuepper, P.M., Perkons, U., Koepke, U., 2014. Contribution of anecic earthworms to biopore formation during cultivation of perennial ley crops. Pedobiologia 57, 47-52.

Kautz, T., Stumm, C., Kosters, R., Kopke, U., 2010. Effects of perennial fodder crops on soil structure in agricultural headlands. Journal of Plant Nutrition and Soil Science 173, 490-501.

Kiers, E.T., Duhamel, M., Beesetty, Y., Mensah, J.A., Franken, O., Verbruggen, E., Fellbaum, C.R., Kowalchuk, G.A., Hart, M.M., Bago, A., Palmer, T.M., West, S.A., Vandenkoornhuyse, P., Jansa, J., Bücking, H., 2011. Reciprocal rewards stabilize cooperation in the mycorrhizal symbiosis. Science 333, 880-882.

Kong, A.Y.Y., Scow, K.M., Córdova-Kreylos, A.L., Holmes, W.E., Six, J., 2011. Microbial community composition and carbon cycling within soil microenvironments of conventional, low-input, and organic cropping systems. Soil Biology and Biochemistry 43, 20-30.

Kopittke, P.M., Menzies, N.W., 2007. A review of the use of the basic cation saturation ratio and the "ideal" soil. Soil Science Society of America Journal 71, 259-265.

Kramer, S., Marhan, S., Ruess, L., Armbruster, W., Butenschoen, O., Haslwimmer, H., Kuzyakov, Y., Pausch, J., Scheunemann, N., Schoene, J., Schmalwasser, A., Totsche, K.U., Walker, F., Scheu, S., Kandeler, E., 2012. Carbon flow into microbial and fungal biomass as a basis for the belowground food web of agroecosystems. Pedobiologia 55, 111-119.

Kraus M., Krause H.M., Spangler S., Kandeler E., Behrens S., Kappler A., Mäder P., Gattinger A., 2017. Tillage system affects fertilizer-induced nitrous oxide emissions. Biology and Fertility of Soils 53, 49-59.

Lagerlof, J., Palsson, O., Arvidsson, J., 2012. Earthworms influenced by reduced tillage, conventional tillage and energy forest in Swedish agricultural field experiments. Acta Agriculturae Scandinavica Section B-Soil and Plant Science 62, 235-244.

Langan, A.M., Shaw, E.M., 2006. Responses of the earthworm Lumbricus terrestris (L.) to iron phosphate and metaldehyde slug pellet formulations. Applied Soil Ecology 34, 184-189.

Lapied, E., Nahmani, J., Rousseau, G.X., 2009. Influence of texture and amendments on soil properties and earthworm communities. Applied Soil Ecology 43, 241-249.

Lavelle, P., Decaens, T., Aubert, M., Barot, S., Blouin, M., Bureau, F., Margerie, P., Mora, P., Rossi, J.P., 2006. Soil invertebrates and ecosystem services. European Journal of Soil Biology 42, S3-S15.

Lehmann, J., Rillig, M.C., Thies, J., Masiello, C.A., Hockaday, W.C., Crowley, D., 2011. Biochar effects on soil biota - A review. Soil Biology and Biochemistry 43, 1812-1836.

Liu, W., Zhang, Y.L., Jiang, S.S., Deng, Y., Christie, P., Murray, P.J., Li, X.L., Zhang, J.L., 2016. Arbuscular mycorrhizal fungi in soil and roots respond differently to phosphorus inputs in an intensively managed calcareous agricultural soil. Scientific Reports 6.

Liu, Z., Griffin, T., Kirkpatrick, T.L., 2014. Statistical and Economic Techniques for Site-specific Nematode Management. Journal of Nematology 46, 12-17.

Mafa-Attoye, T., 2015. Indications of Shifting Microbial Communities Associated with Growing Biomass Crops on Marginal Lands in Southern Ontario. Major Paper http://hdl.handle.net/10214/9281

Maherali, H., Klironomos, J.N., 2007. Influence of phylogeny on fungal community assembly and ecosystem functioning. Science 316, 1746-1748.

Mbuthia, L.W., Acosta-Martínez, V., DeBruyn, J., Schaeffer, S., Tyler, D., Odoi, E., Mpheshea, M., Walker, F., Eash, N., 2015. Long term tillage, cover crop, and fertilization effects on microbial community structure, activity: Implications for soil quality. Soil Biology and Biochemistry 89, 24-34.

Menéndez A.B., Scervino J.M., Godeas, A.M., (2001). Arbucular mycorrhizal populations associated with natural and cultivated vegetation on a site of Buenos Aires province, Argentina. Biology and Fertility of Soils 33: 373-381.

Miller, M.H. (2000). Arbuscular mycorrhizae and the phosphorus nutrition of maize: A review of the Guelph studies. Canadian Journal of Plant Sciences 80: 47-52.

Miller, R.L., Jackson, L.E., 1998. Survey of vesicular arbuscular mycorrhizae in lettuce production in relation to management and soil factors. Journal of Agricultural Science 130, 173-182.

Miura, T., Niswati, A., Swibawa, I.G., Haryani, S., Gunito, H., Kaneko, N., 2013. No tillage and bagasse mulching alter fungal biomass and community structure during decomposition of sugarcane leaf litter in Lampung Province, Sumatra, Indonesia. Soil Biology and Biochemistry 58, 27-35.

Miura, T., Niswati, A., Swibawa, I.G., Haryani, S., Gunito, H., Shimano, S., Fujie, K., Kaneko, N., 2015. Diversity of fungi on decomposing leaf litter in a sugarcane plantation and their response to tillage practice and bagasse mulching: implications for management effects on litter decomposition. Microbial Ecology 70: 646-658

Momma, N., Momma, M., Kobara, Y., 2010. Biological soil disinfestation using ethanol: effect on Fusarium oxysporum f. sp. lycopersici and soil microorganisms. Journal of General Plant Pathology 76, 336-344.

Murray, P.J., Cook, R., Currie, A.F., Dawson, L.A., Gange, A.C., Grayston, S.J., Treonis, A.M., 2006. Interactions between fertilizer addition, plants and the soil environment: Implications for soil faunal structure and diversity. Applied Soil Ecology 33, 199-207.

Niziolomski, J.C., Simmons, R.W., Rickson, R., Hann, M.J., 2016. Tine options for alleviating compaction in wheelings. Soil and Tillage Research 161, 47-52.

Nkoa, R., 2014. Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review. Agronomy for Sustainable Development 34, 473-492.

Noguera, D., Laossi, K.R., Lavelle, P., de Carvalho, M.H.C., Asakawa, N., Botero, C., Barot, S., 2011. Amplifying the benefits of agroecology by using the right cultivars. Ecological Applications 21, 2349-2356.

Oehl, F., Sieverding, E., Ineichen, K., Mader, P., Boller, T. and Wiemken A., 2003. Impact of land use intensity on the species diversity of arbuscular mycorrhizal fungi in agroecosystems of Central Europe. Applied and Environmental Microbiology 69, 2816–2824.

Orgiazzi, A., Bardgett, R.D., Barrios, E., Behan-Pelletier, V., Briones, M.J.I., Chotte, J.-L., De Deyn, G.B., Eggleton, P., Fierer, N., Fraser, T., Hedlund, K., Jeffery, S., Johnson, N.C., Jones, A., Kandeler, E., Kaneko, N., Lavelle, P., Lemanceau, P., Miko, L., Montanarella, L., Moreira, F.M.S., Ramirez, K.S., Scheu, S., Singh, B.K., Six, J., van der Putten, W.H., Wall, D.H., 2016. Global Soil Biodiversity Atlas. Publications Office of the European Union, Luxembourg.

Overstreet, L.F., Hoyt, G.D., Imbriani, J., 2010. Comparing nematode and earthworm communities under combinations of conventional and conservation vegetable production practices. Soil and Tillage Research 110, 42-50.

Paoletti, M.G., 1999. Using bioindicators based on biodiversity to assess landscape sustainability. Agriculture Ecosystems and Environment 74, 1-18.

Pelosi, C., Bertrand, M., Thenard, J., Mougin, C., 2015. Earthworms in a 15 years agricultural trial. Applied Soil Ecology 88, 1-8.

Perkons, U., Kautz, T., Uteau, D., Peth, S., Geier, V., Thomas, K., Holz, K.L., Athmann, M., Pude, R., Koepke, U., 2014. Root-length densities of various annual crops following crops with contrasting root systems. Soil and Tillage Research 137, 50-57.

Pivato, A., Vanin, S., Raga, R., Lavagnolo, M.C., Barausse, A., Rieple, A., Laurent, A., Cossu, R., 2016. Use of digestate from a decentralized on-farm biogas plant as fertilizer in soils: An ecotoxicological study for future indicators in risk and life cycle assessment. Waste Management 49, 378-389.

Ponge, J.-F., Pérès, G., Guernion, M., Ruiz-Camacho, N., Cortet, J., Pernin, C., Villenave, C., Chaussod, R., Martin-Laurent, F., Bispo, A., Cluzeau, D., 2013. The impact of agricultural practices on soil biota: A regional study. Soil Biology and Biochemistry 67, 271-284.

Postma-Blaauw, M.B., de Goede, R.G.M., Bloem, J., Faber, J.H., Brussaard, L., 2010. Soil biota community structure and abundance under agricultural intensification and extensification. Ecology 91, 460-473.

Postma-Blaauw, M.B., de Goede, R.G.M., Bloem, J., Faber, J.H., Brussaard, L., 2012. Agricultural intensification and de-intensification differentially affect taxonomic diversity of predatory mites, earthworms, enchytraeids, nematodes and bacteria. Applied Soil Ecology 57, 39-49.

Ramos, L., Pastorelli, R., Miralles Mellado, I., Fabiani, A., Bastida López, F., Hernández Fernández, M.T., García Izquierdo, C., Solé Benet, A., 2015. Effects of organic amendments and mulches on soil microbial communities in quarry restoration under semiarid climate. Geophysical Research Abstracts 17, EGU2015-15787-1.

Rodriquez A., Sanders, I.R., 2015. The role of community and population ecology in applying mycorrhizal fungi for improved food security. ISMEJ 9, 1053-1061.

Salvio, C., Lopez, A.N., Manetti, P.L., Clemente, N.L., 2011. Effects of granulated baits on meso and macrofauna in soybean soil system. Journal of Environmental Biology 32, 793-799.

Santos, M.J.G., Ferreira, N.G.C., Soares, A.M.V.M., Loureiro, S., 2010. Toxic effects of molluscicidal baits to the terrestrial isopod Porcellionides pruinosus (Brandt, 1833). Journal of Soils and Sediments 10, 1335-1343.

Sanyal, D., Shrestha, A., 2008. Direct effect of herbicides on plant pathogens and disease development in various cropping systems. Weed Science 56, 155-160.

Schloter, M., Bach, H.J., Metz, S., Sehy, U., Munch, J.C., 2003. Influence of precision farming on the microbial community structure and functions in nitrogen turnover. Agriculture Ecosystems and Environment 98, 295-304.

Schmidt, O., Clements, R.O., Donaldson, G., 2003. Why do cereal-legume intercrops support large earthworm populations? Applied Soil Ecology 22, 181-190.

Schmidt, O., Curry, J.P., 2001. Population dynamics of earthworms (Lumbricidae) and their role in nitrogen turnover in wheat and wheat-clover cropping systems. Pedobiologia 45, 174-187.

Schmidt, O., Curry, J.P., Hackett, R.A., Purvis, G., Clements, R.O., 2001. Earthworm communities in conventional wheat monocropping and low-input wheat-clover intercropping systems. Annals of Applied Biology 138, 377-388.

Schreiner, R.P., Bethlenfalvay, G.J., 1997. Mycorrhizae, biocides, and biocontrol. 3. Effects of three different fungicides on developmental stages of three AM fungi. Biology and Fertility of Soils 24, 18-26.

Siepel, H., 1996. Biodiversity of soil microarthropods: The filtering of species. Biodiversity and Conservation 5, 251-260.

Simon, S., Bouvier, J.-C., Debras, J.-F., Sauphanor, B., 2010. Biodiversity and pest management in orchard systems. A review. Agronmy for Sustainable Development 30, 139-152.

Smith, Sally E; Read, David J., 2002. Mycorrhizal Symbiosis (second edition)

Sreenivasa, M.N., Bagyaraj, D.J. (1989). Use of pesticides for mass production of vesicular mycorrhizal inoculum. Plant and Soil 119, 127-132.

Swift, M.J., Heal, O.W., Anderson, J.M., 1979. Decomposition in Terrestrial Ecosystems. Blackwell Scientific Publications, Oxford.

Sylvia, D.M., Chellemi, D.O., 2001. Interactions among root-inhabiting fungi and their implications for biological control of root pathogens. Advances in Agronomy 73, 1-33.

Thompson, J. P., 1987. Decline of vesicular-arbuscular mycorrhizae in long fallow disorder of field crops and its expression in phosphorus deficiency of sunflower. Austrailan Journal of Agricultural Research 38, 847–867.
Tibbett, M., 2015. Decline in soil biodiversity, In: Stolte, J., Tesfai, M., Oygarden, L. (Eds.), Soil in Europe - Threats, functions and ecosystem services. RECARE, pp. 174-184.
Timper, P., Davis, R., Jagdale, G., Herbert, J., 2012. Resiliency of a nematode community and suppressive service to tillage and nematicide application. Applied Soil Ecology 59, 48-59.
Tommerup, I.C., Abbott, L.K., 1981. Prolonged survival and viability of VA mycorrhizal hyphae after root death. Soil Biology and Biochemistry 13, 431-433.
Treonis, A.M., Ostle, N.J., Stott, A.W., Primrose, R., Grayston, S.J., Ineson, P., 2004. Identification of groups of metabolically-active rhizosphere microorganisms by stable isotope probing of PLFAs. Soil Biology and Biochemistry 36, 533-537.
Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005. Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. Ecology Letters 8, 857-874.
Tsiafouli, M.A., Kallimanis, A.S., Katana, E., Stamou, G.P., Sgardelis, S.P., 2005. Responses of soil microarthropods to experimental short-term manipulations of soil moisture. Applied Soil Ecology 29, 17-26.
Valckx, J., Pina, A.C., Govers, G., Hermy, M., Muys, B., 2011. Food and habitat preferences of the earthworm Lumbricus terrestris L. for cover crops. Pedobiologia 54, Supplement, S139-S144.
van Bruggen, A.H.C., Gamliel, A., Finckh, M.R., 2016. Plant disease management in organic farming systems. Pest Management Science 72, 30-44.
Verbruggen, E. Roling, W.F.M., Gamper, H.A., Kowalchuk, G.A., Verhoef, H.A. and van der Heijden, M.G.A (2010). Positive effects of organic farming on below-ground mutualists: large-scale comparison of mycorrhizal fungal communities in agricultural soils. New Phytologist 186: 968–979
Vukicevich, E., Lowery, T., Bowen, P., Úrbez-Torres, J.R., Hart, M., 2016. Cover crops to increase soil microbial diversity and mitigate decline in perennial agriculture. A review. Agronomy for Sustainable Development 36, 48.
Wardle, D.A., Bardgett, R.D., Klironomos, J.N., Setala, H., van der Putten, W.H., Wall, D.H., 2004. Ecological linkages between aboveground and belowground biota. Science 304, 1629-1633.
Wu, Y.P., Li, Y.F., Zheng, C.Y., Zhang, Y.F., Sun, Z.J., 2013. Organic amendment application influence soil organism abundance in saline alkali soil. European Journal of Soil Biology 54, 32-40.
Yeates, G.W., Bardgett, R.D., Cook, R., Hobbs, P.J., Bowling, P.J., Potter, J.F., 1997. Faunal and microbial diversity in three Welsh grassland soils under conventional and organic management regimes. Journal of Applied Ecology 34, 453-470.

12 Soil-improving cropping systems for landslides

J. Poesen and J. Stolte

12.1 Background

A landslide is defined as the movement of a mass of rock, debris, artificial fill or earth down a slope, under the force of gravity, causing a deterioration or loss of one or more soil functions (Huber *et al.*, 2008). Landslides are usually classified on the basis of their type of movement and the type of material involved like rock or fine/coarse soil.

Five principal types of movements are distinguished according to the geomorphological classification proposed by Cruden and Varnes (1996) and Dikau et al. (1996). (i) *Fall*, i.e., a slope of movement for which the mass in motion travels most of the distance through the air, and includes free fall movement by leaps and bounds and rolling of fragments of material. A fall starts with the detachment of material from a steep slope along a surface in which little or no shear displacement takes place. (ii) *Topple*, i.e., a slope movement that occurs due to forces that cause an over-turning moment about a pivot point below the centre of gravity of the slope. A topple is very similar to a fall in many aspects, but do not involve a complete separation at the base of the failure. (iii) *Lateral spreading*, i.e, a slope movement characterized by the lateral extension of a more rigid mass over a deforming one of softer underlying material in which the controlling basal shear surface is often not well-defined. (iv) *Slide*, i.e., a slope movement by which the material is displaced more or less coherently along a recognisable or less well-defined shear surface or band. Slide could be rotational (the sliding surface is curved) or translational (the sliding surface is more or less straight). In some cases a slide can change into a mudslide or slump-earthflow, especially on steep slopes, in clay or silt formations. (v) *Flows*, i.e., a slope of movement characterized by internal differential movements that are distributed throughout the mass and in which the individual particles travel separately within the mass.

Landslides are dominantly considered as a local soil threat in mountainous regions and on slopes (Stolte et al., 2016). Their major driving force is gravity, but local management and controls can be responsible for triggering/preventing them. Among the most common local factors interacting with landslides are topography and the related relief characteristics; soil and bedrock and their specific mechanical and hydrogeological properties; soil depth; hydrological and hydrogeological conditions; vegetation; and anthropogenic activities. However, the most important triggering factor for landslides remains climate and, in particular, precipitation.

Changes in whether (climate) and land-use are the main drivers for landslides. Increases in vegetation/forest cover reduces landslide activity and soil loss (García-Ruiz & Lana-Renault, 2011), and improves the mechanical characteristics of the soil because of root-cohesion. The

abandonment of the lands in the terraced slopes in the Mediterranean environment of southern Europe has led to an increase in shallow landslide activity. Often terraces are retained by dry-stone walls that, if not well maintained, can lose their drainage function and develop saturated horizons at their back slope that can result in their collapse and the triggering of superficial landslides (Camera et al., 2014).

Areas susceptible/vulnerable to landslides are generally under grassland and forests, and not under arable cropping (which is the focus of the SOILCARE project). As a consequence, there is essential no literature on measures to prevent and/or remediate the effects of landslides in arable cropping. Controlling shallow landslides in arable cropping has been addressed in literature for example through cover crops with extensive root systems (De Baets et al., 2011); however this topic is further discussed in the chapter on soil-improving cropping systems for erosion (Chapter 7). In summary, no summary is provided here further on literature related to soil-improving cropping for landslides.

12.2 Conclusions

Landslides refer to the movement of a mass of earth down a slope, under the force of gravity. Landslides occur in mountainous regions and on slopes, following heavy rains, snow melt, deforestation, undermining slope stability, road construction, and/or earth quakes. The actual movement of soil mass often has dramatic effects on food production, human and biological habitats, and cultural heritages.

In general, landslides-specific SICS relate to measures that enhance the stability of the soil and prevent landslides. However, landslides-specific SICS for cropping areas have not been developed and studied yet. Landslide-specific SICS basically involve one mechanism, i.e., redesign. Landslide-prone land should not be used for arable land, but planted with deep-rooted perennial crops, including trees (forest) and left for nature conservation. Terracing and drainage may also help in specific cases. Forest harvesting and site regeneration also need special management attention in landslide-prone sites.

Most promising landslides-specific SICS aim at reducing the risk of landslides (Table 12.1). Changes in profitability are difficult to assess, as these SICS require investments and/or changes in farming practices, which are most likely associated with a drop in income (high-value crops may have to be replaced by low-value crops, including trees).

Table 12.1. *Qualitative assessment of landslides-specific SICS.*

	Components of cropping systems	Components of landslides-specific SICS	Change in profitability	Changes in soil properties		
				Physical	Chemical	Biological
A	Crop rotations	When possible/needed: +permanent, deep-rooting crops	-	+/-	+/-	+/-
B	•Nutrient management	optimal				
C	•Irrigation management	optimal				
D	•Drainage management	Controlled drainage, to increase stability of the soil	-/+	+	+/-	+/-
E	•Tillage management	optimal				
F	•Pest management	Optimal				
G	•Weed management	optimal				
H	•Residue management	optimal				
J	•Mechanization management	Minimal traffic	-	+/-	+/-	+/-
K	•Landscape management	Afforestation, give area back to nature	--	+/-	+/-	+/-

12.3 References

Camera, C., Apuani, T., Masetti, M., 2014. Mechanisms of failure on terraced slopes: the Valtellina case (northern Italy). Landslides, 11, 43-54.

Cruden DM, Varnes DJ. 1996. Landslide Types and Processes. In Landslides: Investigation and Mitigation,Turner AK, Schuster RL (eds.). National Academy Press: Washington DC; 36-75.

De Baets S, J. Poesen, J. Meersmans, L. Serlet 2011. Cover crops and their erosion-reducing effects during concentrated flow erosion. Catena 85, 237–244.

Dikau R, Brunsden D, Schrott L, Ibsen M. (eds.) 1996. Landslide Recognition: Identification, Movement andCauses. Wiley: Chichester.

García-Ruiz, J.M., Lana-Renault, N., 2011. Hydrological and erosive consequences of farmland abandonment in Europe, with special reference to the Mediterranean region – A review. Agriculture, Ecosystems and Environment, 140, 317- 338.

13 Soil-improving cropping systems for water logging

G. Wyseure and H. van Helden

13.1 Background

Water logging is a soil condition whereby during a substantial period in the year either water is ponded at the soil surface and/or the root zone has an excess of water. As a consequence the aeration of the root zone for the cropping system is hampered leading to anoxic conditions for the roots. With the exception of crops like rice or a vegetation like reed in swamps the normal crops require a soil with at least 10 to 15 % volume of air-filled and connected pores. Only in aerobic conditions the roots can function well and take up water and nutrients.

Water logging may be caused by a shallow groundwater table and/or by a soil surface with a limited infiltration capacity, in combination with periods of rainfall exceeding evapotranspiration or rivers overflows. Heavy rainfall or prolonged wet periods leading too high levels in rivers and creating overland flow from fields often lead to flooding. The effect of waterlogging is influenced by the temperature; the microbial life consumes more oxygen if temperature is high and thereby the depletion of oxygen is faster. Flooding during a cold winter therefore is less harmful as compared to during a hot summer.

Water logging results in changes in soil chemistry and microbial activity, including (i) lowered redox potential affect different chemicals. Pezeshki (2001) discusses the different responses in wetlands to soil flooding. Some plant nutrients become less available while other compounds in the soil may become phytotoxic. (ii) denitrification with a loss of nitrate, which leads to a loss of plant-available nitrogen in the soil, and (iii) change in pH.

Crops have different tolerance against flooding which is temperature dependent. Some crops tolerate a short (or long) period of flooding and will recover with some lower production as a consequence, while others are very sensitive and die (Bailey-Serres and Colmer, 2014 and Pucciariello et al., 2014). Water logging leads in general to stress and reduced crop development and growth (as long as the oxygen concentration and supply to the root system is below a critical oxygen level, depending on development stage and duration of water logging). Severe water logging leads leaf yellowing, wilting, senescence, root and tuber rotting.

In addition to direct impact on the root activity, water logging also influences e.g. the soil microbiology, lowers the resistance against compaction and makes grazing and workability more difficult. Harvesting with heavy machinery during waterlogged conditions is detrimental to the soil structure.

13.2 Purpose

The literature review aims firstly at identifying the different aspects/causes of water logging and secondly how cropping systems can be potentially used in different waterlogging situation towards improving soils.

13.3 Results

The key-words "waterlogged cropping systems" gave 17600 results in Google Scholar. A lot of the results concerned tropical cropping systems. Adding "tillage", "drainage", "soil quality" gave similar but somewhat less results. Subsequently more specific literature search was done to the effects of water logging and to the possible actions which could be taken. Below a summary is provided of the main literature results.

13.3.1 Concept

Soil-improving cropping systems (SICS) for water logging aim firstly at preventing water logging and secondly aim at minimizing the effects of water logging on crop growth and soil quality. Hence, SICS and including water basin (catchment) management should try to avoid water-logging, especially on field with agricultural crops. This means that sources of water from outside the cropped field are controlled in such way that the cropped fields are not flooded.

If the water table is too high this can lead to anaerobic conditions in the root zone; In such case better drainage of the land could be considered. If water logging, either by better drainage or water management, cannot be avoided cropping systems should be selected which are adapted to water logged soils and avoid crops sensitive to water excess. Also harvesting by heavy machinery under very wet conditions should be avoided.

13.3.2 Management practices to prevent and mitigate waterlogging

Structural field management practices which prevent water logging and mitigate the effects of water logging are either surface drainage (by mole drainage, raised bed) or subsurface drainage (by tile or ditches) (Abid and Lal, 2009). Drainage reduces or minimizes the detrimental effects of waterlogging. The surface drainage tries to create a drier root zone by evacuating the surface water, which fails to infiltrate, while the subsurface methods aim at lowering the groundwater table, so that capillary rise will not reach the root zone in such a way that sufficient airspace in the root zone allows aerobic conditions. As mole drainage collects water from the soil surface via cracks in the soil and does not aim at lowering the water table it can be seen as a surface rather than a subsurface drainage.

Soil structure improving measures leading to an improved infiltration capacity are beneficial, as these measures minimize the risk of water logging. Avoiding compaction and increasing the organic matter content are elementary (Chapters 8 and 10). Cropping systems that improve the infiltration capacity will have a positive impact (Chapter 16). Cropping systems which require harvesting by heavy machinery under possible very wet conditions late in the growing season should be avoided on soils with sensitive structure, like loamy and clayey soils; on sandy

soils the damage will not be as severe. Harvesting operations under very wet conditions run the risk of destroying the soil structure by compaction and should be avoided therefore (Chapter 8).

Flooding by runoff water from surrounding areas should be avoided. This is not always easy in sloping areas and in case of heavy rains, but discharge rates of rivers and canals should be high enough to circumvent ponding as much as possible. In some case the field can be part of an area earmarked as "*space for water*", and then a temporarily flooding has to be accepted. Specific attention could be given to the management of flood prone fields. Often the pressured from nature conservation claim such areas for pure nature reserve purposes but in some cases, especially for floods risk with return periods of more than 5 years, an adapted agricultural management could also be feasible.

Water logging caused by shallow water tables close to the rootzone should be avoided as well. Hereby water table management by regulating the water level in the ditches and rivers is crucial. Artificial tile drainage is also an option. Partly because of the relatively high risk of waterlogging, some agricultural areas are in pasture. As pastures are not part of the Soilcare project we will not consider such cropping systems, but changing land use to pastures is indeed an effective strategy to cope with a high risk of water logging. Avoiding water logging by good drainage conditions has in general a good effect on soil quality and reduces the leaching of fertilizers (Wesström et al., 2001). On sandy soils it is more easy to regulate the water table and the effects on the hydraulic conductivity are not as drastic. In soils with a high conductivity a relatively simple management of the water level in ditches and rivers is often sufficient for a water table control . On the other hand soils can also have a such a low intrinsic permeability (like heavy clay soils) that even tile drainage is not feasible.

Crops have specific responses and adaptation strategies to stress by water logging (Jackson, 2005). Hereto, special efforts can be made by breeders (e.g. Mendiondo et al., 2016). Of course, breeding specific crops is far beyond the scope of the SOILCARE project.

Selecting cropping systems, which improve the soil structure and the infiltration capacity and hydraulic conductivity of the soil is also a possible strategy. Such cropping systems may include perennials (grasses), cereals like wheat and barley, and alfalfa. However, not all of these crops are equally resistant to temporary waterlogging.

13.4 Conclusions

Flooding is defined as the inundation of land. Water logging is where the soil becomes water-saturated, often due to flooding. Flooding may occur in delta's, plains, and valleys. It may affect humans, flora and fauna, crop yield and quality, infrastructure, cultural heritage, and a range of soil functions.

Flooding-specific SICS aim at (i) preventing flooding and water logging, and (ii) coping with flooded conditions and water logging. Flooding-specific SICS mainly involve changes in input-

output ratios and redesign mechanisms. The first relate to flood prevention and increased discharge/drainage at regional scale. This is the most important measure. Redesign involves growing crops on ridges and growing crops that are less sensitive to temporary flooding. At the landscape scale water storage buffer zones may be created, and/or excess water may be redirected. Evidently, the latter is beyond the scope of an individual farm, and also not a SICS in sensu stricto. On the other hand, some water logging may be needed in delta's to lower the risk of soil subsidence. This is particular the case in organic soils and in recently reclaimed clay soils in polders.

Most promising flooding-specific SICS (i) reduce the risk of flooding, and (ii) reduce the impacts of flooding (Table 13.1). Greatest effects can be expected from drainage management. However, lowering ground water level and creating water buffering basins may not be possible at farm level; it may have to be done at regional level. When flooding can be prevented, the benefits on crop yield will be large. Flooding will also reduce nutrient losses.

Table 13.1. *Qualitative assessment of flooding-specific SICS.*

	Components of cropping systems	Components of flooding-specific SICS	Change in profitability	Changes in soil properties		
				Physical	Chemical	Biological
A	Crop rotations	When possible/needed: +flood-tolerant crops	-/+	+/-	+/-	+/-
B	•Nutrient management	optimal				
C	•Irrigation management	optimal				
D	•Drainage management	Lower groundwater level; create buffer capacity	++	+	+/-	+/-
E	•Tillage management	Ridging, to enhance aerobicity	+/-	+/-	+/-	+/-
F	•Pest management	Optimal				
G	•Weed management	optimal				
H	•Residue management	optimal				
J	•Mechanization management	Optimal				
K	•Landscape management	Creation of water buffer zones	-/+	+/-	+/-	+/-

13.5 References

Abid, M., Lal, R., 2009. Tillage and drainage impact on soil quality: II. Tensile strength of aggregates, moisture retention and water infiltration. Soil and Tillage Research 103, 364–372. doi:10.1016/j.still.2008.11.004

Bailey-Serres, J., Colmer, T.D., 2014. Plant tolerance of flooding stress - recent advances: Plant tolerance of flooding stress. Plant, Cell & Environment n/a-n/a. doi:10.1111/pce.12420

Jackson, M.B., 2005. Response and Adaptation by Plants to Flooding Stress. Annals of Botany 96, 501–505. doi:10.1093/aob/mci205

Mendiondo, G.M., Gibbs, D.J., Szurman-Zubrzycka, M., Korn, A., Marquez, J., Szarejko, I., Maluszynski, M., King, J., Axcell, B., Smart, K., Corbineau, F., Holdsworth, M.J., 2016. Enhanced waterlogging tolerance in barley by manipulation of expression of the N-end rule pathway E3 ligase PROTEOLYSIS6. Plant Biotechnology Journal 14, 40–50. doi:10.1111/pbi.12334

Pezeshki, S.R., 2001. Wetland plant responses to soil flooding. Environmental and Experimental Botany 46, 299–312.

Pucciariello, C., Voesenek, L.A.C.J., Perata, P., Sasidharan, R., 2014. Plant responses to flooding. Frontiers in Plant Science 5. doi:10.3389/fpls.2014.00226

Shaw, R.E., Meyer, W.S., McNeill, A., Tyerman, S.D., 2013. Waterlogging in Australian agricultural landscapes: a review of plant responses and crop models. Crop and Pasture Science 64, 549. doi:10.1071/CP13080

Wesström, I., Messing, I., Linner, H., Lindström, J., 2001. Controlled drainage—effects on drain outflow and water quality. Agricultural Water Management 47, 85–100.

14 Soil-improving cropping systems for desertification

A.J.D. Ferreira, J. Cuevas González, G. Wyseure, O. Vizitiu, V. Pinillo, Y. Canton, C. Stoate, A.K. Boulet and C.S.S. Ferreira

14.1 Background

Desertification has been defined as "land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities" (United Nations General Assembly, 1994). Generally, desertification is the result of (i) soil erosion, (ii) loss of soil fertility, and (iii) long-term loss of natural or desirable vegetation. In arid areas, there can be also involvement of wind erosion and salinisation.

Land degradation in dryland areas results in the reduction or loss of biological or economic productivity of rain-fed croplands, or range, pasture, forest and woodlands. Driving factors for desertification and land degradation are human induced activities, including over grazing (Ferrol et al., 2004; Davis, 2005; Ibáñez et al., 2007; D'Odorico et al., 2013), salinization (Amezketa, 2006; Myyazono et al., 2015; D'Odorico et al., 2013), urbanization (Barbero-Sierra et al., 2013; Ferreira et al., 2017), deforestation and vegetation harvesting (Ferrol et al., 2004), and climate change. Figure 14.1 provides a map of the desertification prone areas in Europe.

Desertification prone areas in Europe include the south of the Alentejo Region in Portugal (Costa and Soares, 2012), the Apulia Region (Ladisa et al., 2012) and southern Italy (Salvati, 2014), Murcia (Hooke and Sandercock, 2012), Almeria and the south of Spain (Martínez-Valderrama et al., 2016). Furthermore, some of the Canarias Islands, particularly the leeward areas of the islands (especially Fuerteventura and Lanzarote) have a high level of aridity. In addition, the Porto Santo Island in the Madeira Archipelago also possesses a semi-arid climate where the risk of desertification is high. In addition, Ibáñez et al. (2007) reported desertification resulting from land degradation in extensive livestock-farming systems, such as (i) cattle or sheep farmed at dehesas systems in south-western Spain, (ii) goat farming on pastures in south-eastern Spain, and (iii) pig farming at montado dehesas systems in southern Portugal.

Bakr et al. (2012) identified three agricultural land use systems in dryland areas: irrigated cropland, rain-feed cropland and rangeland grazing. In practice, multiple crop systems can be found, both in rain-fed and irrigated agriculture areas, combining tree orchards and grain or vegetable crops. Some of the desertification prone areas (e.g. the El Ejido region near Almeria in Spain) are used to produce cash crops, namely vegetables and fruits, mainly in greenhouses. These crops are highly demanding in water, and the current irrigation practices result in groundwater abstraction or the transfer of water from other river catchments. It has been estimated that about half of the irrigation water is at the expense of aquifer depletion that are not recoverable in less than half a century (Custodio et al., 2016). The most intensively exploited

aquifers are some of the coastal ones located close to the large irrigation, urban, and tourist areas, those of the highlands of Murcia (Altiplano Murciano) and the High and Mid Vinalopó basin, further to the Campo de Dalías. In these areas, the groundwater use, based on old water rights, exceed recharge so most aquifers tend to be depleted, except those in the Segura river headwaters area (Custodio et al., 2016).

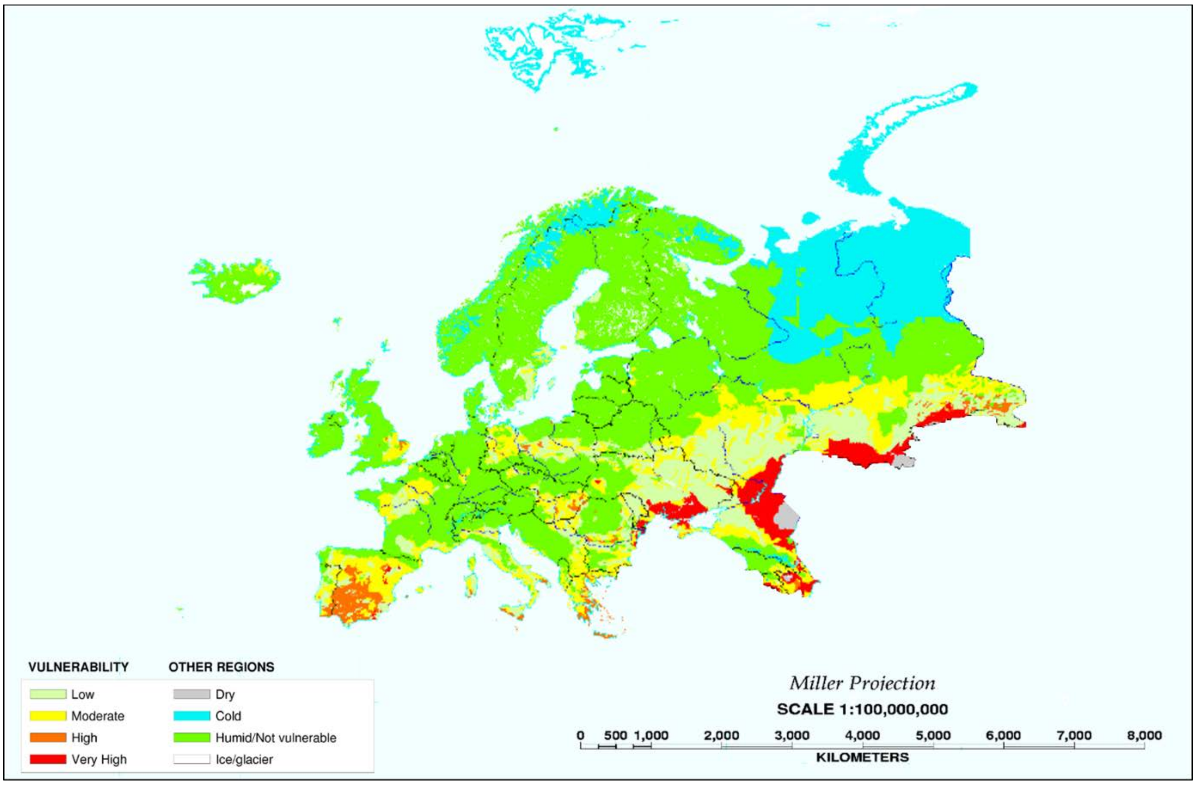


Figure 14.1. *Vulnerability to desertification in Europe (adapted from: The Desertification Vulnerability map, USDA-NRCS, Soil Science Division, World Soil Resources, 1998).*

The water scarcity problem is solved in part by the transfer of water from other basins with higher hydrological availability, or using unconventional water resources, such as treated and untreated wastewater (Pedrero et al., 2010; Becerra-Castro et al., 2015; Bedbabis et al., 2015; Garcia-Orenes et al., 2015) and desalinated sea water (Díaz et al., 2013). The use of treated and untreated wastewater is not without risks to human health (Weber et al., 2006), and has therefore to be carefully planned and managed (Bichai et al., 2012).

Intensive land uses and poor management, together with increasing occurrence of extreme climate events lead to increased incidences of massive forest fires. This is particularly importante in the European sub-humid Mediterranean regions, which despite the high annual

rainfall amounts (exceeding in many cases 1000 mm yr^{-1}), result in high erosion rates and desertification (Ferreira et al., 2015).

14.2 Purpose

Review of literature about how soil-improving cropping systems contribute to combat desertification and improve soil quality.

14.3 Results and Discussion

14.3.1 Concept and drivers of desertification

Drylands harbour the most desertification landscapes in Europe, with the Mediterranean region as the most vulnerable (World Atlas of Desertification (WAD), 2015). Drought, wind and water erosion, loss of organic material, soil crusting, salinization, and other processes gradually render infertile soils that may be lost to desertification. Desertification can be seen as a loss of sustainability and resources (Puigdefabregas, 1995).

The Spanish Plan of Action to Combat Desertification (Magrama, 2008, 2014) takes into account the definition given by UNCCD, which considers land degradation resulting from both natural (e.g. such as climate variations) and human (e.g. human activities) factors. The main factors causing desertification are: aridity, drought, erosion, forest fires and land degradation linked to unsustainable use of water resources.

Soil erosion can be considered as one of the main causes (López-Bermúdez and García-Ruiz, 2008). It constitutes a major problem in the farming agriculture of Mediterranean areas, where intense storms fall over bare soils (after long and dry summers), often located on steep slopes, and soils subject to intensive tillage, or the occurrence of large uncultivated patches like in orchards. Deforestation for cropping has strongly increased soil erosion, causing gully and landslides with the subsequent increase of runoff, enhancing catastrophic floods, sediment load in rivers, reservoir sedimentation and degradation of water quality (López-Bermúdez and García-Ruiz, 2008; Cerdà, 2008). All these effects are interrelated and trigger the desertification processes. It is very well known that agricultural systems without conservation measures enhance soil erosion (Soto et al., 1995; Lasanta et al., 2006a,b), which in turn decrease the soil capacity of production up to irreversible or almost irreversible situations, resulting in shallow stony soils which once abandoned rarely reach the previous climax vegetation (Lasanta et al., 2006 a,b).

The expansion of irrigation fields in Spain has contributed to increased desertification. Initially these cropping systems were very profitable, as a result of the low cost of water and the high demand for cash crops, which triggered the expansion of the irrigated area. This in turn led to aquifer overexploitation, sea water intrusion in coastal regions, salinization and soil degradation, as well as river flow reductions and loss of wetlands. All these processes are closely linked and are drivers of desertification, especially water resources overexploitation and

soil salinization. Wade et al. (2012) ranked Spain among the countries with the highest rates of groundwater depletion.

14.3.2 Measures to combat desertification

Portnov and Safriel (2004) identify several approaches and practices to combat desertification, namely (i) the control of shrubland grazing; (ii) development and implementation of afforestation programs; (iii) water transfer and importation and (iv) increase irrigation efficiency in agriculture. Davis (2005) made a plea for the use of indigenous knowledge to combate desertification, while Rossi et al. (2015) present a novel technological approach to stabilize CO_2 in water bodies and drylands using cyanobacteria, green algae and some autotrophic bacteria. Other authors propose the establishment of vegetation and plant cover crops at specific strategic locations at sub-catchment scale, in order to enhance runoff and sediment sinks (Hooke and Sandercock, 2012).

Reduced/ No tillage

Conservation agriculture tecnhiques have been applied to mitigate desertification factors. They consist in management practices that mitigate modifications in soil composition, structure and biodiversity, thus reducing soil erosion and degradation. These techniques include zero tillage (direct seeding), minimum tillage and maintaining soil surface cover (e.g. with vegetation).

In general, the reduction of tillage operations have been demonstrated to improve soil water content or water available for plants (Fernández-Ugalde et al., 2009; Bescansa et al., 2006; Lampurlanés et al., 2001). The decrease in runoff and sediment losses with reducing tillage intensity also reduced soil organic matter losses (Almagro et al., 2013; Gomez et al., 2009). Nevertheless, a considerable variability is found in the reported rates and losses (Table 14.1) as a consequence of the differences in soil type, slopes, plots sizes etc.

Garcia-Ruiz (2010) reported a three-fold reduction of soil and water losses under reduced tillage in areas characterised by high intensity-low frequent rainfalls, responsible for most soil erosion. Moreover, reduced tillage can be an effective soil management practice for wind erosion prevention, especially in areas prone to wind erosion, such as semi-arid drylands of Central Aragón in NE Spain. López et al. (1998; 2000) observed less soil erosion by wind after reduced tillage than after conventional tillage. Conservation tillage practices have been demonstrated to reduce soil erosion and increase soil water content which may lead to higher crop yields (Fernandez-Ugalde et al., 2009). Reduced tillage contributed also to a higher SOC content than conventional tillage (Almagro et al., 2013).

Table 14.1. *Effects of soil conservation operations on erosion, soil water content (SWC) and soil organic carbon (SOC) content; overview of literature results.*

Cropping system	Rainfall (mm)	Region	Soils	Crop	Erosion	SWC	SOC	Reference
Reduced tillage	330	SE Spain	Silt loam Calcisol	Almond orchards	-25.6%	+15%	+48%	Almagro et al., 2013
Reduced tillage	484	SW Spain	Sandy clay loam Entisol	Crop rotation	-	-	+3%	Lopez-Garrido et al., 2014
Reduced tillage + green manure	330	SE Spain	Silt loam Calcisol	Almond orchards	-55.6%	+15%	+48%	Almagro et al., 2013
Reduced tillage + green manure	370	SE Spain	Loam Petro-calcic Calcisol	Almond orchards	-	-	+14% (compared with reduced tillage)	Franco et al., 2016
Reduced tillage + green manure	370	SE Spain	Silt loam Calcisol	Almond orchards	-	+11% + 17% (compared with reduced tillage)	+26% (compared with reduced tillage)	Almagro et al., 2013
No tillage	370	SE Spain	Silt loam Calcisol	Almond orchards	-	+11% + 17% (compared with reduced tillage)	-	
No tillage	460	N Spain	Loamy Clayey Sandy Xerothent	Vinyards	-76.5% (-60% runoff)	-	-	Lasanta & Sobron 1998

SoilCare
SOILCARE FOR PROFITABLE AND SUSTAINABLE CROP PRODUCTION IN EUROPE

Cropping system	Rainfall (mm)	Region	Soils	Crop	Erosion	SWC	SOC	Reference
No tillage	484	SW Spain	Sandy clay loam Entisol	Crop rotation	-.	-	+20%	Lopez-Garrido et al., 2014
No tillage	655	Central Spain	Silty Clay Vertisol	Olive trees	+238%	-	-	Gomez et al., 2009
No tillage	595	Central Spain	Loamy Xerofluvent	Cotton, maize irrigated with fallow	-88.6%	-	-	Boulal et al., 2011
No tillage	565	S Spain	Loamy Xerorthent	Olive trees	+449%	-	-	Francia et al., 2006 (Gomez et al., 2003 and Martinez et al., 2002)
No tillage herbicide	479	E Spain	Loamy	Orchards	+32.5%	-	-	Garcia-Orenes et al., 2009
No tillage weeds chopped	479	E Spain	Loamy	Orchards	-94.2%			
No tillage weeds chopped + straw mulch	479	E Spain	Loamy	Orchards	-100%		+3.2%	
No tillage weeds chopped + pruned branches	479	E Spain	Loamy	Orchards	-98.9%			
Cover crops	386	Central Spain	Sandy clay loam Luviosol Calcic	Vineyards covered by Brachypodium distachyon	-86.7		SOC loss -66.6%	Ruiz-Colmenero et al., 2013
Cover crops	386	Central Spain	Sandy clay loam Luviosol Calcic	Vineyards covered by Secale cereal spring-mown	-78.4%		SOC loss -66.6%	

Cropping system	Rainfall (mm)	Region	Soils	Crop	Erosion	SWC	SOC	Reference
Cover crops	655	Central Spain	Silty Clay Vertisol	Olive trees with barley cover	-70%			Gomez et al., 2004
Cover crops	565	S Spain	Loamy Xerorthent	Olive trees with barley cover	-76.9%			Francia et al., 2006
Cover crops	534	SW Spain	Sandy loam Petrocalcic	Olive trees	-97.4% (OC losses)			Gomez et al., 2009
Cover crops	481	S Spain	Loamy Hypercalcic calcisoil	Almond orchards oats & oat vetch cover		-46.2%	+66.7%	Ramos et al., 2010
Regulated Deficit irrigation and reclaimed water	300	SE Spain	Loam Soil	Irrigated mandarine Orchard Drip irrigation	Water savings of 15%, salt concentration at the root zone, no significant reduction of yields under reclaimed water use			Mounzer et al., 2013
Regulated Deficit irrigation	429	E Spain	Clay loam soils	Irrigated Citrus Clemen-tina Orchard Drip irrigation	Water savings of 15%-19%. Irrigation strategies may lower fruit size and therefore economic income			Ballester et al., 2014
Regulated Deficit irrigation	250	S Spain	Sandy clay loam	Vineyards Drip irrigarion	Water savings of 14%-22%. No negative impact on yield			Pinillos et al., 2016

No tillage is frequently preferred by many farmers, because it saves labour and fuel. Although no tillage increases organic carbon and nitrogen accumulation in soil, comparing to conventional tillage, it may worsen some soil physical conditions. After 5 years of no tillage, López-Garrido et al. (2014) observed in a xero-fluvent soil cropped with a wheat-sunflower-fodder pea crop rotation, a strong increase in soil penetration resistance at the time of seedling emergence, which reduced seed yield of sunflower and seed quality (34% of oil in no tillage versus 48% in traditional tillage and 50% in reduced tillage.

The presence of vegetation cover (either natural vegetation or green manure) prevents the loss of soil water by evaporation and improves infiltration, intercept rainfall and reduces runoff and erosion, the later also through soil stabilization by plants roots (Fernandez- Ugalde et al., 2009; Gomez et al., 2009; Ruiz-Colomero et al., 2013; Lasanta and Sobrón, 1988).

Soil covered by vegetation is less susceptible to soil crusting and thereby reduces runoff and erosion by up to 60% (de Vente, 2008; Garcia-Franco et al., 2015; López-Garrido et al 2014). Plant residues from green manure and their incorporation in soils by reduced tillage promote the formation of new aggregates and activate the subsequent physico-chemical protection of SOC through the formation of organo-mineral complexes (Garcia-Franco et al., 2015). Green manure and reduced tillage have demonstrated their positive effects on soil structure, and soil carbon sequestration in cereal crops and almonds and olives yards in Spain (Table 14.1).

Cover crops

Among soil conservation practices, cover crops are being adopted mainly by olive growers, as a promising method to reduce soil and water losses. Runoff and soil losses decreased 22% and 76%, respectively, after two years of cover crop implementation (Espejo-Pérez et al., 2013). Water consumption from cover crops, however, is a major concern to farmers, thus, it is not a common practice in semiarid agroecosystems, because of the competition for water resources between green manure/cover crop and the main crop (Martinez-Mena et al., 2013). Nevertheless, it is expected that runoff decrease compensates additional water consumption provided by cover crops, thus favouring the water balance in agriculture fields. However, the relationship between a greater soil water content and crop production is not always clear (Almagro et al., 2013); it depends also on the improvement that these practices generate in plant nutrients availability or reducing soil penetration resistance and thus improving root growth (Ferreras, 2000; Martinez-Mena et al., 2013).

Terracing

Terraces are also very frequent on hillslopes and mountainous regions, although they have been used extensively across diverse landscapes such as areas subject to severe drought, water erosion, mass movement and landslides from steep slopes. These problems threaten the security of land productivity, the quality of local environment and human infrastructures (Lasanta et al., 2001; Wei et al., 2016).

Terracing can increase crop yield and help to mitigate famine, particularly when water scarcity and soil erosion become the main concerns in many mountainous regions (Rockström and Falkenmark, 2015). Terracing can mitigate drought by facilitating soil moisture conservation and accumulating nutrients for crops, thus increasing their production potential. Also, a more favourable interaction between water and fertilizer also can occur with terracing, since soil water retention increases under terracing (Liu et al., 2011).

Numerous examples in Mallorca Island, inner Catalonia and Aragón of well-preserved traditional bench terraces over centuries, demonstrate their efficiently to reduce soil erosion (Garcia-Ruiz, 2010). In the south of Spain, many terraces cultivated with olive and almond trees have recently been increased in size to allow the possible use of machinery or the installation of infrastructures, such as drainage systems. In Portugal terracing was widespread throughout the mountain regions during the periods where soil for cultivation was scarce. Since then they hav been gradually abandoned, but remain the main structural factor holding back erosion and desertification processes.

For example, in Carcavo basin, Spain, the density of terraces decreased by 27%, enlarging the width of the fields from 10–30 m to 50–140 m. As a consequence, 50% of terraces exceeded the critical soil erosion threshold in 2005, compared to 0% in 1956 (Bellin et al., 2009). Ramos and Martinez-Casasnovas (2009) found that the recent vineyards terraces enlargement led to instability as a consequence of the increase in the height between terraces and often the upper soil levels were dug out, mixing the fertile topsoil with the subsoil (Ramos and Martínez-Casasnovas, 2006), which promoted a rapid sealing and the consequent reduction of infiltration, increase in runoff and in soil losses (Ramos et al., 2000). As consequence a decrease up to 50% in grape production in the new terraces compared to small traditional ones was found (Ramos and Martínez-Casasnovas, 2006).

Regulated irrigation

A common perception is that increasing water productivity of irrigated agriculture in arid areas is among the most efficient water policies to fight desertification and rural exodus. Water productivity can be increased through different strategies of water, based on alternatives forms of irrigation and management practices improving water use and re-use efficiency. Regulated deficit irrigation (RDI), partial root drying irrigation, conveyance efficiency improvement are different ways to improve water productivity. Besides that, irrigation with alternative water sources, such wastewater or desalinated water, can reduce the problem of water scarcity and groundwater depletion in semiarid zones.

Regulated deficit irrigation (RDI) is an irrigation strategy that puts crops deliberately under a certain degree of water stress during 'drought-tolerant' growth stages, while ample water is applied during 'drought-sensitive' stages. Besides saving water, RDI allows to save energy and fertilizers and obtain optimal water productivity. RDI strategy have the potential to conserve more water with less impact on yields than any other alternatives (Fereres and Soriano, 2007;

Geerts and Raes, 2009; Rodrígues and Pereira, 2009). RDI has been applied satisfactorily in annual crops, fruit and vine crops (Evan and King, 2012).

Panagopoulos et al. (2014) compared different irrigation practices - RDI, conveyance efficiency improvement and wastewater reuse, to improve water productivity or increase water savings, in a prone desertification area of Greece. The study concluded that a number of different BMPs (best management practices) at an affordable implementation cost, along with a tiered water pricing system that could address socioeconomic heterogeneities, would form a sustainable action plan against desertification in the highly water-deficient Pinios basin. In this study, a process-based model for management simulation in large agricultural landscape called SWAT has been used.

Economic incentives

To promote the implementation of sustainable soil management practices among farmers, it is needed to demonstrate that the practice is economically worthwhile. Galati et al (2015) analysed the difference in net incomes and replacement costs in vineyards managed by distinct soil management practices (conventional tillage and cover crops), and reports that the incentive ranged from 315 € ha^{-1} (loss of income) to 1,088 € ha^{-1} (ecosystem service benefit). They found that the maximum payment does not necessarily correspond to the maximum ecosystem benefit, because the maximum payment may reduce incentive efficiency, and the incentive will differ based on the morphological condition of the vineyard soil. New approaches providing a global view. For example, including the off-site effects of erosion such as pollution, siltation, clogging waterways and flooding of low lands, is needed to ensure that farming systems will provide the needed ecosystems benefits.

14.4 Conclusions

Desertification is the degradation of land in arid and semi-arid areas, as a result of loss of vegetation due to climatic fluctuations and human activities, including over-grazing, fires, soil erosion, salinization, and/or nutrient depletion through withdrawal of harvested crop without return of nutrients. Degraded soils lose their ability to capture and store water, nutrients and carbon, and to support biological processes. Desertification negatively affects food and other biomass production potential, the storage, filtering, buffering and transformation of carbon and nutrients, and the biological habitat and gene pool.

Measures to prevent/mitigate desertification mainly involve mechanisms that change input-output ratios and may involve redesign mechanisms. External inputs of water and nutrients may be needed to enhance the soil fertility and productivity of the soil and thereby to prevent degradation. However, the main mechanism is redesign of the land-use and incorporating suitable landscape elements. For instance, whenever possible C-4 grasses and crops with high water use efficiency (WUE) should be grown, whereas overgrazing must be prevented, as well as long-term animal camping sites (to improve nutrient recycling). Measures to minimize or control runoff are needed to minimize erosion risk and downstream flooding during incidental

rains. Landscape elements such as tree lines and hedges may also contribute to minimizing erosion and land degradation, and to water harvesting.

Most promising desertification-specific SICS aim at reducing (i) the risk of desertification and (ii) the impacts of desertification (Table 14.2). They may have a significant impact on landscape and resource use efficiency.

Table 14.2. *Qualitative assessment of desertification-specific SICS.*

	Components of cropping systems	Components of desertification-specific SICS	Change in profitability	Changes in soil properties		
				Physical	Chemical	Biological
A	Crop rotations	When possible/needed: +permanent vegetation & crops with high WUE	+	+	+/-	+/-
B	•Nutrient management	optimal				
C	•Irrigation management	Targeted (drip) irrigation	+	+/-	+/-	+/-
D	•Drainage management	optimal				
E	•Tillage management	Reduced tillage	+	+/-	+/-	+/-
F	•Pest management	Optimal				
G	•Weed management	optimal				
H	•Residue management	Surface mulching, to reduce evaporation	+/-	+	+/-	+
J	•Mechanization management	Optimal				
K	•Landscape management	Treelines, hedges, agroforestry	+	+/-	+/-	+/-

14.5 References

Almagro M., de Vente J., Boix-Fayos C., García-Franco N., Melgares de Aguilar J., González D., Solé-Benet A., Martínez-Mena M., 2013. Sustainable land management practices as providers of several ecosystem services under rainfed Mediterranean agroecosystems. Mitig. Adapt. Strateg. Global Change 21, 7, 1029–1043.

Almagro M., Martínez-Mena M., 2014. Litter decomposition rates of green manure as affected by soil erosion, transport and deposition processes, and the implications for the soil carbon balance of a rainfed olive grove under a dry Mediterranean climate. Agric. Ecosys. Environ. 196: 167–177.

Amezketa E., 2006 An integrated methodology for assessing soil salinization, a pre-condition for land desertification. Journal of Arid Environments 67, 594–606.

Bakra, N., Weindorf D.C., Bahnassy, M.H., El-Badawi, M.M. 2012 Multi-temporal assessment of land sensitivity to desertification in a fragile agro-ecosystem: Environmental indicators. Ecological Indicators 15, 271–280.

Becerra-Castro, C., Lopes, A.R., Vaz-Moreira, I., Silva, E.F., Manaia, C.M., Nunes, O.C., 2015. Wastewater reuse in irrigation: a microbiological perspective on implications in soil fertility and human and environmental health. Environ. Int. 75, 117–135.

Bedbabis, S., Rouina, B.B., Salvatore C., Clodoveo M.L., 2015, Effect of the seasonal climatic variations on the composition and quality of "Chemlali" virgin olive oil. African Journal of Agricultural Research, 10(49), 4424-4431.

Bellin N., VanWesemael B., Meerkerk A., Vanacker V., Barberá G.G., 2009. Abandonment of soil and water conservation structures in Mediterranean ecosystems. A case study from south east Spain. Catena 76: 114–121.

Bescansa P., Imaz M.J., Virto I., Enrique A., Hoogmoed W.B., 2006. Soil water retention as affected by tillage and residue management in semiarid Spain. Soil Tillage Res 87: 19–27.

Bichai, F., Polo-López, M.I., Ibañez, P.F., 2012 Solar disinfection of wastewater to reduce contamination of lettuce crops by Escherichia coli in reclaimed water irrigation. Water Research, 46, 6040-6050.

Cerdà, A. (Ed.), 2008. Erosión y degradación del suelo agrícola en España. Universidad de Valencia, Valencia. 238 pp.

Costa A.C., Soares A., 2012 Local spatiotemporal dynamics of a simple aridity index in a region susceptible to desertification. Journal of Arid Environments 87, 8-18.

Custodio E., Andreu-Rodes J.M., Aragón R., Estrela T., Ferrer J., García-Aróstegui J.L., Manzano M., Rodríguez-Hernández L., Sahuquillo A., del Villar, A., 2016. Groundwater intensive use and mining in south-eastern peninsular Spain: Hydrogeological, economic and social aspects. Sci. Total Environ. 559: 302–316.

Davis D.K., 2005 Indigenous knowledge and the desertification debate: problematising expert knowledge in North Africa. Geoforum 36, 509–524.

D'Odorico P., Bhattachan A., Davis K.F., Ravi S., Runyan C.W., 2013 Global desertification: Drivers and feedbacks. Advances in Water Resources 51, 326–344.

De Vente, J., 2008. Reduced contour tillage of cereals in semi-arid environments. WOCAT (https://qt.wocat.net/qt_summary.php?lang=english&qt_id=162)

Díaz F.J., Tejedor M., Jiménez C., Grattan S.R., Dorta M., Hernández J.M., 2013 The imprint of desalinated seawater on recycled wastewater: Consequences for irrigation in Lanzarote Island, Spain. Agricultural Water Management 116, 62–72.

Espejo-Pérez A.J., Rodríguez-Lizana A., Ordóñez R., Giráldez J.V., 2013. Soil Loss and Runoff Reduction in Olive-Tree Dry-Farming with Cover Crops. Soil Water Manag Cons 77(6): 2140-2148.

Evans R.G., King B.A., 2012. Site-specific sprinkler irrigation in a water-limited future. Transactions of the ASABE 55 (2): 493-504.

Fereres E., Soriano M.A., 2007. Deficit irrigation for reducing water use. Journal of Experimental Botany 58: 147-159.

Fernández-Ugalde O., Virto I., Bescansa P., Imaz M.J., Enrique A., Karlen D.L., 2009. No-tillage improvement of soil physical quality in calcareous, degradation-prone, semiarid soils. Soil Tillage Res 106: 29–35.

Ferreira A.J.D., Prats-Alegre S., Coelho C.O.A., Shakesby R.A., Páscoa F.M., Ferreira C.S.S., Keizer J.J., Ritsema C., 2015. Strategies to prevent forest fires and techniques to reverse degradation processes in burned areas. Catena 128: 224-237.

Ferreras LA, Costa JL, Garcia FO, Pecorari C (2000) Effect of no-tillage on some soil physical properties of a structural degraded Petrocalcic Paleudoll of the southern "Pampa" of Argentina. Soil Tillage Res 54:31–39

Ferrol N., Calvente R., Cano C., Barea J.-M., 2004. Analysing arbuscular mycorrhizal fungal diversity in shrub-associated resource islands from a desertificationthreatened semiarid Mediterranean ecosystem. Applied Soil Ecology 25, 123–133

Concepción Azcón-Aguilar, Ferreira C.S.S., Walsh R.P.D., Blake W.H., Kikuchi R., Ferreira A.J.D., 2017. Temporal dynamics of sediment sources in an urbanizing Mediterranean catchment. Land Degrad. Dev. in press.

Galati A., Borsellinon V., Crescimanno M., Pisano G., Schimmenti E., 2015 Implementation of green harvesting in the Sicilian wine industry: Effects on the cooperative system. Wine Economics and Policy 4, 45–52.

Garcia-Franco N., Albadalejo J., Almagro M., Martínez-Mena M., 2015. Beneficial effects of reduced tillage and Green manure on soil aggregation and stabilization of organic carbón in a Mediterranean agroecosystem. Soil Tillage Res 153: 66–75.

García-Ruiz J.M., 2010. The effects of land uses on soil erosion in Spain: A review. Catena 81: 1-11.

Geerts S., Raes D., 2009. Deficit irrigation as an on-farm strategy to maximize crop water productivity in dry areas. Agric. Water Manage. 96: 1275-1284.

Gómez J.A., Guzmán M.G., Giráldez J.V., Fereres E., 2009. The influence of cover crops and tillage on water and sediment yield, and on nutrient, and organic matter losses in an olive orchard on a sandy loam soil. Soil Tillage Res 106:137–144.

Grattan et al., 2015

Hookea J., Sandercock P., 2012 Use of vegetation to combat desertification and land degradation: Recommendations and guidelines for spatial strategies in Mediterranean lands. Landscape and Urban Planning 107, 389– 400

Ibáñez J., Martínez J., Schnabel S., 2007 Desertification due to overgrazing in a dynamic commercial livestock–grass–soil system. Ecological Modelling 205, 277–288.

Ladisa G., Todorovic M., Liuzzi G.T., 2012 A GIS-based approach for desertification risk assessment in Apulia region, SE Italy. Physics and Chemistry of the Earth, 49, 103–113.

Lampurlanés J., Angás P., Cantero-Martínez C., 2001. Root growth, soil water content and yield of Barley under different tillage systems on two soils in semiarid conditions. Field Crop Res 69:27–40.

Lasanta T., Sobrón I., 1988. Influencia de las prácticas de laboreo en la evolución hidromorfológica de suelos cultivados con viñedo. Cuadernos de Investigación Geográfica 14: 81–97.

Lasanta T., Arnaez J., Oserin M., Ortigosa L.M., 2001. Marginal lands and erosion in terraced fields in the Mediterranean mountains: a case study in the Camero Viejo (northwestern Iberian System, Spain). Mt. Res. Dev. 21(1): 69–76.

Lasanta T., Beguería S., García-Ruiz J.M., 2006a. Geomorphic and hydrological effects of traditional shifting agriculture in a Mediterranean mountain, Central Spanish Pyrenees. Mt. Res. Dev. 26 (2): 146–152.

Lasanta T., González-Hidalgo J.C., Vicente-Serrano S.M., Sferi, E., 2006b. Using landscape ecology to evaluate an alternative management scenario in abandoned Mediterranean mountain areas. Landscape Urban Plann. 78: 101–114.

Liu X., Zhang S., Zhang X., Ding G., Cruse R.M., 2011 Soil erosion control practices in Northeast China: A mini-review. Soil & Tillage Research 117 44–48.

Lopez M.V., Sabre M., Gracia R., Arrúe J.L., Gomes L., 1998. Tillage effects on soil erosion surface conditions and dust emission by wind erosion in semiarid Aragon (NE Spain). Soil Tillage Res. 72 (1): 53-64.

Lopez M.V., Gracia R., Arrúe J.L., 2000. Effects of reduced tillage on soil Surface properties affecting wind erosion in semiarid fallow lands od central Aragon. Eur. J. Agron. 12 (3-4): 191-199.

López-Bermúdez, F., García-Ruiz, J.M., 2008. La degradación del suelo por erosion hídrica en España. In: Cerdà, A. (Ed.), Erosión y degradación del suelo agrícola en España. Universidad de Valencia, Valencia, pp. 11–49.

Lopez, M.V., Sabre,M., Gracia, R., Arrúe, J.L. and Gomes, L., 1998. Tillage effects on soil erosion surface conditions and dust emission by wind erosion in semiarid Aragon (NE Spain). Soil and Tillage research 72 (1): 53-64

López-Garrido R., Madejón E., León-Camacho M., Girón I., Moreno F.,Murillo J.M., 2014. Reduced tillage as an alternative to no-tillage under Mediterranean conditions: a case study. Soil Till. Res. 140: 40–47.

MAGRAMA, 2014. 3rd Monitoring Report of the National Plan for Adaptation to Climate Change. Oficina Española de Cambio Climático, Madrid 97 http://www.magrama.gob.es/es/cambio-climatico/temas/impactos-vulnerabilidad-y-adaptacion/plan-nacional-adaptacion-cambio-climatico/

MAGRAMA, 2008. Programa de Acción Nacional contra la Desertificación. Madrid. 262 p http://www.magrama.gob.es/es/desarrollo-rural/temas/politica-forestal/desertificacion-restauracion-forestal/lucha-contra-la-desertificacion/lch_pand_descargas.aspx

Ministerio de Medio Ambiente (MMA) (2002-2012). Inventario Nacional de Erosión de Suelos. Available in: http://www.magrama.gob.es/es/desarrollo-rural/temas/politica-forestal/inventario-cartografia/inventario-nacional-erosion-suelos/

Mártinez R.A., Ruiz S., Cárceles B., Durán Z.V.H., Francia J.R., Arroyo P.L., 2002. Comparison of two cultivation techniques in olive plantations with slopes higher than 25%: green plant cover and no tillage. In: Villalobos FJ, Testi L, editors. Proceedings VII ESA congress of the European society for agronomy, Cordoba, Spain; 519–520

Martínez-Mena M., García-Franco N., Almagro M., Ruiz-Navarro A., Albaladejo J., Melgares de Aguilar J., Gonzalez D., Querejeta J.I., 2013. Decreased foliar nitrogen and crop yield in organic rainfed almond trees during transition from reduced tillage to no-tillage in a dryland farming system. Eur J Agron 49:149–157

Martinez-Casasnovas J.A., Ramos M.C., 2009. Erosión por cárcavas en el área de viña del Alt Penedés-Anoia (NE España). Cuadernos de Investigación Geográfica 35 (32): 223-238.

Martínez-Valderrama J., Ibáñez J., Del Barrio G., Sanjuán M.E., Alcalá F.J., Martínez-Vicente S, Ruiz A., Puigdefábregas J., 2016 Present and future of desertification in Spain: Implementation of a surveillance system to prevent land degradation Science of the Total Environment 563–564,169–178

Miyazono S., Patiño R., Taylor C.M., 2015 Desertification, salinization, and biotic homogenization in a dryland river ecosystem. Science of the Total Environment, 511, 444–453

Panagopoulos Y., Makropoulos C., Gkiokas A., Kossida M., Evangelou L., Lourmas G., Michas S., Tsadilas C., Papageorgiou S., Perleros V., Drakopoulou S., Mimikou M., 2014 Assessing the cost-effectiveness of irrigation water management practices in water stressed agricultural catchments: The case of Pinios. Agricultural Water Management 139, 31–42.

Pedrero F., Kalavrouziotis I., Alarcóna J.J., Koukoulakis P., Asano T., 2010 Use of treated municipal wastewater in irrigated agriculture—Review of some practices in Spain and Greece. Agricultural Water Management 97, 1233–1241.

Portnova B.A., Safriel U.N., 2004 Combating desertification in the Negev: dryland agriculture vs. dryland urbanization. Journal of Arid Environments 56, 659–680.

Puigdefabregas J., 1995. Desertification: stress beyond resilience, exploring a unifying process structure. Ambio 24 (5): 311–313.

Ramos M.C., Martínez-Casasnovas J.A., 2006. Impact of land levelling on soil moisture and runoff variability in vineyards under different rainfall distributions in a Mediterranean climate and its influence on crop productivity. J. Hydrol. 321: 131–146.

Ramos M.C., Nacci S., Pla I., 2000. Soil sealing and its influence on erosion rates for some soils in the Mediterranean area. Soil Sci. 165: 396–403.

Rodrigues G.C., Pereira L.S., 2009. Assessing economic impacts of deficit irrigation as related to water productivity and water costs. Biosystems Eng. 103: 536-551.

Rockström J., Falkenmark M., 2015. Agriculture: increase water harvesting in Africa. Nature 519 (7543): 283–285.

Rossi F., Olguín E.J., Diels L., de Philippis R., 2015 Microbial fixation of CO2 in water bodies and in drylands to combat climate change, soil loss and desertification. New Biotechnology, 32, 1, 109-120

Ruiz-Colmenero M., Bienes R., Eldridge D.J., Marques M.J., 2013. Vegetation cover reduces erosion and enhances soil organic carbon in a vineyard in the central Spain. Catena 104:153–160.

Salvati L., 2014 A socioeconomic profile of vulnerable land to desertification in Italy. Science of the Total Environment 466–467, 287–299

Soto B., Basanta R., Pérez R., Díaz-Fierros F., 1995. An experimental study of the influence of traditional slusch-and-burn practices on soil erosion. Catena 24: 13–23.

United Nations General Assembly, 1994 Elaboration of an international convention to combat desertification in those countries experiencing serious drought and/or desertification, particularly in Africa A/RES/49/234 95th plenary meeting 23 December 1994.
Wade A.J., Smith S.J., Black E.C.L., Brayshaw D.J., Holmes P.A.C., El-Bastawesy M., Rambeau C.M.C., Mithen S.J., 2012 A new method for the determination of Holocene palaeohydrology. J. Hydrology 420–421, 1–16.
Webera S., Khanb S., Hollender J., 2006 Human risk assessment of organic contaminants in reclaimed wastewater used for irrigation. Desalination 187, 53–64.
Wei W., Chen D., Wang L., Daryanto S., Chen L., Yu Y., Lu Y., Sun G., Feng T., 2016. Global synthesis of the classifications, distributions, benefits and issues of terracing. Earth Sci. Rev. 159: 388–403.
World Atlas of Desertification (WAD), 2015 Mapping Land Degradation and Sustainable Land Management Opportunities. Join Research Center.

15 Soil-improving cropping systems for soil acidification

P. Cermak, L. Mensik, H. Kusa, E. Kunzova, M. Frac, B. Toth, Z. Toth and J. Lipiec

15.1 Background

Acidification of soils refers to the loss of base cations (e.g. calcium, magnesium, potassium, sodium) in the soil, and the replacement of these base cations by acidic cations, mainly aluminium (and iron) complexes. Soil acidification is a natural degradation process, defined by a decrease of the soil buffering capacity to neutralize acid (acid-neutralizing capacity).

Buffering substances in the soil are a crucial factor determining how much of the acidifying compounds are neutralized over a certain period. Acidifying substances in the atmosphere can have natural sources such as volcanism, however, the most significant ones are those that are due to anthropogenic emissions, mainly the result of fossil fuel combustion (e.g. in power plants, industry and traffic) and due to intensive agricultural activities (emissions of ammonia, NH_3). Emissions of sulphur dioxide (SO_2) and nitrogen oxides (N_{OX}) to the atmosphere increase the natural acidity of rainwater, snow or hail. After deposition to ecosystems, the conversion of NH_4 - to either amino acids or nitrate (NO_3^-) is an acidification process. In many areas, N_{OX} and NH_3 are now identified as the main acidifying agents (eusoils.jrc.ec.europa.eu/SOER2010/).

Acidification affects the natural environment, including soils, waters, flora and fauna. A decline of soil pH is an effect of soil acidification, which can have a negative influence on crop yields. With a decrease in soil pH, the availability of some essential plant nutrients (Ca, Mg,) decreases, and the mobility of certain toxic elements (e.g. heavy metals) and uptake in the food chain increases (Ministry of Agriculture of the Czech Republic, 2015). A low soil pH may lead to a destruction of soil structure, and to a lower quality of the soil organic matter. As a result, soil biology can be damaged by acidification as certain biota are unable to adapt to changes in soil acidity and chemistry.

Figure 15.1 shows the atmospheric deposition of acids (mainly nitrogen and sulphur compounds) to land surfaces in 1980 and 2010. The critical load of 1200 proton (H^+) equivalent ha^{-1} $year^{-1}$ was exceeded in large parts of Europe in 1980. Thirty years later in 2010, the atmospheric deposition of S and N had strongly decreased, through series of emission reduction measures in power plants, industry, traffic and also agriculture (EEA 2010). The regulatory controls initiated from the 1980s have had a significant impact on the emissions of pollutants that cause acidification, mainly as a result of decreased SO_2 emissions. By 2020, it is expected that the risk of ecosystem acidification will only be an issue at some hot spots, in particular at the border area between the Netherlands and Germany (EEA 2010). A number of local and regional studies have shown that the impact of emissions reduction schemes in many parts of the United Kingdom, Germany and Scandinavia is especially evident with acid levels

declining, rapidly in some parts, or are at least stabilising (Ruoho-Airola et al., 1998; Fowler et al., 2007; Kowalik et al., 2007; Carey et al., 2008, EEA 2010).

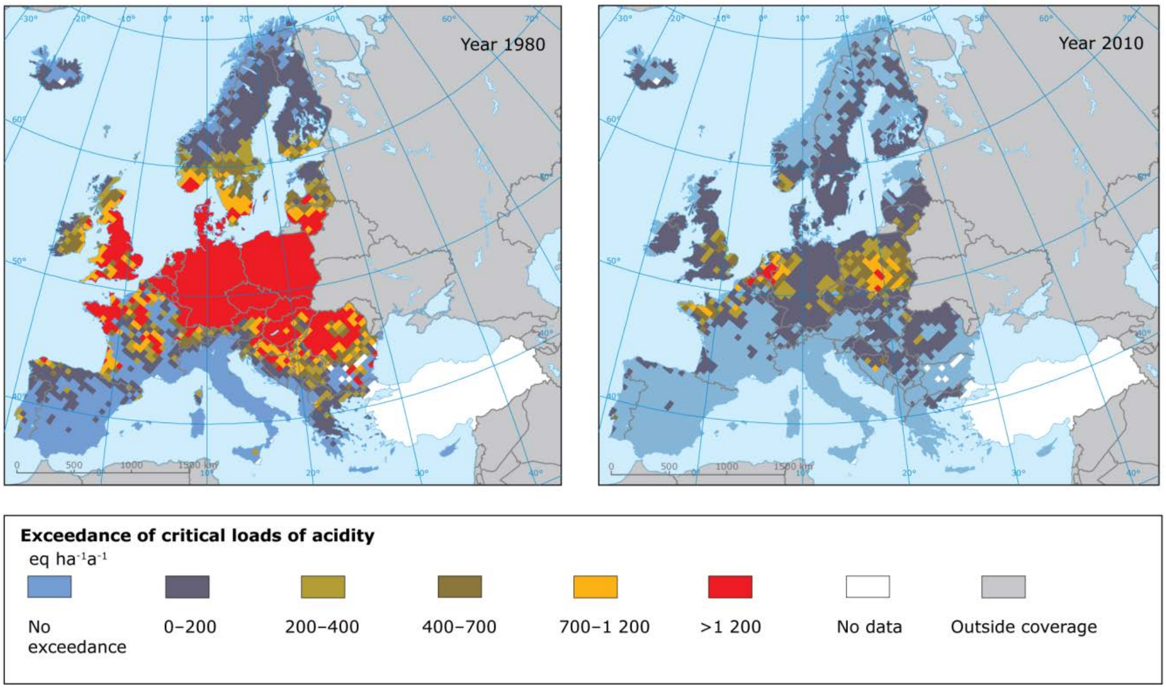


Figure 15.1. *Maps showing changes in the extent to which European ecosystems are exposed to acid deposition (i.e. where the critical load limits for acidification are exceeded. The left figure shows the situation in 1980s; areas with exceedances of critical loads of acidity (i.e. higher than 1200 equivalent ha^{-1} $year^{-1}$ are in red colour) cover large parts of Europe. By 2010, the areas where critical loads are exceeded have shrunk significantly (right-hand figure). These improvements are expected to continue to 2020, although at a reduced rate. Source: Deposition data collected by European Monitoring and Evaluation Programme (EMAP); Maps drawn by Coordination Centre for Effects (CCE); EEA 2010.*

The effects of air pollutants on people and the environment are considerable. Damage to health, changes in global climate, acidification of fresh waters, corrosion of materials, erosion of cultural treasures, losses in biodiversity and in agricultural crop yields, are some of the more obvious effects. Acidification as an environmental problem was first given serious attention in the late 1960s. However, its effects began to appear long before that, and we now know that emissions of acidifying substances cause serious damage to nature, to ourselves and to our built environment (Elvingson, Ågren 2004).

15.2 Purpose

The aim of this review is to collect and assess data and information on soil-improving cropping systems reducing soil acidification.

15.3 Results and Discussion

15.3.1 Concept of soil acidification

Soil acidity occurs naturally in higher rainfall areas and can vary according to the landscape geology, clay mineralogy, soil texture and buffering capacity. Soil acidification is a natural process, accelerated by some agricultural practices.

Most plant material have a positive cation (Σ(Ca, Mg, K, Na)) – anion (Σ(S, P)) balance, depending also on the form of the uptake of N (NO_3^-/NH_4^+/N_2). As a result plants have a slightly alkaline composition, and removal by grazing or harvest leaves residual hydrogen ions in the soil. Over time, as this process is repeated, the soil becomes acidic. Major contributors are lucerne and legume crops (as they fix N_2). Alkalinity removed in animal products is low. However, concentration of dung in stock camps adds to the total alkalinity exported from grazed grassland in animal production systems (http://soilquality.org.au/factsheets/soil-acidity). When plant material is removed from the paddock, alkalinity is also removed. This increases soil acidity.

Acidification may also occur through the use of ammonium-based N fertilizers. These fertilizers are acidic, because the nitrification of NH_4^+ to NO_3^- releases two equivalents of acidity (H^+). Soil acidification is aggravated by nitrate leaching. The build-up of organic matter in soils also contributes to acidification.

15.3.2 Soil-improving cropping systems to combat acidification

To suppress the negative effects of acidification it's necessary to take advantage of specific agronomical tools and measurements (soil improving cropping systems), based mainly on positive results from the long-term field experiments. Effective measures include:

- Application of lime ($CaCO_3$) and other materials (e.g. stone meal) with acid neutralizing capacity;
- Application of organic fertilizers (farmyard manure-FYM, slurry, straw);
- Application of nitrate-based N fertilizers.

Liming of soils can offset the effect of acidification, but in some circumstances it can have undesirable effects on soil biota and flora through the elimination of certain species. However, in post-communist countries of Central and Eastern Europe farmers have decreased the use of basic inputs to agriculture, including limy materials as an effective factor against acidification. In fact, soil acidification is still a significant problem in many Central and Eastern European countries (Sumner, Noble 2003).

Table 15.1 provides an overview of studies that examined the effects of liming and/or manure application on soil pH. Some studies reported on more than one treatment.

Table 15.1. *Tested agro-management techniques on soil acidification.*

Tested SICS	Reference
straw to NPK	Hejcman, M., Kunzová, E., Šrek, P. (2012)
pig slurry and straw to NPK	Hlisnikovský, L., Kunzová, E., Menšík, L. (2016)
straw to NPK	Hlisnikovský L., Kunzová, E., Klír, J., Hejcman, M., 2014.
pig slurry and straw to NPK	Hlisnikovský, L., Kunzová, E., 2014
FYM to NPK	Hlisnikovský, L., Kunzová, E., 2014
FYM to NPK	Hlisnikovský, L., Kunzová, E., 2014
Pig slurry to NPK	Hlisnikovský, L., Kunzová, E., 2014
FYM to NK	Hlisnikovský, L., Kunzová, E., 2014
Cattle slurry and straw to PK	Hlisnikovský, L., Kunzová, E., 2014
FYM to NPK	Hlisnikovský, L., Kunzová, E., 2014
Cattle slurry and straw to NPK	Hlisnikovský, L., Kunzová, E., 2014
FYM to NPK	Hlisnikovský, L., Kunzová, E., 2014
FYM to N2PK	Hlisnikovský, L., Kunzová, E., 2014
FYm to N3PK	Hlisnikovský, L., Kunzová, E., 2014
lime	Klement et al. (2012)
manure	Xun et al., 2008
manure	Zhang, H. M., Wang, B. R., Xu, M. G. and Fan, T. L. 2009
nutrient balance: P addition to N	Liu et al.(2010)
straw addition to NP	Liu et al.(2010)
FYM to NP	Liu et al.(2010)

The effect of the treatments was expressed in a change in soil pH, for all 21 treatments of the long term experiments. Figure 15.2 presents the effect size (delta pH), as the change in pH due to the treatment relative to the reference case: a positive value indicates an increase of the soil pH. Most data are about the effect of organic materials. Data were normalised by log transformed to calculate the standard deviation. Some studies also reported crop yields.

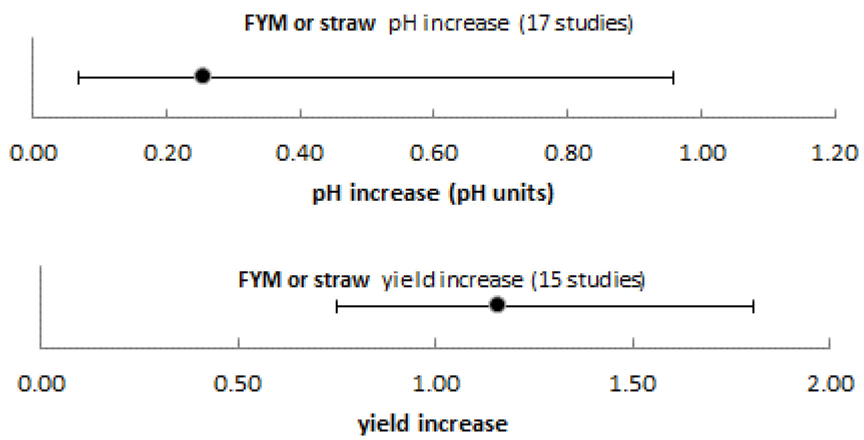


Figure 15.2. *Effect size based on a meta-analysis of all studies reported in Table 15.1 for reported changes in soil pH (n = 17) and reported crop yields, in tonne per ha per year (n = 15).*

15.4 Conclusions

Acidification refers to a decrease of the acid neutralizing capacity of the soil, followed by a drop in pH. Soil acidification is a natural process but accelerated by atmospheric deposition of acidifying elements (mainly nitrogen and sulphur oxides and ammonia in dry and wet deposition), withdrawal of harvest crop, urine droppings by grazing animals, and acidifying (ammonium-based) fertilisers. Soil acidification may lead to distorted root growth, nutrient imbalances, low crop yield and quality, and low biological activity. It increases the risk of uptake of toxic elements in plants. The risk of soil acidification is largest in soils with low acid neutralizing capacity, i.e., sandy soils with low content of base-cations, and in a climate where precipitation exceeds evapotranspiration, i.e. with a rainfall surplus.

Acidification-specific SICS prevent and nullify/remediate the effects of acidification; they involve substitution and redesign mechanisms (Table 15.2). Acidifying nitrogen fertilizers should be replaced by nitrate-based fertilizers. Applications of manures, composts, crop residues also enhance the acid-neutralizing capacity of the soil. Applying acid-neutralizing substances (lime, primary soil minerals) has been practiced successfully already since Roman times, to raise soil pH values to agronomic recommended levels. Redesign mechanisms may involve the growth of crops that tolerate relatively high soil acidity; this may be needed in local areas for example near coal mines where coal wastes have been dumped, and in areas with naturally occurring acid-sulphate (sub)soils.

Table 15.2. *Qualitative assessment of acidification-specific SICS.*

	Components of cropping systems	Components of Acidification-specific SICS[1]	Change in profitability	Changes in soil properties		
				Physical	Chemical	Biological
A	•Crop rotations	When possible/needed: +acid-tolerant crops	+/-	+/-	+/-	+/-
B	•Nutrient management	Liming, manuring; no acidifying N fertilizers	++	+	++	+
C	•Irrigation management	No excess irrigation/leaching	+/-	+/-	+/-	+/-
D	•Drainage management	optimal				
E	•Tillage management	optimal				
F	•Pest management	optimal				
G	•Weed management	optimal				
H	•Residue management	No removal of crop residues	+/-	+	+	+
J	•Mechanization management	optimal				
K	•Landscape management	optimal				

[1]: The term 'optimal' for specific agro-management techniques refers to the need to optimize agro-management techniques in general so as to improve soil quality and functioning (including crop yields); these management techniques do not have soil threat-specific impacts.

Most promising acidification-specific SICS include regular monitoring of soil pH, application of acid-neutralizing substances. In specific cases, replacement of nitrogen fertilizer types and crop types may be needed. These SICS increase resource use efficiency

15.5 References

http://eusoils.jrc.ec.europa.eu/SOER2010/
http://soilquality.org.au/factsheets/soil-acidity

Carey, P.D., Wallis, S.M., Emmett, B.E., Maskell, L.C., Murphy, J., Norton, L.R., Simpson, I.C., Smart, S.S. 2008. Countryside Survey: UK Headline Messages from 2007. Centre for Ecology and Hydrology, Lancaster, the United Kingdom.

EEA; 2010. The European environment - state and outlook 2010: air pollution. European Environment Agency, Copenhagen.

Ellmer, F., H. Pescke, W. Kohn, F.M. Chmielewski, M. Baumecker, 2000. Tillage and fertilizing effects on sandy soils. Review and selected results of long-term experiments at Humboldt-University Berlin. J. Plant. Nutr. Soil Sci. 163, 267-272.

Elvingson, P., Agren, C. 2004. Air and the Environment. The Swedish NGO Secretariat on Acid Rain, Goteborg, Sweden.

Fowler, D., Smith, R., Muller, J., Cape, J.N., Sutton, M., Erisman, J.W., Fagerli, H., 2007. Long term trends in sulphur and nitrogen deposition in Europe and the cause of non-linearities. Water, Air & Soil Pollution: 7, 41–47.

Hejcman, M., Kunzová, E., Šrek, P. (2012). Sustainability of winter wheat production over 50 years of crop rotation and N, P and K fertilizer application on illimerized luvisol in the Czech Republic. Field Crop Research, 139, 30-38.

Hlisnikovský L., Kunzová, E., Klír, J., Hejcman, M., 2014. Vliv hnojení a osevních postupů na výnosy a cukernatost cukrové řepy. Listy cukrovarnické a řepařské 130, 50-56.

Hlisnikovský, L., Kunzová, E., 2014. The content of topsoil nutrients, pH and organic carbon as affected by long-term application of mineral and organic fertilizers. Agriculture (Poľnohospodárstvo) 60 (4), 142-148. DOI: 10.1515/agri-2015-0003,

Hlisnikovský, L., Kunzová, E., Menšík, L. (2016). Winter wheat: results of long-term fertilizer experiment in Prague-Ruzyně over the last 60 years. Plant Soil Environ. 62, 99-104.

Kidd, J., P. Manning, J. Simkin, S. Peacock, E. Stockdale, 2017. Impacts of 120 years of fertilizer addition on a temperate grassland ecosystem PLoS ONE 12(3) e0174632.

Klement et al. (2012): forty years of Long-term Field Experiments in ÚKZÚZ. ISBN978-80-7401-062-0.pp.95

Kowalik, R.A., Cooper, D.M., Evans, C.D., Ormerod, S.J. 2007. Acidic episodes petard the biological recovery of upland British streams from chronic acidification. Global Change Biology 13, 2439–2452.

Kurtinecz, P. (2003) Die dauerdüngungs- und kalkungsversuche aus nord-west Rumänien. Archives of Agronomy and Soil Science, 49:5,503 — 510

Liu. E.K. et al.(2010): Long-term effect of chemical fertilizer, straw, and manure on soil chemical and biological properties in northwest China. Geoderma 158:173–180

Obour, A.K., M.M. Mikha, J.D. Holman, P.W. Stahlman, 2017. Changes in soil surface chemistry after fifty years of tillage and nitrogen fertilization. Geoderma 308, 46-53.

Ruoho-Airola, T., Syri, S., Nordlund, G. 1998. Acid deposition trends at the Finnish Integrated Monitoring catchments in relation to emission reductions. Boreal Environment Research 3: 205–219.

Situační a výhledová zpráva "Půda" 2015, report of Ministry of Agriculture of the Czech Republic, Prague, 2015, 134 p., ISBN 978-80-7434-252-3.

Sumner, M.E., Noble A.D. 2003. Soil acidification: the world story. In: Zdenko Rengel (ed.), Handbook of soil acidity. Marcel Dekker, Inc. New York, NY., USA. p. 1–28.

Vigovskis, J., A.Jermuss, A. Svarta, D. Sarkanbarde, 2016. The changes of soil acidity in long-term fertilizer experiments. Zemdirbyste-Agriculture, vol. 103, No. 2, 129–134.
Wang, C., M. Zheng, W. Song, S. Wen, B. Wang,C. Zhu, R. Shen, 2017. Impact of 25 years of inorganic fertilization on diazotrophic abundance and community structure in an acidic soil in southern China. Soil Biology & Biochemistry 113, 240-249.
Xun, W., W. Xiong, T. Huang, W. Ran, D. Li, Q. Shen, Q. Li, R. Zhang. (2016): Swine manure and quicklime have different impacts on chemical properties and composition of bacterial communities of an acidic soil. Applied Soil Ecology 100:38–44.
Zhang, H. M., Wang, B. R., Xu, M. G. and Fan, T. L. 2009. Crop yield and soil responses to long-term fertilization on a red soil in southern China. Pedosphere 19(2): 199–207.
Zhang, Y., W. de Vries, B. W. Thomas, X. Haod, X. Shia, 2017. Impacts of long-term nitrogen fertilization on acid buffering rates and mechanisms of a slightly calcareous clay soil. Geoderma 305, 92-99.
Zheg, M., W. de Vries, L.T.C. Bonten, Q. Zhu, T. Hao, X. Liu, M. Xu, X. Shi, F. Zhang, J. Shen, 2017. Model-Based Analysis of the Long-Term Effects of Fertilization Management on Cropland Soil Acidification Environ Sci. Technol. 51, 3843-3851.

16 General soil-improving cropping systems

T. Dalgaard, C. Christen, G.L. Lövei, M. Ferrante, V. Bhaskar, C. Kjeldsen, A. Ferreira, O. Green, I.S. Kristensen, M. Heinen, R. Rietra, P. Yang, R. Hessel and O. Oenema

16.1 Background

This chapter discusses soil-improving cropping systems (SICS) and the related areas of farm management that have not been designed specifically for a soil threat but have a general mode of soil quality improvement, and thereby contribute to a general improvement of soil functioning. These general SICS are based on the notion that all soils in agriculture need good management, so as to improve soil quality (irrespective of the aforementioned soil threats). The concept of 'soil threat' is not well-perceived and/or accepted in agricultural practice, in part because of the negative connotation, and this may hinder implementation of soil threat specific SICS. Stakeholders may perceive the concept of 'soil threats' as a policy construct, meant to implement restrictive regulations (which farmers often do not like). Though soil threats may occur throughout Europe and other parts of the world, not all soils are prone to one of the 11 soil threats (Chapter 4). Also, soil threats are not always recognized and understood, and hence, soil threat specific SICS may not be taken up easily.

Yet, many farmers are concerned about soil quality. Some of the concerns of farmers that are not (sufficiently) addressed by the concept of 'soil threats' include for example

a) improving soil structure, so as to ease seedbed preparation and the workability and earliness of the soil in spring, as well as the harvestability of the soil in autumn,
b) enhancing yield potential, closing yield gaps and improving gross margin,
c) enhancing soil nutrients and balanced nutrition (addressing all 14 essential nutrient elements[12]), while reducing nutrient losses through GHG emissions, leaching and denitrification, and
d) spatial variations in soil quality and soil functioning, which may cause a yield penalty (lower yield due to insufficient input optimization) or a cost penalty (due to high inputs in the wrong places).

Hence, there is also a need for 'general SICS', focussed on improving soil quality, productivity and sustainability, and on decreasing the environmental impacts of crop production.

Agronomists commonly define the crop yield potential of a site (land/field) by three main 'yield factors' (Van Ittersum and Rabbinge, 1997; Evans and Fisher, 1999), i.e.

i) yield <u>defining</u> factors: climate, carbon dioxide concentration, and genetic potential of the crop,

[12] N, P, K, Mg, Ca, S, Fe, Mn, Zn, Cu, B, Mo, Cl, Ni

ii) yield limiting factors: water and nutrient availability, and
iii) yield reducing factors: pests, diseases, weeds and pollutants (including high concentrations of salts) and excess water (causing oxygen stress). Soil threats have not been mentioned explicitly, but these may also reduce crop yield.

In this concept, soil quality boils down to its role in crop yield limiting and reducing factors. Based on this concept, a total of six indicators may be used to assess the capacity of the soil to produce biomass (crop yield):

1) Water retention and delivery to crops, i.e., soil depth and water holding capacity.
2) Nutrient retention and delivery to crops, i.e., fertility indices.
3) Control of pathogens and weeds.
4) Soil structure and tilth.
5) Control of pollutants.
6) Control of organic matter content and quality.

The first five indicators directly follow from the yield limiting and reducing factors. The sixth indicator (soil organic matter content) has been added because of its overarching role in the five main crop yield limiting and reducing factors, but also because soil organic matter content can be managed. Quantifying the first five indicators requires the measurement of a range of soil characteristics.

16.2 Purpose

The aim of this review is to perform a literature search on soil-improving cropping systems (SICS) that improve soil quality, productivity and the sustainability of the cropping systems. Results of specific components of SICS have been quantified as relative effects, i.e., the ratio of the specific treatment and the reference (control treatment) (see Chapter 5).

The remainder of this chapter first provides a summary qualitative assesment of SICS with focus on i) crop yield limiting and reducing factors and ii) aspects of cropping system sustainability, also to provide a coherent overview (Section 16.3). Secondly, it provides a more detailed quantiatitive descriptions of impacts and assesments of SICS on a) crop rotions, b) nutrient management, c) irrigation and fertigation, d) controlled drainage, e) tillage, f) pest management, g) weed management, h) residue management, i) mechanization management, j) landscape management, as well as k) the role of Environmental conditions and socio-economic factors (Section 16.4).

16.3 Summary of qualitative assessments of SICS

16.3.1 Crop yield limiting and reducing factors

An overview of the effects of components of SICS on soil characteristics that influence crop yield limiting and reducing factors is presented in Table 16.1, using the above-mentioned 6 main indicators (1) soil water delivery, (2) soil nutrient delivery, (3) control of soil-borne pathogens and weeds, (4) soil structure and tilth, (5) control of pollutants, and (6) control of

soil organic matter content and quality. The reference (control) has been given a score of 0 (zero), a positive effect of the specific treatment on cropping system sustainability has been given the score + or ++, while a negative score has been given the score – or --. The greatest number of subcomponents is shown in Table 16.1 for crop types and crop rotations, and yet this list is just a very short summary of all possible crop types and crop rotations.

Table 16.1. *Overview and qualitative assessment of components of SICS: crop yield limiting and reducing factors.*

Components of SICS	Water delivery	Nutrient delivery	Control of Pathogens	Improving Structure	Control of Pollutants	Improving SOM
Monocultures (reference)						
wide rotations (1:6)	+	+/-	++	+	+/-	+
narrow rotations (1:3)	+	-/+	+	+/-	-/+	+
+ root crops (1:2)	+	-/+	+/-	-	-/+	-
+ legumes(1:3)	+	+	+	+	-/+	+
+ allelopathic plants (1:4)	-/+	-/+	+	-/+	+/-	-/+
+ cover crops(1:1)	-/+	+/-	+/-	+	-/+	-/+
+ intercropping	+	+/-	+	+/-	+/-	+/-
+ green manures (1:1)	-/+	+/-	+/-	+	+/-	+
+ phytoremediation	+	-/+	+/-	+/-	+	+/-
Fallow/set-aside (1:6)	++	+	+	+	+/-	-/+
No fertilization (reference)						
organic fertilization	+	++	-/+	+	-	++
mineral fertilization	+	++	-/+	+/-	-	+/-
No irrigation (reference)						
irrigation	++	+/-	-/+	-/+	+/-	-/+
fertigation	++	++	-/+	+/-	-/+	-/+
No drainage (reference)						
drainage	+/-	+/-	-/+	+	+	-
No tillage						
conventional tillage	-/+	+/-	+	+/-	+/-	-
minimum tillage	-/+	+/-	+/-	+/-	+/-	-/+
No pest management (ref.)						
chemical control	+	+	++	-/+	--	+/-
biological control	+	+	++	+/-	+/-	+/-
No weed control (reference)						
chemical weed control	+	+	+	-/+	--	+/-
biological/mechanical control	+	+	+	+/-	-/+	+/-
No mulching						
organic mulching	+	+/-	-	+	+/-	+
plastic mulching	++	+/-	-/+	-	-	+/-
No controlled trafficking						
controlled trafficking	+/-	+/-	+/-	+	+/-	+/-
No landscape management (ref.)						
landscape management	+	+/-	+	+/-	+/-	+/-

For example, the crop statistics of Eurostat distinguishes 17 categories for cereals and 29 for other main crops, 40 categories for vegetables, 41 for permanent crops. Within a crop type, large differences in varieties can exist, which can have a profound effect on crop productivity, farm income, resource use and environmental impacts.

Crop rotations do have a positive effect on soil functioning, compared to monocultures, which is mainly related to enhancing biodiversity and suppressing soil-borne diseases and weed infestations (Table 16.1). Crop rotations have a positive effect on yield and soil biodiversity. They may have a positive effect on soil water and nutrient delivery, because healthy crop rotations often explore a greater volume of soil. Crop rotations also tend to have a positive effect on soil structure and soil tilth, because of the diversity of rooting patterns and soil organic matter sources. Root crops in crop rotations often have a negative effect on soil structure due to the disturbance of soil structure during harvesting and the low amounts of residual biomass left in the soil (but are often financially attactive). This effect may be mitigated/restored again by a subsequent cereal crop or oilseed crops.

Fertilization enhances the capacity of the soil to deliver nutrients, and thereby increases crop production and residual crop biomass returned to the soil (Table 16.1). However, fertilization commonly increases the environmental impacts through leaching of nutrients (mainly N and P) to surface waters, and through the emission of nitrous oxide (N_2O) and ammonia (NH_3) to air. The fertilization source (inorganic vs organic) has a large effect on the nutrient delivering capacity, soil carbon sequestration and emissions. Fertilization indirectly enhances also the water delivery capacity of the soil and the water use efficiency, because a more vigorous crop explores a larger volume of soil. The production of synthetic fertilizers is energy intensive and is associated with CO_2 emissions.

Drainage is extremely important in the case of temporary water logging and high groundwater levels. Drainage will increase the rooting depth, decrease the heat capacity of the soil and thereby accelerate the warming up of the soil in early spring. Drainage may also increase the mineralization of soil organic matter and thereby lower the soil organic matter content and increase the release of carbon oxide (CO_2) to the air. Drainage may decrease nutrient losses via denitrification (Table 16.1). Controlled drainage significantly reduces the volume of drainage water and the corresponding N-load.

Irrigation enhances the water delivering capacity of the soil, and indirectly the nutrient delivery (because of the increased volume of roots and the increased solubility and accessibility of soil nutrients), and thereby may result in increased yields and increased resource use efficiencies (WUE (water use efficiency), NUE (nutrient use efficiency), water productivity). However, irrigation may increase the risk of leaching and denitrification/N_2O emission (Table 16.1).

Tillage is important for weed control and seedbed preparation. Interestingly, the invention and improvements of the plough have greatly contributed to soil productivity in history (Mazoyer

and Roudart, 2006), but tillage is currently associated with organic matter decline, high energy use, erosion and loss of biodiversity. As a result, reduced tillage (minimum and zero tillage) is promoted. However, reduced tillage often leaves crop residues on the soil surface, which has been associated with increased infestations of crop diseases, which then may require additional inputs of chemicals. Deep ploughing is locally practiced to bring 'virgin' and high quality subsoil to the top and at the same time bury the less desirable top soils. Results of deep ploughing are variable and often questioned, because of the high energy use. Subsoil lifting is done to alleviate subsoil compaction; again results are often variable, but with the development of new cultivation machinery e.g. low disturbance subsoiler, there is the potential to obtain some of the no-till benefits without all the negatives (Table 16.1).

Pest management has greatly contributed to the increased crop yields obtained during the last century. Two variants are often distinguished, i.e. chemical pest control and biological pest control. The first allows somewhat more narrow rotations at the expense of pesticides. Chemical pest control is also debated because of its negative effects on biodiversity and insects abundance (which in turn are needed for fertilization of many crop species, and the biological control of pests). Biological control is based on wide rotations, multispecies crops, buffer strips and landscape management, which allow the development of species-rich insects. The best option is often a combination of the two: integrated pest management (Table 16.1).

Weed management is also extremely important in agriculture, as weed infestations can ruin the target crop. Again, two variants are often distinguished, i.e. chemical weed control and mechanical/biological weed control. The first variant makes use of herbicides, while the second variant makes use of mechanical weeding, ploughing and target crop rotations. Chemical weed control is also debated because of its negative effects on biodiversity and insect abundance (which are needed for fertilization of many crop species). The best option is often a combination of the two: integrated weed management. Proper selection of crops in rotation may greatly contribute to weed suppression (Table 16.1).

Mulching is often practiced in combination with zero tillage, also to reduce evaporation and water erosion, and thus to enhance crop yield and water productivity. Plastic mulching is extensively practiced in semi-arid regions of for example China and India and in intensively managed horticulture cropping systems in Europe, as a method to increase water productivity and the temperature of the soil in early spring, and to suppress weed development (Table 16.1). However, plastic mulching often leaves large amounts of plastic fragments in soil and the wider environment.

Traffic management is important in mechanized agriculture where wheel loads are often too high to prevent subsoil compaction. Controlled trafficking is a way to minimize traffic on land, in combination with using the same wheel tracks more often (to spare the remainder of the land). It has often a positive effect on crop yield, soil quality and energy use (Table 16.1).

Landscape management goes beyond the farm scale and is not yet much considered in cropping system management. It often involves more stakeholders than just farmers. In the UK, there have been initiatives to set up "farm clusters" and "river catchment clusters", to get all the stakeholders working together towards a common goal. There is increasing evidence that landscape management may contribute to soil quality, crop productivity and sustainability, as it may contribute to the control of various threats (e.g., erosion, desertification, acidification, pollution, loss of biodiversity and flooding) and may affect the micro-climate and the control of pests. Landscape management allows to broaden the sources of income and market orientation. A special aspect is the integration of crop-livestock production systems, which has advantages also for the environmental sustainability of livestock production (Table 16.1).

16.3.2 Aspects of cropping system sustainability

An overview and first qualitative assessment of components of SICS in terms of cropping system sustainability is presented in Table 16.2. The impacts of components of SICS are assessed in terms of five indicators for cropping system sustainability:

(i) soil quality (see above; composite of the six crop yield limiting/reducing factors),
(ii) crop yield and crop quality,
(iii) farm income, i.e., the net balance between sales and production costs,
(iv) resource use efficiency, a measure for the ratio of output over inputs of resources, and
(v) environmental effects (emissions of nutrients, pollutants and greenhouse gases).

Table 16.2 distinguishes the same components of SICS as in Table 16.1. The reference (control) has been given a score of 0 (zero), a positive effect of the specific treatment on cropping system sustainability has been given the score + or ++, while a negative score has been given the score – or --.[13]

Crop rotations, fertilization, irrigation, drainage, and pest and weed control all have a large effect on farm income. Tillage, mulching, traffic management and landscape management have in general a modest effect on farm income. Fertilization, irrigation, drainage, and pest and weed control often have a negative effect on the environment, but the assessment differs when the effects are based on a product or area basis. The environmental effects often have a minimum at optimal inputs of fertilizers, irrigation, drainage, and pest and weed control when the environmental effects are expressed on a product basis (De Wit, 1992; Van Groenigen et al., 2010). The same holds for resource use efficiency. High (excessive) inputs generally have negative environmental effects, both expressed on a product and area basis. Hence, the assessment of the effects of inputs depend on (i) the level (rate) of input, and (ii) the units chosen, i.e. area or product basis.

[13] The scores highly depend on the reference situation, i.e., positive effects will be obtained only if the reference situation does not have an optimal soil quality and/or result in optimal crop production, and vice versa.

Table 16.2. *Qualitative assessment of components of SICS: aspects of cropping system sustainability.*

Components of SICS	Crop yield & quality	Soil quality	Farm income	Resource use efficiency	Environmental impacts
Monocultures (reference)					
Wide rotations (1:6)	+	+	+	++	++
Narrow rotations (1:3)	+/-	+/-	++	+/-	+/-
+ root crops (1:2)	++	-	++	+/-	-
+ legumes(1:3)	+	+	+	++	+
+ allelopathic plants (1:4)	-/+	+	-/+	+/-	0
+ cover crops(1:1)	+	+	-/+	+	+
+ intercropping	++	+	+/-	++	+
+ green manures (1:1)	++	++	+/-	+	+
+ phytoremediation	+/-	+	+/-	+/-	+
Fallow/set-aside (1:6)	--	+	--	--	-
No fertilization (reference)					
organic fertilization	++	+	++	+	-
mineral fertilization	++	+	++	+	-
No irrigation (reference)					
irrigation	+	+/-	+	+/-	+/-
fertigation	++	+	++	++	+/-
No drainage (reference)					
drainage	+	+	+	+	+/-
No tillage (reference)					
conventional tillage	+	-/+	-/+	-/+	-/+
minimum tillage	+	+/-	+/-	+/-	+/-
No pest management (reference)					
chemical control	++	-	++	++	-
biological control	++	+/-	++	++	+/-
No weed control (reference)					
chemical weed control	++	-/+	++	++	-
biological/mechanical control	++	-/+	++	++	+/-
No mulching					
organic mulching	+/-	+/-	+	+	+
plastic mulching	+	-/+	+	+	-/+
No controlled trafficking (reference)					
controlled trafficking	+	+	+/-	+	+
No landscape management (ref.)					
landscape management	+/-	+/-	+/-	+/-	+/-

The assessments in Table 16.2 do not consider possible interactions between components, which can be positive (synergistic) and negative (antagonistic). For example, fertilization is most attractive when there are no other growth constraints than nutrient elements. The same applies to irrigation; it is economically most profitable when no other growth limiting and reducing factors occur.

Based on a comprehensive literature study, the following sections review the quantitative effects of the key areas of SICS management practices. This include the effects of

a. Crop rotions,
b. Nutrient management,
c. Irrigation and fertigation,
d. Controlled drainage,
e. Tillage,
f. Pest management,
g. Weed management,
h. Residue management,
i. Landscape management, and
j. The role of Environmental conditions and socio-economic factors.

As a starting point, the 734 papers collected in the review by Hedlund et al. (2016), were used to overview literature references in relation to the identified dimensions of SICS, in addition to the results reviewed in the other chapers of the present report. Preference was given to published meta-analyses and reviews, or otherwise based on own reviews of selected studies.

16.4 Quantitative impacts and assessments of SICS

16.4.1 Crop rotations

A meta-analysis of the effect of pre-crops in Europe, America and Australia, using 831 comparisons between wheat after wheat or wheat after other break crops, show convincing benefits of crop rotation on wheat yield (Angus et al., 2015). The yield benefit of crop rotation ranges between 500 and 1500 kg ha^{-1} $year^{-1}$ (Figure 16.1).

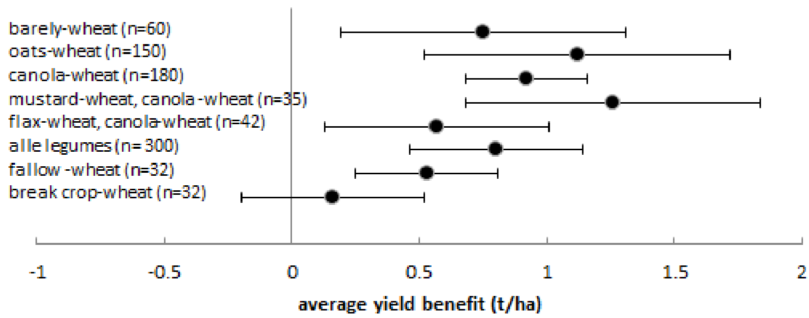


Figure 16.1. *Benefit of a break crop on wheat yield compared to wheat after wheat. Note: absolute yield increases are presented on the x-axis, because the benefit is not proportional to yield (Angus et al., 2015) (n is the number of comparisons) (bars show 95% confidence interval).*

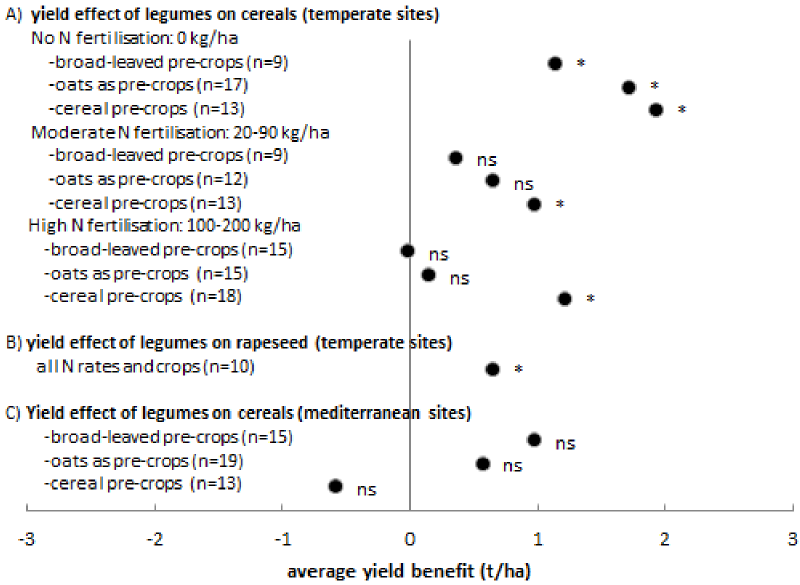


Figure 16.2. *Benefit of legumes as break crop on wheat yield compared to wheat after wheat in Europe (Preissel et al., 2015). ns: not significant; *: all data are positive (n is the number of comparisons).*

Another meta-analysis examined the effect of pre-crops on the yield of wheat, and specifically legume pre-crops (Preissel et al., 2015). Results indicated that the effect depends on the nitrogen fertilisation: yield benefits are highest under low nitrogen fertilisation (Figure 16.2). Fertilisation to subsequent crops can be reduced to 60 kg N/ha on average and 23-31 kg N/ha when aiming at maximising. In the studies reviewed, 35 out of 53 crop rotations with grain legumes were competitive with comparable non-legume rotations (Preissel et al., 2015).

Rotations increase soil microbial biomass C and N (Figure 16.3; McDaniel et al., 2014). Soils under a higher diversity of crops in rotation also produce a higher microbial richness and diversity. Whether the overall rotation-effect on microbial diversity promotes ecosystem functioning in terms of nutrient cycling and resilience to stress remains unclear (Venter et al., 2016).

Various meta-studies have been performed on the effect of crop rotation on environment (GHG emissions) and soil quality (Figure 16.3; Sainju, 2016; West and Post, 2002). C sequestration rates from 67 long-term agricultural experiments, consisting of 276 paired treatments indicate that enhancing rotation complexity can sequester on average 20±12 g C m^{-2} (excluding a change from continuous corn to corn-soybean). Soil organic matter may reach a new equilibrium after an enhancement in rotation in approximately 40 to 60 years (West and Post, 2002). Due to the C sequestration crop rotation results in lower greenhouse gas emissions, on product basis, but especially per hectare (Sainju, 2016).

Various studies have examined also the effect of crop rotation on other issues, including soil structure (Munkholm et al., 2013), pesticide use (Andert et al., 2016), and energy output (Deike et al., 2008; Tomasoni et al., 2011). These studies, in general, show perspectives for intensification of production in combination with more diverse crop rotations.

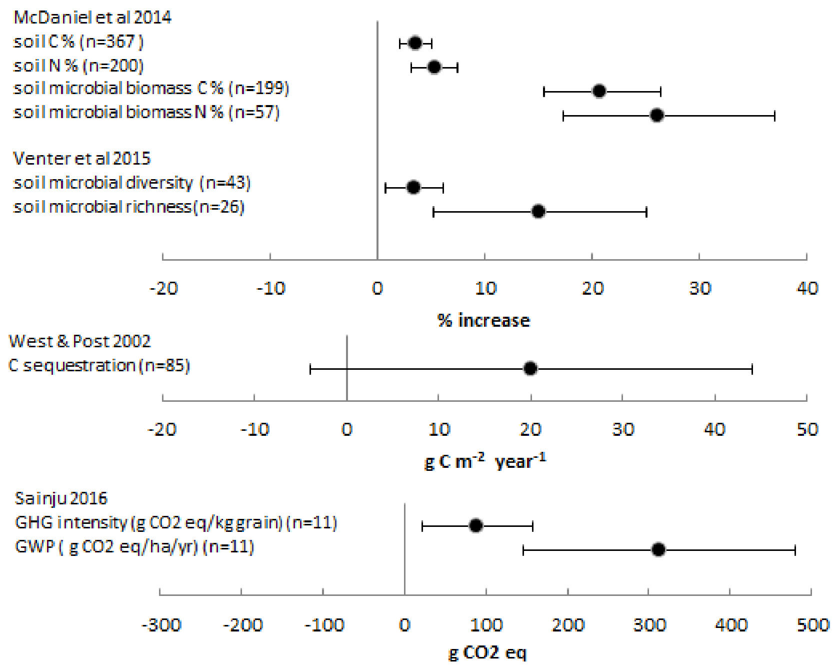


Figure 16.3. *Results of various meta-analyses studies: effects of crop rotations on soil, soil carbon and GHG (global)(McDaniel et al., 2014; Sainju, 2016; Venter et al., 2016; West and Post, 2002), (n is the number of comparisons).*

In a meta-study, Vicente-Vicente et al. (2016) examined the effect of cover crops in permanent cropping systems (Figure 16.4). Perennial grasses had a significant positive effect on soil organic matter in vineyards (n=33), but the effect was not significant in olive trees (n=18)). Vegetation change from arable to the perennial Miscanthus showed a small positive effect on soil organic matter content (Figure 16.4; Poeplau and Don, 2014). Another study on 31 sites (930 plots) comparing mixed grasslands versus monocultures showed 32% higher yields in mixed systems, all including four species with different traits including nitrogen-fixing species, persistent and fast growing species (Figure 16.4; Finn et al., 2013).

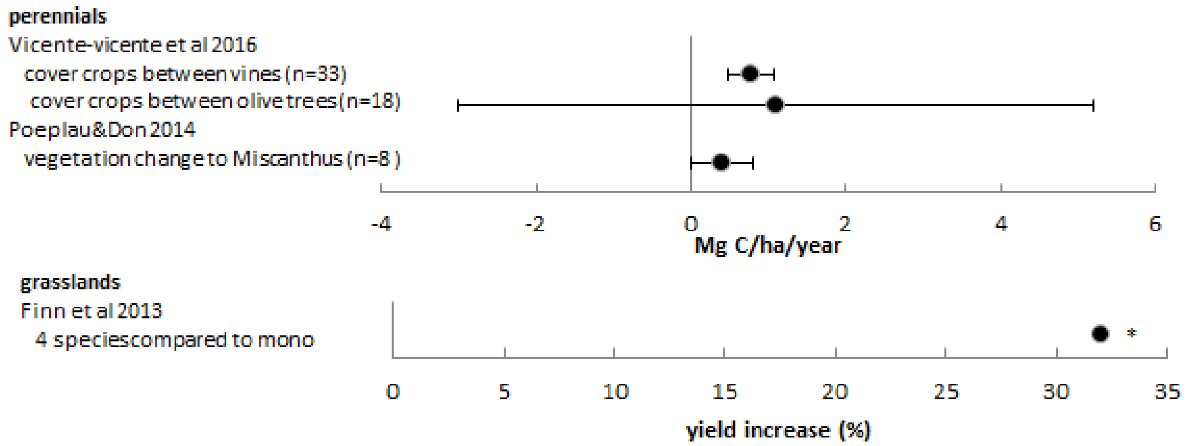


Figure 16.4. *Results of three meta-analyses studies: effects of cover crops and mixed species in perennials on yield..*

Meta-analyses studies have shown that the effects of intercropping on the total yield of intercropped crops is positive (Figure 16.5). The overall effect depends on sowing densities, sowing time and fertilisation (Bedoussac et al., 2015; Yu et al., 2016) but also on soil factors (Cong et al., 2015a; Cong et al., 2015b).

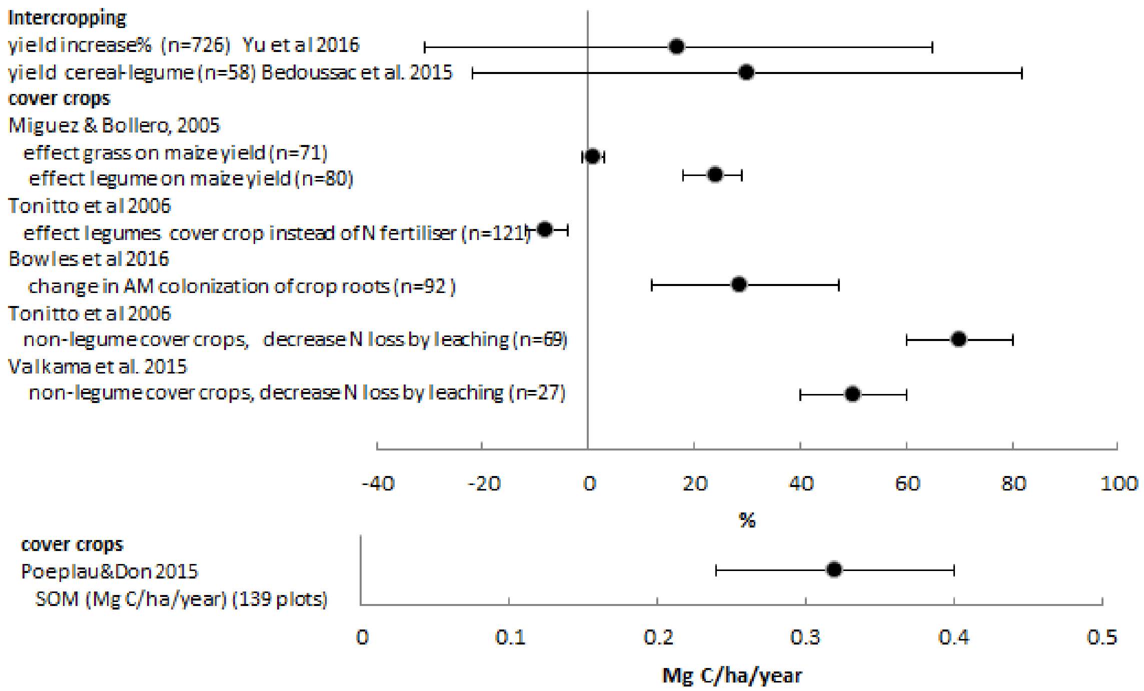


Figure 16.5. *Results of various meta-analyses studies: effects of intercropping, mixed crops, cover crops on yield and environmental impacts.*

The effect of cover crops has been studied in various meta-analysis studies (Figure 16.5). While grass as a cover crop showed no effect, legume cover crops show significant higher yields of the cash crops in similar fertilised systems (Miguez and Bollero, 2005). The potential of the use of legume cover crops to replace fertilisers depends on the N supply from the legume cover crops. On average, without making distinction between high and low N supply by legumes, yield decreased only slightly, showing that legume cover crops have a potential to replace N fertilisers (Tonitto et al., 2006). Cover crops can increase N supply but also increase mycorrhiza colonization of cash crop roots (Bowles et al., 2016). While legume cover crops can supply N to crops, non-leguminous cover crops can decrease N loss by leaching (Tonitto et al., 2006; Valkama et al., 2015). Compared to fallow soils, cover crops increase the soil organic matter content (Poeplau and Don, 2015).

In summary, meta-analysis studies on crop rotations confirm the hypothesis that rotation versus mono-cropping results in higher yield. While rotation also has an effect on soil and microbial diversity, the effect of the soil improving factors remains unclear in terms of nutrient cycling and resilience to stress, because studies have not focused on these aspects. Intercropping, mixed crops and cover crops can increase yields. Legume cover crops can be a substitute for N fertilisers, while other cover crops can decrease N losses via leaching. The soil improving effects are at least the improved mycorrhiza colonization of cash crops and enhanced C sequestration due to cover crops, and the improved decomposition and subsequent nutrient release during intercropping. The mixing of species in grassland systems can be designed in such a way that it results in less use of N fertilisers and higher yields compared to monocultures.

16.4.2 Nutrient management

This section reviews the effects of nitrogen (N) and phosphorus (P) management in arable farming systems, in the form of soil quality benefits, yield optimization and the reduction of environmental pollution (e.g. nitrate leaching and soil erosion).

From a socio-economic and environmental viewpoint of nutrient management, N fertilisation must be based on the balance between crop requirement and N input from manure and fertiliser, considering soil N release (Schoumans et al., 2011). Current Agro-environmental policies therefore recommend nutrient budgeting (farm-gate, field and farm-system budgets) that take in account of nutrient inputs, storage/retention and outputs to advance farm nutrient efficiency and to minimise various forms of nutrient leakages (Dalgaard et al., 2012; Langeveld et al., 2007). The review by Follett & Hatfield (2001) indicated the importance of integrating nutrients from fertilizer, legumes or recycled manures and composts. The EU Nitrate Directive specifies maximum manure application standard as 170 kg N/ha, especially for Nitrate Vulnerable Zones (Schoumans et al., 2011).

The P balance should be based on the available soil P (Schoumans et al., 2011). Organic experiments by Ivarsson et al. (2001) found P from harvested crops was on average 10

kg/ha/year without P fertilization in a six-year rotation with cereals, legumes, a tuber crop and a green manure crop. Despite P output, the plant-available soil P content reduced substantially only at one site. Soil P studies by Chardon & Schoumans (2007); Sharpley et al. (1994) revealed that most European soils do not require additional P fertilization to support crop productivity. In general, at high agronomic soil P status, achieving negative P balance is crucial, allowing reduced manure or fertiliser inputs by considering crop response and soil nutrient status (Schoumans et al., 2011).

Attention should be paid to the type and amount of manure because modifying N can cause surplus P and K. Similarly, one should take into account the chemical form of nutrient fertilizers (e.g., NO_3^- vs. NH_4^+) along with their dissolution rate (slow or rapidly available N or P) (Schoumans et al., 2011). Additional consideration includes placement and timing of manure or fertilizer. Sørensen & Thomsen (2005) showed a significantly lower N pollution by incorporating solid manure from slurry in spring just before crop seeding. Hofman et al. (1992) reported that for certain crops having larger inter-row distances, with limited root distribution and immediately earthed-up after fertilization, row or band application effected lower residual mineral N at harvest, and reduced leaching and ammonia volatilisation. In doing so, P fertilization can also be reduced up to 75% over broadcasting (Van Dijk & Van Geel 2012). Other strategies are: split-nutrient fertilization, controlled release fertiliser (for example, composting solid manure reduce readily available N thus decreasing leaching losses), improving organic matter through soil amendments and avoiding manure and fertiliser application before predicted heavy rainfall periods (Schoumans et al., 2011; Ulén et al., 2010). Manure injection or ploughing-in manure directly after application potentially result in increased nutrient-use efficiency and decreased nutrient losses (Uusi-Kamppa & Mattila 2010).

Soil management measures, especially reducing tillage intensity or by direct drilling, proved to reduce soil erosion and particle P losses, compared to inversion tillage (Ulén et al., 2010). Similar results with shallow cultivation were achieved from erodible clay loams, silty and clayey soils grown with cereals (Strauss & Smid 2004). A review by Wivstad et al. (2005) indicated that by employing reduced tillage, and with grass-ley as an integral component, nitrate concentration in autumn and winter was reduced by 25-50% and soil erosion was prevented. In general, for erodible soils, Lundekvam (2007) demonstrated reduced soil erosion and total P losses up to 80% by spring ploughing over traditional autumn ploughing for a spring crop. An experimental study by Stenberg et al. (1999) has also demonstrated that delayed ploughing from early autumn to late autumn or spring reduce N leaching risk.

Amount of crop residues returned to the soil is crucial for internal nutrient flow, especially meeting P demand, if conditions favour mycorrhizal fungi (Li et al., 1990). In addition, the authors demonstrated the influence of type of residue, for example lucerne crop residues provided a larger quantity of extractable P than residues from pea and wheat. Long-term studies by Paustian et al. (1992) showed crop residue return on increasing soil organic matter content in different soil pools, compared to continuous fallow during 30 years and also cereal

cropping without straw return on a sandy clay loam. Recently, Raberg et al. (2017) studied three contrasting application treatments of recycling residual and green-manure biomass namely A) leaving the biomass on the field at harvest, B) biomass redistribution without anaerobic digestion, and C) biomass distribution with anaerobic digestion as bio-fertilizer in a 3-year designed non-legume crop sequence. The results indicated that compared with treatment A, treatment B increased lentil yields intercropped with oat in one of the two studied years. In both the years, biomass yield of the cover crop following winter rye was significantly higher in treatment B than A. The legume proportion in the green-manure ley was significantly higher in treatment B and C with additional benefits of maintaining crop yields similar to that of treatment A. Thus, biomass recycling with or without anaerobic digestion contribute to efficient internal nutrient cycling. Nutrient-use efficiency can be improved by varying rooting pattern. Deep rooting annual crops access and retain N, otherwise it moves below the accessible rooting depth. Studies by Grant & Lafond (1994) demonstrated smaller N accumulation in 60-120 cm soil depth after winter wheat (20-25 kg N/ha) than after spring sown wheat (27-116 kg N/ha) or flax (31-91 kg N/ha). Similarly, low levels of leachable soil N was detected in pea-barley intercrop, compared to single-cropped pea by Hauggard-Nielsen et al. (2001). This effect was partly due to fast growth of barley roots accelerating the onset of soil N uptake and decreasing early leaching losses.

Biological reactive nitrogen fixation (BNF) by legumes (grain or forage) or green manure crops supplies N not only to the legume but also to the whole crop rotation, replacing synthetic N input needs (Peoples et al., 1995). Amossé et al. (2014) evaluated four relay intercropped legumes (Medicago lupulina, Medicago sativa, Trifolium pratense and Trifolium repens) on their N contribution to the associated and subsequent cash crops in six organic farms. The results showed increased subsequent spring crop N uptake encouraging 30% higher maize yields, without negative effect on the associated winter wheat N uptake. In addition, BNF by legumes varied between 38 and 67 kg/ha. Wivstad et al. (2005) reported grass/clover BNF as high as 250 kg/ha in southern Sweden, largely depending on climatic conditions. A recent long-term organic and conventional 4-year arable crop rotation experiment in Denmark by Pandey et al. (2017) indicated higher BNF in coarse sandy and loamy sandy soils having organic with grass-clover, organic with grain legumes and conventional with grain legumes, compared to similar rotation under sandy loam soils. The studies also indicated that legumes in rotation can maintain BNF with or without animal manure and with or without mineral fertilizer. Time of green manure incorporation also needs attention. Studies by Känkänen et al. (1998) showed that too early autumn incorporation effected higher N leaching during the first winter, while, too late incorporation in autumn or early spring affected succeeding crop yields and potential N loss at the end of the growing season, during the following winter (Bergström & Kirchmann 2004). Increased availability and uptake of P, due to rhizosphere interspecific facilitation in legume based bi-cropping systems, was reported by several experimental studies for example, soybean-wheat (Bargaz et al., 2017); maize-faba bean (Li et al., 2007); maize-cowpea (Latati et al., 2016) and barley-pea (Hauggaard-Nielsen et al., 2009).

Diversified crop sequences (with or without legumes) within the rotation are essential for obtaining comparable or even higher crop yields with less or no N inputs (Mitchell et al., 1991). A study by Nevens & Reheul (2001) confirmed significant yield increase of silage maize when preceded by leguminous fodder beet and leguminous field bean, compared with silage maize upon silage maize. The relative yield gains were highest without N fertilization. However, the positive rotation effect disappeared almost completely at a N rate of 180 kg/ha. Long-term six-cropping system experiments (2004-2010) by Plaza-Bonilla et al. (2015) with three 3-year rotations involving different number of grain legumes (GL1, GL2 and GL0) with or without cover crops showed decreased cumulative N leaching when increasing the number of grain legumes in the rotation and with cover crop inclusion. The study also indicated that the cash crops N uptake increased when increasing the number of grain legumes without being negatively affected by the reduction in the amount of N fertilizer in the GL1 and GL2 rotations, compared to GL0.

Depending on the growing season (long or short), catch crops can be sown after main crop harvest or by undersowing in cereals (Karlsson-Strese et al., 1998; Constantin et al., 2010). Experimental studies in temperate regions by Thomsen (2005) and Hansen & Djurhuus (1997) found that Italian/perennial ryegrass undersowing in spring barley significantly reduced N leaching when allowed to grow during autumn and winter. In a Finnish experiment by Känkänen & Eriksson (2007), a catch crop mixture of timothy and Italian ryegrass proved to be efficient in using the time available for crop growth, as N uptake in the Italian ryegrass was highest in autumn, whilst for timothy this was in spring. Similarly, studies by Bergkvist et al. (2011) and Aronsson et al. (2014) showed that undersown clover/grass mixtures reduced soil mineral N content and leaching to the same extent as a pure grass catch crop.

A meta-analysis integrating 35 Nordic studies by Valkama et al. (2015) found that non-legume catch crops, specifically ryegrass species reduced N leaching loss by 50%, and soil inorganic N by 35% in autumn in comparison with no catch crops. Contrastingly, legume catch crops (white/red clovers) did not have any effect on N leaching. In addition, the risk of N leaching was consistent across studies conducted on different soil textures (clay and coarse-textured mineral soils) with different ploughing times (autumn or spring), N fertilization rates (up to 160 kg/ha), and amounts of annual rainfall. However, quantitative reduction (kg/ha) by catch crops was substantial on sandy soils due to their high levels of N leaching and soil nitrate N. Subsequently, with increasing N rates, the quantitative reduction in N leaching was higher, as the percentage decrease remained consistent. The study also showed that non-legume catch crops reduced grain yield by 3% although no effect in grain N content was found, whilst legume catch crops increased both grain yield and grain N content by 6%.

Other meta-analytic studies by Quemada et al. (2013) and Tonitto et al. (2006) also showed N leaching reduction from 50 to 70% with non-legume catch crops. The variable effects are mainly due to the differences in soil and climate, the weather conditions of the year and the cropping system. Another experimental studies by Hansen et al. (2007) compared undersown

Italian ryegrass in silage spring barley over barley grown to maturity with (perennial ryegrass) or without an undersown catch crop and with all treatments receiving 0, 60 or 120 kg of ammonium-N/ha from cattle slurry. The results indicated that treatments without catch crop, silage barley/Italian ryegrass reduced leaching by 63–320 kg N/ha, and the perennial ryegrass reduced leaching by 34-86 kg N/ha. In addition, during the subsequent growing year, the leaching following catch crops was substantially lower than for the bare fallow.

The review by Aronsson et al. (2016) showed from lysimeter and field scale studies that catch crops not necessarily have an effect on total P losses by runoff and leaching. Contrastingly, prevailing climatic conditions such as winter freezing-thawing may pose risk of dissolved P losses from catch crop biomass if not P is adsorbed efficiently by the soils.

In summary, from a SICS viewpoint, appropriate site-specific (considering weather risk) multifunctional rotational design through diverse crop types, increasing the frequency of legume or non-legume catch crops and when needed, nutrient correction using precision application of soil amendments improve functioning of cropping systems by enhancing soil nutrient cycling with other ecological services (Table 16.3).

Table 16.3. *Nutrient management indicators on environmental protection*

Components	Nitrate leaching	Soil erosion/ P loss	Cash crop yields	Profitability
Catch crop (legume or non-legume)	- -	0/+	+	+
Reduced tillage + diverse crop types and crop sequence pattern within rotation	- -	- -	+/-	+
Less inputs/precision fertilisation	- -	- -	+/-	++

16.4.3 Irrigation and fertigation

Based on 29 reported studies from the Mediterranean, complemented with studies from Australia (4), California (15) and Chile(2), Cayuela et al. (2016) performed a meta-analysis of the N_2O emissions from Mediterranean cropping systems, and proposed a more robust and reliable regional emission factor (EF) for N_2O, distinguishing the effects of, amongst others, water management (irrigation). Based on all irrigation studies they concluded that the EF is 37.4% less (significantly) than the IPCC default value of 1% (Figure 16.6). From the four irrigation methods studied, only sprinkler irrigation did not differ significantly from the IPCC default.

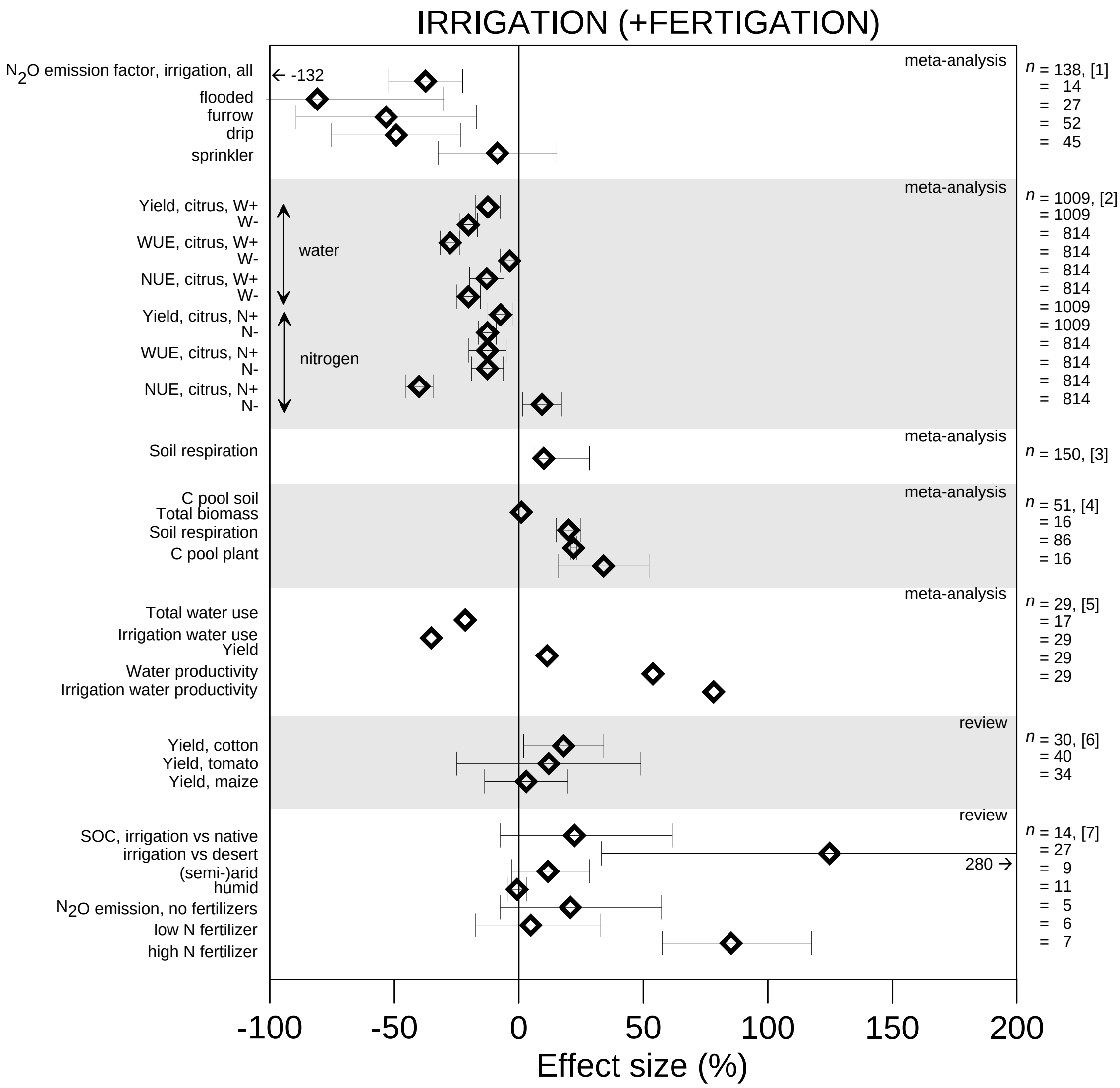


Figure 16.6. *Summary overview of various meta-analyses studies. Mean effect sizes and corresponding confidence intervals of the overal effects of irrigation and fertigation ([1]: Cayuela et al., 2016; [2]: Qin et al., 2016; [3]: Zhou et al., 2016a; [4]: Zhou et al., 2016b; [5]: Jagannath et al., 2013) and two reviews ([6]: Lamm, 2016; [7]: Trost et al., 2013). At the left the effect studied is mentioned, and on the right hand side the number of data-pairs (n) and the link to the reference between [] is given.*

Qin et al. (2016) reported a meta-analysis study on water and nitrogen (N; often supplied via fertigation) effects on citrus yield (number of observations n = 1009 in 55 studies), water use efficiency (WUE) and nitrogen use efficiency (NUE) (both: n = 819). They defined as a reference the situation with the highest yield in a certain study or the WUE and NUE associated with optimal water and optimal N inputs. Except for sub-optimal supply of N, both yield, WUE and NUE were less under over-optimal (W+, N+) and sub-optimal (W-, N-) water or N supply (Figure 16.6). All reported effects were significant.

Zhou et al. (2016a) performed a meta-analysis of 150 multiple-factor studies to examine the main and interactive effects of global change factors on soil respiration (and its two

components: autotrophic, and heterotrophic) and reported a significant increase of 9.7% due to irrigation (Figure 16.6). The authors also considered the effects of elevated CO_2, warming and nitrogen addition, and mutual interactions: all factors and interactions significantly increased CO_2 respiration.

Zhou et al. (2016b) carried out a meta-analysis of 179 published studies to examine responses of soil C storage and associated C fluxes and pools to drought and irrigation. Irrigation did not significantly increase the soil C pool (1.3%), but significantly affected total biomass production (20.4%), soil respiration (21.9%), and total plant C pool (34.4%) (Figure 16.6).

Jagannath et al. (2013) reported a meta-analysis based on data from 29 published studies comparing SRI (System of [intensive] Rice Intensification) and non-SRI methods (reference) for irrigated rice production that had reported results from a total of 251 comparison trials. Total water use and irrigation water use is decreased by 22% and 35%, respectively; yield, water productivity and irrigation water productivity is increased by 11%, 54%, and 78%, respectively (Figure 16.6). No information was provided on the level of significance.

Lamm (2013) reviewed the benefits of subsurface drip irrigation (SDI) over other types of surface irrigation: 16 studies for cotton, 16 studies for tomato, 12 studies for maize. Only for cotton a significant increase in yield (18%) was obtained, whereas the yield increase for tomato (12%) and maize (3%) were not significant.

Trost et al. (2013) published a review on the effect of irrigation on soil organic carbon (SOC) and N_2O emissions (denitrification), based on 22 studies spread over the world (14 used for SOC; 8 used for N_2O emission). Based on their reported original data (their Tables 2&3[14]) the effect size was recalculated according to the procedure given in Chapter 5 (ln-transformation was applied). SOC increased most for the studies where irrigated (+fertilized) desert soils were compared to native desert soils: 125% (significant). Lower increases in SOC were obtained in studies in arid and semi-arid conditions: 22% (irrigated arable land versus land with native vegetation; not significant) and 12% (irrigated arable land compared to non-irrigated arable land; not significant) (Figure 16.6). In humid climate the effect of irrigation was negligible (-1%; not significant). On average the N_2O emission increased under irrigation. The increase was 21% (not significant) for the studies without consideration of N-fertilizer inputs. In case of low N-fertilizer inputs the increase was 5% (not significant; excluding 1 extreme observation), and in case of high N-fertilizer inputs the increase was 85% (significant) (Figure 16.6).

In summary, irrigation may increase yield and resource use efficiency (WUE, NUE, water productivity), provided proper management has been implemented. The effect of irrigation on the C pool of the soil is only marginal, unless desert soils are irrigated. Soil respiration, however, increases significantly under irrigated soils. For highly fertilized soils under irrigation a

[14] The summary data in their Figures 4&5 are not always reproducible from the data in their Tables 2&3. Here we used all data provided in their Tables 2&3.

significant increase in N_2O emission is reported. However, the N_2O emission factor for different irrigation systems is less than the standard IPCC emission factor. Unfortunately, no meta-analysis or review studies have been published on the quantification of the effect of irrigation on N-leaching. Both WUE and NUE are negatively influenced in case of over-supplying water and/or nitrogen which likely results in increased leaching.

16.4.4 Controlled drainage

This section summarizes results from one meta-analysis study and three reviews on (controlled) drainage, and these pertain to environmental effects only, i.e. reduction in drain water volume, reduction in N-losses via the drainage water, and methane emission.

Artificial drainage is used in agriculture to remove excess water from poorly drained soils i) to enhance crop production and ii) to ensure trafficable conditions for field operations. With controlled drainage, adjustable head structures are used to prevent discharge when the water table is shallower than the outlet elevation. In this way discharge of water and nutrients may be reduced. Resulting changes in soil environmental factors above the drains may also have an effect on biological and chemical conversions.

Amenumey et al. (2009) performed a meta-analysis with fifty-three controlled drainage water volume reduction results selected from twenty papers published between 1979 and 2008. The effectiveness of controlled drainage varied across different soils, crops and locations: drainage volumes reductions ranged from -8% to 95%, with 50% of the results in the range 30% to 67%. The mean effect was a significant reduction of 47% (Figure 16,7; a reduction of 47% means *ES* = -47%). The decrease was highest in loamy sand soils compared to other loam soils. Except for potatoes, for which the highest drainage volume decrease was observed, there was no effect of crop type (Figure 16.7). Amenumey et al. (2009) also considered the influence of the hardiness (harshness) zone and concluded that controlled drainage was most effective for moderate climate conditions (60%; n = 13), least effective in a cold climate (23%; n = 4) and hardly varied for the other four zones (42-47%; n = 4-18).

Skaggs et al. (2010) reviewed the effectiveness of controlled drainage in terms of reduction in drainage volume and nitrogen losses via the drains (12 studies, 8 references in the period 1979-2008). Here their reported data have been analysed and summarized according to the ln-transformed Y_T/Y_C ratio (see Chapter 5). The averages and confidence interval have been back-transformed and expressed again in terms of *ES*. The thus obtained average reduction in drainage water volume was 55% (range 16% to 85%) and the reduction in N loss via the drainage water was 61% (range 18% to 85%), with both significantly different from zero (Figure 16.7). In two studies the authors reported an increase in surface runoff of 28% and 38% for the treatments with controlled drainage.

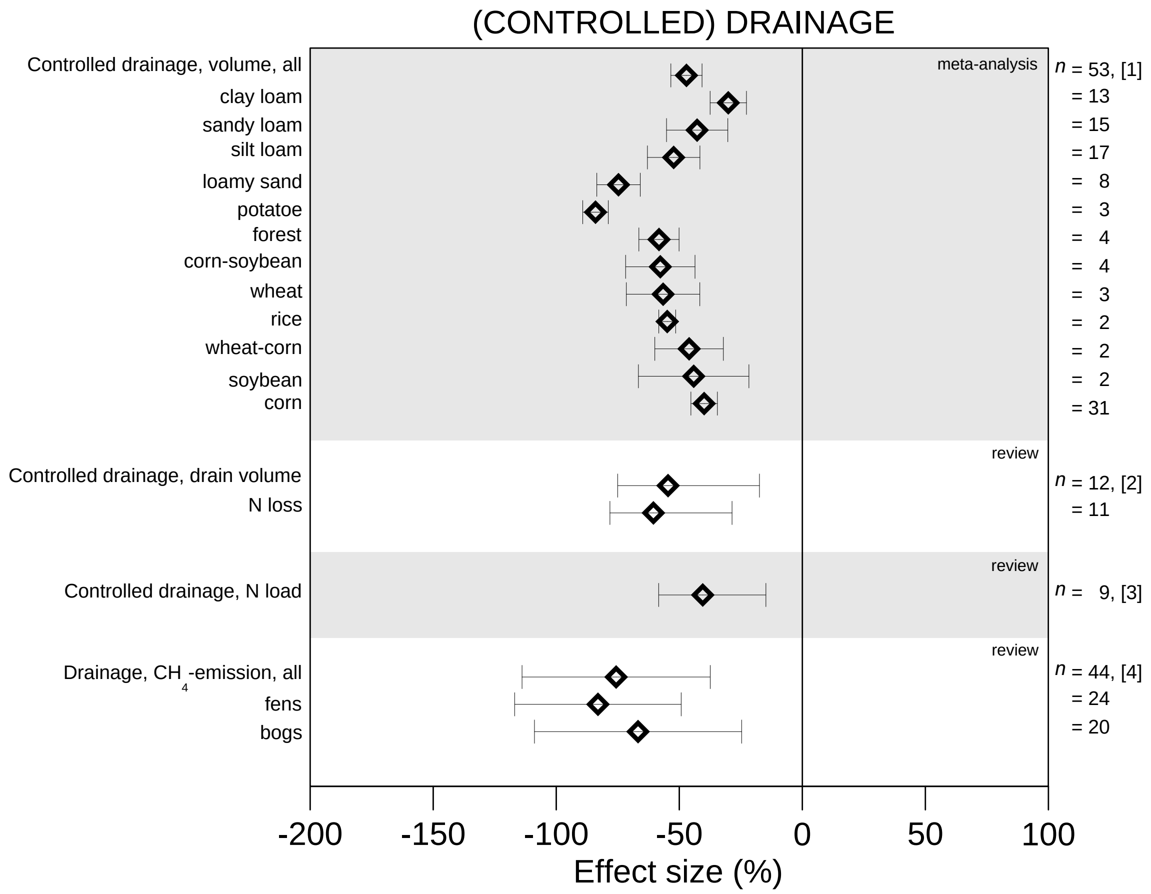


Figure 16.7. *Summary overview of various meta-analyses studies. Mean effect sizes and corresponding confidence intervals of controlled drainage ([1]: Amenumey et al., 2009) and three reviews ([2]: Skaggs et al., 2010; [3]: Christianson et al., 2013; [4]: Abdalla et al., 2016). At the left the effect studied is mentioned, and on the right hand side the number of data-pairs (n) and the link to the reference between [] is given.*

Christianson et al. (2013) reviewed, amongst others, the N-load reduction under controlled drainage for nine studies. For each study either the mean or the range was provided; here we used the means and replaced the reported ranges by the corresponding average value. Here their reported data have been analysed and summarized according to the ln-transformed Y_T/Y_C ratio. The averages and confidence interval have been back-transformed and expressed again in terms of *ES*. The thus obtained average reduction in N-load was 40% (range 15% to 75%), with both significantly different from zero (Figure 16.7).

Abdalla et al. (2016) reviewed the impact of drainage on methane (CH_4) emission from drained versus natural peat lands. Here in total 44 paired data from 22 studies could be used. Since in some cases zero or negative CH_4 fluxes were reported, we cannot analyse the results in terms of ln-transformed Y_T/Y_C ratios. Instead, here the average (± standard deviation) of *ES* of all 44 data pairs are reported. On average the CH_4 emission was (significantly) reduced by 76% (±38%) (Figure 16.7). There was not much difference between peat types fens and bogs (Figure 16.7), although the authors reported that this difference was significant ($P < 0.05$; using more

data than the paired data used here). The authors also reported that for drained peat lands the CH_4 emission differed significantly between land uses (natural > woodland/shrubs > grassland > crop land).

In summary, controlled drainage significantly reduces the volume of drainage water and the corresponding N-load. The effectiveness differs slightly between soil types and climatic zones, but is hardly dependent on crop type. Draining peat soils will reduce CH_4-emission.

16.4.5 Tillage

No-tillage and reduced tillage (NT/RT) management practices are being promoted in agroecosystems to reduce erosion, sequester additional soil C and reduce production costs.

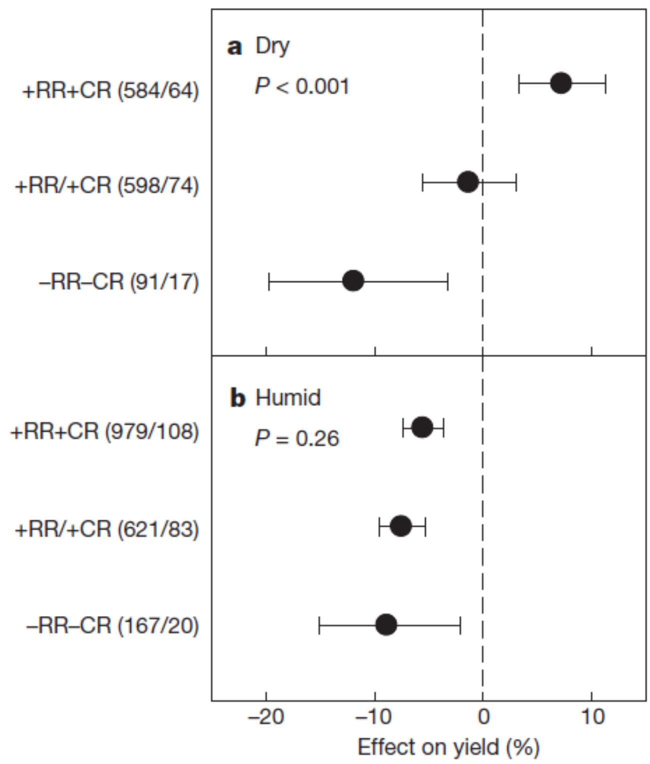


Figure 16.8. *Comparison of yield in no-till versus conventional tillage systems in relation to the other two principles of conservation agriculture under dry (upper panel) and humid conditions (lower panel); CR is crop rotation; RR is residue retention (return to the soil) (Pittelkow et al., 2015a,b).*

A global meta-analysis evaluated the influence of various crop and environmental variables on no-till relative to conventional tillage yields using 678 studies with 6005 paired observations, representing 50 crops and 63 countries (Figure 16.8; Pittelkow et al., 2015a, b). Side-by-side yield comparisons were restricted to studies comparing conventional tillage to no-till practices

in the absence of other cropping system modifications. Crop category was the most important factor influencing the overall yield response to no-till followed by aridity index, residue management, no-till duration, and N rate. No-till yields matched conventional tillage yields for oilseed, cotton, and legume crop categories. Among cereals, the negative impacts of no-till were smallest for wheat (−2.6%) and largest for rice (−7.5%) and maize (−7.6%). No-till performed best under rainfed conditions in dry climates, with yields often being equal to or higher than conventional tillage practices. Yields in the first 1–2 years following no-till implementation declined for all crops except oilseeds and cotton, but matched conventional tillage yields after 3–10 years except for maize and wheat in humid climates. Overall, no-till yields were reduced by 12% without N fertilizer addition and 4% with inorganic N addition.

Using meta-analysis, Luo et al., (2010) assessed the response of soil organic carbon (SOC) to conversion of management practice from conventional tillage (CT) to no-tillage (NT) based on global data from 69 paired-experiments, where soil sampling extended deeper than 40 cm. Conversion from CT to NT changed distribution of C in the soil profile significantly, but did not increase the total SOC except in double cropping systems. After adopting NT, soil C increased by 3.15 ± 2.42 t ha^{-1} (mean ± 95% confidence interval) in the surface 10 cm of soil, but declined by 3.30 ± 1.61 t ha^{-1} in the 20–40 cm soil layer. Overall, adopting NT did not enhance soil total C stock down to 40 cm. Increased number of crop species in rotation resulted in less C accumulation in the surface soil and greater C loss in deeper layers. Increased crop frequency seemed to have the opposite effect and significantly increased soil C by 11% in the 0–60 cm soil. Neither mean annual temperature and mean annual rainfall nor nitrogen fertilization and duration of adopting NT affected the response of soil C stock to the adoption of NT. The results highlight that the role of adopting NT in sequestrating C is greatly regulated by cropping systems. Increasing cropping frequency might be a more efficient strategy to sequester C in agro-ecosystems.

No-till (NT) farming has been proposed also as an alternative approach to conventional tillage (CT) in reducing soil phosphorus (P) export from agricultural land to aquatic ecosystems. Daryanto et al. (2017) conducted a meta-analysis to understand which variables control the concentration and load of different P fractions (dissolved P, particulate P) in agricultural runoff and leaching. In comparison with CT, particulate P loss was significantly lower with NT (45% and 55% reduction in concentration and load, respectively), but an increase in dissolved P loss was observed. In comparison with CT, NT was not effective in reducing particulate P concentration during wet years and particulate P load on steep slopes (4–9%). Total P concentration was also similar with CT at sites under prolonged NT duration (~10 yr) and at NT fields planted with soybean [Glycine max (L.) Merr.]. The presented results underscore the need to consider the covarying physical and management factors when assessing the potential of NT farming in controlling P loss in the environment. The limited impact of NT on dissolved P loss remains a serious impediment toward harnessing the water quality benefits of this management practice.

The impact of NT/RT on N_2O emissions has been variable with both increases and decreases in emissions reported (Van Kessel et al., 2013). A meta-analysis was conducted on 239 direct comparisons between conventional tillage (CT) and NT/RT. Averaged across all comparisons, NT/RT did not alter N_2O emissions compared with CT. However, NT/RT significantly reduced N_2O emissions in experiments >10 years, especially in dry climates. No significant correlation was found between soil texture and the effect of NT/RT on N_2O emissions. When fertilizer-N was placed at >5 cm depth, NT/RT significantly reduced area-scaled N_2O emissions, in particular under humid climatic conditions. Compared to CT under dry climatic conditions, yield-scaled N_2O increased significantly (57%) when NT/RT was implemented <10 years, but decreased significantly (27%) after >10 years of NT/RT. There was a significant decrease in yield-scaled N_2O emissions in humid climates when fertilizer-N was placed at >5 cm depth. Therefore, in humid climates, deep placement of fertilizer-N is recommended when implementing NT/RT. In addition, NT/RT practices need to be sustained for a prolonged time, particularly in dry climates, to become an effective mitigation strategy for reducing N_2O emissions.

In summary, soil tillage practices have diverse effects on crop productivity and sustainability. No-till practices in general decrease crop yield, save labour and fuel costs, and have diverse effects on soil carbon sequestration, N_2O emissions and P losses to surface waters, depending on site-specific conditions.

16.4.6 Pest management

This section summarizes the results from two meta-analyses and additional 17 articles on the effect of fertilizers on arthropods. Relevant literature was searched on Web of Science using the keywords "fertiliz*" + "arthropod" (search performed on 29.08.2017). The meta-analysis by Butler et al. (2012) did not appear in the initial search and was included later.

Because they significantly increase primary production, fertilizers are an essential element of current agricultural production, and of soil improving cropping systems. Many ecological models predict that increasing productivity, and thereby the yield, will be followed by an increase in biodiversity at the levels of both primary and secondary consumers. However, empirical data do not provide unequivocal supporting evidence of those predictions.

Garratt et al. (2011) performed a meta-analysis including thirteen articles (reference period not indicated) on the effect of fertilizers on pest arthropods and their natural enemies. No significant mean effect of fertilizers was found on pests. However, natural enemies benefited from organic fertilizers, with a ca. 130% increase (estimated from figure) in the mean effect size of the measured responses.

In their meta-analysis, Butler et al. (2012) considered more than 200 articles (published between 1994 and 2011) on the effect of fertilizers, mostly N-based, on herbivorous insects. Overall, fertilizers had a positive effect, with a ca. 40 % increase (estimated from figure) in the mean effect size of the measured responses (i.e. mostly herbivore abundance).

Of the 17 articles collected from WoS, thirteen showed at least some positive effect, three exclusively no effects, and a single study showed exclusively a negative effect (Table 16.4). Considered the multiple responses examined, positive effects occurred more often (55.8% of the time) than negative (9.3% of the time), or no effects (34.9% of the time, Table 16.4).

Eleven articles (64.7%) investigated the effect of N fertilizers, four (23.5%) of NPK fertilizers, one of NP fertilizers, and one the addition of N, P, K, B, Cu, Fe, Mn, Zn, and Mg fertilizers (Table 16.4). When only studies using N-fertilizers were considered, the proportion of positive, negative, and no effects did not substantially change (56.3%, 9.4%, and 34.4% of the time, respectively).

Eight studies (47.1%) measured exclusively abundance of at least one arthropod taxon; four (23.5%) measured exclusively species richness, four (23.5%) measured both abundance and species richness, and one study measured the survivorship, development time and reproduction of a single species (Table 16.4).

Only four studies (23.5%) focused on plant biomass benefit. Working in subalpine hay meadow, Andrey et al. (2014) found that fertilizers approximately doubled biomass (from 200 gm^2 to 400 gm^2; from figure). In rice fields, De Kraker et al. (2000) found that plants grew taller with increasing N-level, but grain yield was the highest at medium level of N-fertilizer application. In grassland, Haddad et al. (2000) found an increase in plant biomass (from 290 g m^{-2} with no fertilizer application, to ca. 380 g m^{-2} at 28 g m^{-2} N-application). In grassland, Siemann (1998) found a linear increase in biomass with increasing N-application only under modern (i.e. short-term N-addition) fertilization treatment (200 g m^{-2} in the control; 400 g m^{-2} with low fertilization; 620 g m^{-2} with high fertilization), but not for “historical” (long-term N-addition) fertilization rates, which reduced biomass at high N-application (387 g m^{-2} in the control; 437 g m^{-2} with low fertilization; 370 g m^{-2} with high fertilization). Precise data are typically not presented in text, and had to be estimated from figures. A possible positive effect was reported by Garratt et al. (2011), in relation to use of organic fertilizers, and the corresponding ca. 130% increase in the mean effect size of natural enemies of pest arthropods.

In conclusion, fertilization has a mostly positive effect both on primary and secondary consumers even though the response variables are not always easily comparable. The interpretation of these findings is difficult, because the focus of these (few) studies was not on biomass production. In those studies where such data are available, the increase in yield is mostly accompanied by an increase in primary and/or secondary consumers. It is possible that without herbivore pressure, the production gain would have been even greater; however, it is also plausible to assume a compensatory reaction, i.e. a mild or medium herbivore pressure triggers compensatory growth in plants, so they overperform in response. Finally, the increase in natural enemies may have the consequence of reducing the herbivore impact. The evidence base is too small to decide about these alternatives. What seems to be certain is that the

increased density or species richness of herbivores and natural enemies in response to fertilisation did not cause a drastic reduction in plant production.

Table 16.4. *Summary of the papers characterising arthropod responses to fertilisation.*

Habitat	Effect type			Response parameter	Fertilizer	Reference
	Postive	Negative	None			
Grassland	0	1	0	total insect richness	N	Haddad et al., 2000
Grassland	0	1	0	predator richness	N	Haddad et al., 2000
Grassland	0	1	0	herbivore richness	N	Haddad et al., 2000
Grassland	1	0	0	detritivore richness	N	Haddad et al., 2000
Grassland	0	0	1	parasitoid richness	N	Haddad et al., 2000
Grassland	1	0	0	arthropod abundance	N	Haddad et al., 2000
Grassland	0	0	1	parasitoid abundance	N	Haddad et al., 2000
Tropical forest	1	0	0	arthropod abundance	N, P, K, B, Cu, Fe, Mn, Zn, Mg	Yang et al., 2007
Rice field	0	0	1	predator abundance	N	Yang et al., 2016
Rice field	1	0	0	herbivore abundance	N	Yang et al., 2016
Spruce forest	1	0	0	arthropod richness	N	Behan et al., 1978
Maple orchard	1	0	0	predatory mite abundance	N	Prado et al., 2015
Apple orchard	1	0	0	rove beetle richness	NPK	Miñarro et al., 2009
Apple orchard	0	0	1	ant richness	NPK	Miñarro et al., 2009
Apple orchard	0	0	1	carabid richness	NPK	Miñarro et al., 2009
Apple orchard	0	0	1	spider richness	NPK	Miñarro et al., 2009
Grassland	1	0	0	predator richness	N	Siemann 1998
Grassland	1	0	0	herbivore richness	N	Siemann 1998
Grassland	1	0	0	detritivore richness	N	Siemann 1998
Grassland	1	0	0	parasite richness	N	Siemann 1998
Grassland	1	0	0	predator abundance	N	Siemann 1998
Grassland	1	0	0	herbivore abundance	N	Siemann 1998
Grassland	1	0	0	detritivore abundance	N	Siemann 1998
Grassland	1	0	0	parasite abunance	N	Siemann 1998
Subalpine meadow	0	0	1	arthropod richness	N	Andrey et al., 2014
Tomato crop	1	0	0	herbivore abundance	organic +NPK	Yardim et al., 2003
Tomato crop	1	0	0	predator abundance	organic +NPK	Yardim et al., 2003
Greenhouse	0	0	1	development time on *Stephanitis pyrioides* *	N	Casey & Raupp 1999
Greenhouse	0	0	1	survivorship on *S. pyrioides*	N	Casey & Raupp 1999

Habitat	Effect type Postive	Negative	None	Response parameter	Fertilizer	Reference
Greenhouse	0	0	1	reproduction on *S. pyrioides*	N	Casey & Raupp 1999
Rice field	1	0	0	herbivore abundance	N	De Kraker et al., 2000
Rice field	1	0	0	predator abundance	N	De Kraker et al., 2000
Rice field	1	0	0	parasitoid abundance	N	De Kraker et al., 2000
Spruce forest	0	0	1	arthropod richness	N	Edenius et al., 2012
Spruce forest	0	0	1	arthropod abundance	N	Edenius et al., 2012
Prairie	0	1	0	arthropod richness	NPK	Hartley et al., 2007
Wheat field	1	0	0	rove beetle density	N	Krooss & Schaefer 1998
Loess Plateau	1	0	0	arthropod abundance	NP	Lin et al., 2013
Grassland	1	0	0	spider abundance	NPK	Patrick et al., 2012
Grassland	0	0	1	spider richness	NPK	Patrick et al., 2012
Willow and cottonwood forest	0	0	1	arthropod abundance	N	Wiesenborn 2011
Willow and cottonwood forest	0	0	1	spider abundance	N	Wiesenborn 2011
Willow and cottonwood forest	1	0	0	homopteran abundance	N	Wiesenborn 2011

16.4.7 Weed management

The following section discusses long-term cost-effective weed management strategies to reduce yield loss, and improve soil health and their ability to provide ecological services.

Under conventional production systems, weed control is frequently delegated to the use of herbicides combined with intensive soil inversion tillage (Armengot et al., 2015). This approach have kept weeds to a very low level, often only a few percentages of soil cover (Kristensen & Halberg 1995; Kristensen & Hermansen 1988; 1986). Long-term data analysis of organic versus conventional production systems (Ryan et al., 2010, Posner et al., 2008 and Treadwell et al., 2007) showed that despite higher weed abundance, at least four times more in organic than in conventional, the potentials for crop yields were comparable for the two types of soil management. Smith et al. (2010) used soil Resource Pool Diversity Hypothesis (RPDH) in mediating competition for soil resources between weeds and crops. The authors compared five studies with 9 unique crop-weed competitions and found that organic yield decrease relative to conventional yield ranged from 0 to 18% when organic weed abundance was between 29 to 2000%. Despite the higher weed range in organic systems, comparatively smaller yield differences were observed, implying that crop-yield and weed abundance associations differ

between management systems, and are largely governed by the interaction of farming intensity and climate.

In organic farming settings, in general, soil management by reducing tillage intensity (e.g. no-tillage or reduced tillage), if applied properly in conjunction with crop diversification both in sequence and associations, and with complex crop rotations, provide higher synergistic effects for tackling weeds and reducing weed seedbank (Peigné et al., 2007). A long-term study by Armengot et al. (2015) investigated the effects of organic reduced tillage and contrasting crop sequence on weeds and crop yields, and concluded that crop yields were unaffected; although, weed abundance was at least doubled under reduced tillage, and favoured mostly perennials (e.g. *Convolvulus arvensis L., Taraxacum officinale* weber) over time. In addition, reduced tillage did not promote an increase in weed diversity, specifically grass species cover over years, and weed infestation appeared higher for sunflower than for cereals.

A meta-analysis by Cooper et al. (2016) indicated that with deep inversion tillage (≥ 25 cm) as control, and with 94 observation pairs from 13 studies (sample size (i.e. the number of control-treatment pairs)/number of experiments), the overall average increase in weed incidence was 54% with only shallow inversion (26 observation pairs from 5 studies) and no tillage (3 observation pairs from 2 studies) showing no increase in weed incidence. Moreover, with shallow inversion (< 25 cm) as control and with 68 observation pairs from 9 studies, the overall weed abundance was 56% with no difference in weed incidence between tillage systems (non-inversion 10-25cm – 26 observation pairs from 5 studies; non-inversion <10cm – 36 observation pairs from 6 studies and no till – 6 observation pairs from 1 study).

The authors plotted the mean effect size for yield and for weed incidence to check the relationship between weed pressure, reducing tillage intensity and organic crop yields. A negative relationship between the two parameters indicates lower yields were associated with higher weed incidence. With deep inversion tillage as control, reducing tillage by double-layer plough (P = 0.049) and shallow inversion (< 25 cm) (P = 0.023) tillage only showed associations for lower yields and higher weed incidence. Such associations were not found for non-inversion tillage at various depth. Though, deep non-inversion tillage effected greater yield reduction (11.6% to that of deep inversion tillage) there was no significant weed incidence and yield relationship. Using shallow inversion tillage as control, there was negative correlation between weed incidence and crop yields only for deep non-inversion tillage (P = 0.03). However, for shallow non-inversion tillage this relationship was non-significant. These results indicated that weed incidence may not be the only factor for organic yield reductions when reducing tillage intensity. Other factors however, like N limitation can have more influence on yields than weeds have. In addition, the study indicated that the yields were comparable under shallow non-inversion and deep inversion tillage. Shallow inversion tillage also resulted in significantly higher carbon stocks with better weed control. Similarly, the review by Govaerts et al. (2010) showed increased carbon stocks with reduced tillage at least in upper soil layers in 40 out of 78 studies.

Reviews by Anderson (2015) and Nicholas et al. (2015) indicated that crop rotation with different planting dates combined with limited soil disturbance can disrupt weed-crop associations in addition to reducing yield loss and rebuilding soil fertility. Effective crop rotation encompasses diverse crop groups such as cereals (alternative winter and spring), legumes, root crops, field vegetables and broad-leaved arable crops. A 14-year study by Buhler et al. (1994) found higher perennial weeds under reduced tillage with monocultural maize relative to two year rotation of maize and soybean. Daugovish et al. (1999) demonstrated that downy brome and jointed goatgrass weeds were completely eliminated, when maize or sunflower was included into winter wheat-fallow rotation. From an ecological viewpoint, perennial weeds favour undisturbed or less disturbed soils. But employing appropriate crop sequence patterns within crop rotation can potentially combat perennial weeds (Nicholas et al., 2015). Rotation with allelopathic crops (e.g. oat, barely, brassica, mustard) can also be an alternative weed control option (Nicholas et al., 2015). Polycultures need not be necessarily limited only to crop species. Catch crops, cover crops, shrubs and trees can all be effective tactics to reduce weeds from increasing to severe infestation levels. Inserting legumes or green manures in particular, outcompete unwanted weeds, in addition to BNF (Biological Nitrogen Fixation; Verret et al., 2017). A review by Liebman & Dyck (1993) indicated that weed biomass was reduced in 87-90% of cases when the main crop is intercropped with a "smother" crop. Similarly, a recent meta-analysis study by Verret et al. (2017) estimated the effects of different companion crops (26 legume species) (as living mulch, synchronised sowing or relay-intercropping) on regulating weeds and annual cash crop yields (15 crops) from 34 scientific articles, corresponding to 476 experimental units. Intercropping improved weed control and increased crop yields in 52% of experimental units compared to non-weeded treatments and 36% experimental units compared to weeded control. The meta-analysis detected that companion crops on suppressing weeds were highest in non-weeding, compared to weeding. The greatest benefits of intercropping was observed in maize, with yields increasing by 37% compared to non-weeded controls.

Other favoured cultural tactics for optimising weed suppression and enhancing crop yields include crop competition. Hucl (1998) found yield-gain of 7 to 9% from competitive over non-competitive spring wheat genotypes under weedy and partially-weedy situations. Murphy et al. (2008) evaluated 63 historical and modern spring wheat cultivars and found that the top five ranked cultivars for weed competitive ability had lower weed mass per plot by 573% than the bottom five ranked. Stahlman & Miller (1990) found that winter wheat showed more weed tolerance without yield decrease, if downy brome (*Bromus tectorum*) emerged 21 days after wheat, as compared to synchronized emergence. O'Donovan et al. (1985) demonstrated spring wheat weed tolerance relative to later emerging wild oats (*Avena sp.*). Nevertheless, cereal cultivars are genetically diverse in their capacity to compete with weeds, and crop-weed interactive response largely depend on environmental factors (Andrew et al., 2015). Additionally, increasing within-crop diversity (e.g. mixtures and populations) can effectively restrict weed species fitness with additional benefits of yield stability, adaptability to climatic

variation and allowing more heterogeneity in soil fertility (Wolfe et al., 2008). Wu et al. (2000) screened cereal cultivars for allelopathic potential and observed ryegrass root growth inhibition by 24 to 91% among 452 cultivars within wheat crop. A meta-analysis study by Li et al. (2016) indicated that cultivars (e.g. wheat) that had ability to form active mycorrhizal associations directly suppressed weak host weeds (e.g. *Brassica sp.*) or indirectly suppressed the growth of strong host weeds by improving strong host crop suppressive ability.

Reviews by Andrew et al. (2015) and Nicholas et al. (2015) reported increased seeding density of fast growing wheat, barley and rice cultivars achieved concurrently higher yields and lower weed biomass. Mohler (2001) demonstrated decrease in crop row-spacing reduced weed pressure with or without yield advantage. Planting arrangements like uniform crop spacing, optimal seeding depth together with increased seeding density significantly reduced weed competition and improved yields in a variety of crops (Nicholas et al., 2015; Olsen et al., 2005). Weiner et al. (2001) found 30% lower weed density and increased grain yield of 9% in grid-pattern sown wheat with spacing between rows and plants of 4 cm and 2.5 cm. Manipulating drilling dates can provide asymmetric advantage to the crop over weeds, if the lifecycle of weed species is known. Early planting in rice-wheat systems allows crop to establish before *Phalaris minor* and thus reduced yield loss (Nicholas et al., 2015). Contrastingly, delayed planting allows use of stale seedbeds to control weeds before sowing, therefore less weed pressure of some weed species during growth stages of winter wheat and barley cultivars (Andrew et al., 2015). Mechanical weeding (harrowing or inter-row hoeing) can be effective, but is usually dependent on fossil fuels and can lead to soil degradation if practiced frequently over longer-term (Holland 2004).

Crop-weed competition also depend on type, timing and application method of fertilizers. Menalled et al. (2005) showed composted swine manure applied to the soils reduced foxtail (*Setaria faberi* Herrm), velvet leaf (*Abutilon theophrasti* Medik.) and common waterhemp (*Amaranthus rudis* Sauer) seedling emergence by 57%, 23% and 76% respectively. Agenbag & Villiers (1989) reported dormancy break of certain weed species with increased N. From Rothamsted long-term field study Moss et al. (2004) demonstrated that response of weed species differ with increasing synthetic nitrogen (N) levels. Forcella (1984) found greater ryegrass (*Lolium rigidum*) suppression by wheat, when N was applied before 3-leaf stages than later. For spring-sown wheat, barley, canola and peas, yield increase and reduced weed pressure was frequently achieved with spring compared to autumn-applied fertilizers (Blackshaw et al., 2004). Deep fertilizer banding reduced weed competitive ability than surface banding or broadcasting (Rasmussen et al., 1996; Nicholas et al., 2015). Several soil microorganisms including fungi, bacteria and actinomycetes have demonstrated weed control ability and exploring microbial bio-controls can be an attractive option to improve soil quality and limiting yield loss (Nicholas et al., 2015).

In summary, from a SICS viewpoint, effective weed management should involve a multitude of preventive tactics to reduce the persistence of seeds in the soil and avoid weeds from being

recurrently successful. Cultural tactics mainly include stabilising crop growth and weed suppressive ability to reduce yield loss and limit weed infestation. Table 16.5. lists cropping system indicators on weeds, soil, yields and profitability.

Table 16.5. *Cropping system indicators on weeds, soil, yields and profitability.*

Components	Weed infestation levels	Crop yields	Enhancing soil quality and ecological services	Profitability
Organic or low-input	+/-	+/-	++	+
Reduced tillage + cover crops + crop rotation	- -	+/-	++	+
Competitive cultivars	-	+	++	+
Crop spacing + seeding density + seed spacing	-	+	++	+
N applications	+/-	+	+/-	+/-

16.4.8 Residue management

Soil mulching (with plastic or straw) reduces evaporation, modifies soil temperature and thereby affects crop yields. Reported effects of mulching are sometimes contradictory, likely due to differences in climatic conditions, soil characteristics, crop species, and also water and nitrogen (N) input levels. Qin et al. (2015) reported on a meta-analysis of the effects of mulching on wheat and maize, using 1310 yield observations from 74 studies conducted in 19 countries. The results indicate that mulching significantly increased yields, WUE (yield per unit water) and NUE (yield per unit N) by up to 60%, compared with no-mulching (Figure 16.9). Effects were larger for maize than for wheat, and larger for plastic mulching than for straw mulching. Interestingly, plastic mulching performed better at relatively low temperature while straw mulching showed the opposite trend. Effects of mulching also tended to decrease with increasing water input. Mulching effects were not related to soil organic matter content. In conclusion, soil mulching can significantly increase maize and wheat yields, WUE and NUE, and thereby may contribute to closing the yield gap between attainable and actual yields, especially in dryland and low nutrient input agriculture. The management of soil mulching requires site-specific knowledge.

Pittelkow et al. (2015b) mentioned that no-till in combination with residue return (mulching) and crop rotation significantly increases rainfed crop productivity in dry climates. This suggests that soil mulching may become an important climate-change adaptation strategy for ever-drier regions of the world.

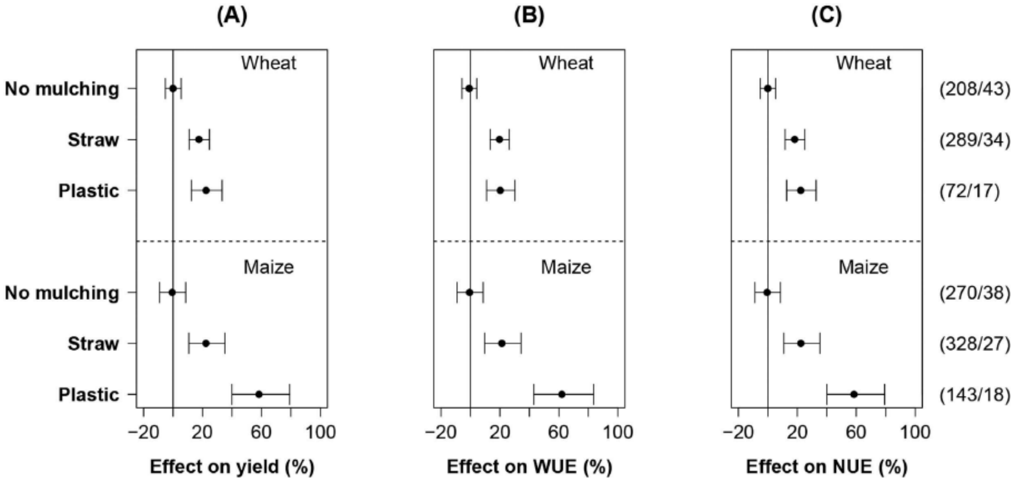


Figure 16.9. *Effect of mulching on crop yield (A), water use efficiency (B) and nitrogen use efficiency (C) of wheat (upper panels) and maize (lower panels). Dots show means, error bars represent 95% confidence intervals. The number of observations and total number of studies for each treatment are displayed in parentheses on the right-hand side of the figure, respectively.*

16.4.9 Landscape management

This section summarizes the results of two meta-analyses and thirteen additional research articles. Landscape management from a strictly agricultural point of view mostly concerns the field level and not landscape features like steep hills or marshland unsuitable for farming. Historical sites and buildings are usually also not considered and neither are societal drivers of land-use change or farmers' motivations. The focus tends to be on the part of the landscape that is covered by the mosaic of arable fields and pastures as determined by physical elements. Elements of interest can be woodland, herbaceous field margins or field margins determined by historical boundaries like drystone walls, hedgerows and roads as well as streams and possible associated buffer strips.

The reviewed landscape elements all have a direct influence on farming practices and associated nutrient flows as well as field specific microclimate and biodiversity. Together, they influence soil development and quality and ecosystem services like carbon sequestration, good water quality and, of course, crop yield.

The landscape elements a farmer has most control over and that at the same time also have the most impact on soil and yield include field margins in the form of hedgerows and shelterbelts. Shelterbelts and hedgerows have been prominent parts of the agricultural landscape for a long time and have played an important role in shaping the properties of many agricultural soils. According to Follain et al. (2009), the present soil organization cannot simply be explained by local geography or today's field margin layout. Even long vanished manmade landscape structures are still noticeable in their effect on field soil distribution and dynamics.

As an example from Denmark for how early shelterbelts were utilised to prevent soil degradation and crop damage by wind erosion, organized planting began there as early as 1874 (Veihe et al., 2003). After a devastating storm in 1938, the government stepped in and approximately 43000 km of shelterbelts were established over the following thirty years.

In a meta-analysis of 120 articles, Kort (1988) summarized the effect of shelterbelts on wind erosion. Shelterbelts reduce wind speeds, thereby limiting soil erosion and sandblast damage to crops to varying degrees depending among other things on structure height and time of year. These effects are the primary driver for shelterbelt establishment in many parts of the world. In confirmation of this general assumption, Sauer et al. (2007) found a marked increase in silt deposition on the leeward side of a shelterbelt.

In Europe, the risk of wind erosion is greatest in Lower Saxony, The Netherlands, western Denmark, southern Sweden and the east and southeast of England as well as northeastern Spain (Riksen et al., 2003). The loss of fertile topsoil leads to yield reduction and long term soil degradation. Soils with poor moisture retaining capability and low organic matter content are the most vulnerable. Hedgerow removal and subsequent increase in wind speeds has been shown to dramatically increase wind erosion risk under the right meteorological circumstances (Fullen, 1983). Veihe et al. (2003) found that the erosion risk in western Denmark was greatest at the time of sowing, with fine particles of organic matter making up a disproportionate amount of the eroded material.

Since tree roots act as a fast path for water to deeper soil layers (Liang et al., 2009), runoff from heavy rainfall and associated erosion from hillslopes can be greatly reduced by contour shelterbelts. A few years after planting, the shelterbelts develop an up to 60% higher infiltration rate than the adjacent field or pasture, acting as a sink for runoff and a filter for sediment (Carroll et al., 2004).

Follain et al. (2009) summarized the findings of six research articles that on sloping land, organo-mineral soil horizons thicken uphill from hedges with the accumulated soil thickness proportional to the uphill drainage area. They showed that soil horizon geometry on the slope scale is clearly influenced by these structures, even where they have been removed a long time ago. Deeper, mineral soil horizons don't seem to be affected.

In relation to Soil organic matter (SOM) and soil organic carbon (SOC), hedgerows have a substantial effect on the amount and distribution of organic carbon in field soils. Even though SOC distribution close to hedges is very variable, overall SOC reliably decreases the further away from the structure the sample is taken (Follain et al., 2007). C/N ratio is also highest close to a hedgerow and the absolute amount is positively correlated with the density of trees in the structure. Increased input of organic matter increases SOM which leads to increased mineralisation and activation of soil biofauna, thereby increasing SOC. On sloping land this effect is increased due to soil accumulation on the uphill side of the hedgerow that leads to an

increased A-horizon thickness. Sauer et al. (2007) found that SOC content in the top 15 cm of soil in a shelterbelt on average was about 370 g m^{-2} higher than in the adjacent cultivated field.

Shelter affects the micro climate and improves growing conditions by better preservation of soil moisture and lower evaporation rates as well as lessening temperature extremes (Kort, 1988). Spring melting of large amounts of trapped snow can both be beneficial as well as detrimental by preserving winter moisture on dry soils or causing waterlogged conditions on heavy soils.

In a meta-analysis of 26 articles, Kuemmel (2003) found that field margins in general have yield depressions, regardless of whether a hedge is present or not. Average yield depressions ranged from 7 to 45% in cereals, 10% for potatoes and 26% for sugar beet. Shading appeared to be the main source of a drop in yield, but competition for nutrients and damage from pests also played a role.

Competition from weeds spreading into the crop from the hedge is limited if it happens at all. The weed flora of arable land tends to be largely unrelated to the plant species found at hedge bottoms. (Marshall and Arnold, 1995). Seed dispersal from the field into the margin is more likely than the other way around (Devlaeminck et al., 2005). Field margin vegetation should be left undisturbed as their predominantly perennial species hinder development of annual weed species that may invade the field (Moonen and Marshall, 2001). In summary, the meta-analysis by Kort (1998) paints an overall positive figure of the actual effect of shelter on yield (Table 16.6).

Table 16.6. *Relative responsiveness of various crops to shelter (Kort, 1988). Meta-analysis of effect of shelter on yield. No. of field-years is the total of growing seasons covered (120 studies).*

Crop	No. of field-years	Weighted mean yield increase (%)
Spring wheat	190	8
Winter wheat	131	23
Barley	30	25
Oats	48	6
Rye	39	19
Millet	18	44
Corn	209	12
Alfalfa	3	99
Hay (grass - legumes mix)	14	20

The effect of shelter on yield is a function of the hedge height (h) and the distance from the hedge expressed in multiples of h (Figure 16.10). Van Vooren et al. (2016) found that in a distance of 0 to 1.64 h the microclimate and competition is detrimental to crop growth.

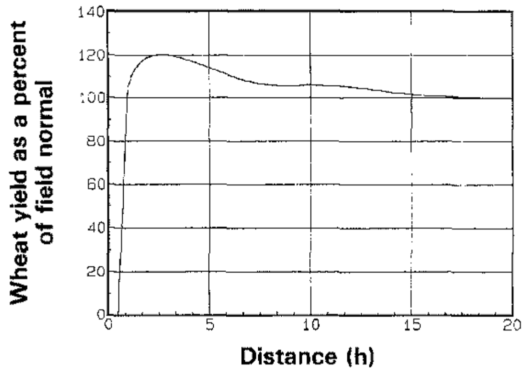


Figure 16.10. *Wheat yield as a percent of field normal. Positive effect on yield in this example is greatest at a distance of 2-3 times hedge height (h) and peters out at between 15 and 20 h (Kort, 1988). Reproduced from Agriculture, Ecosystems and Environment.*

Structures such as shelterbelts and hedgerows occupy land, thereby reducing the available arable field area. As such, they have an inbuilt yield reduction that any positive effect has to outweigh if the system is to make economic sense. Establishment and maintenance is another cost factor. Compared to the business as usual option without these structures, their establishment outside areas with very high wind erosion risk only becomes viable with subsidies like basic and greening payments (Van Vooren et al., 2016).

16.5 Conclusions

Soil improving cropping systems (SICS) are "*cropping systems that improve soil quality (and hence its functions), prevent and/or minimize soil threats, and have positive impacts on the profitability and sustainability of cropping systems*". The SICS concept is rather new, and in this report SICS have been addressed from the viewpoint of soil-threat specific SICS (Chapters 6-15), and those that contribute to a general improvement of soil functioning (this chapter).

Soil improving cropping systems (SICS) are a combination of crop rotations and 9 agro-management techniques: 1) nutrient management, 2) irrigation and fertigation, 3) controlled drainage, 4) tillage, 5) pest management, 6) weed management, 7) residue management, 8) mechanization management, and 9) landscape management.

Based on published meta-analyses, reviews and own reviews information on the effect of crop rotations and the 9 agro-management techniques could in some cases be quantified. As always in agricultural research, such outcomes are highly influenced by temporal and spatial variability, and are also influenced by interaction effects (e.g. nutrient use efficiency is highly influenced by soil water dynamics). This partly explains why not always significant effects have been reported or follow from a meta-analysis. Nevertheless, there are often clear trends visible in the outcome. Therefore, it was decided to summarize these findings in a more qualitative

manner, where SICS are assessed for 1) crop yield limiting and other reducing factors (water delivery, nutrient delivery, control of pathogens, improving structure, control of pollutants, improving SOM) (see Table 16.1), and 2) aspects of cropping system sustainability (crop yield & quality, soil quality, farm income, resource use efficiency, environmental impacts) (see Table 16.2). Based on these quantitative and qualitative scorings of SICS it can be concluded that there are enough ways to improve the soil quality (or soil health) if proper crop, crop rotations, and agro-management techniques are chosen.

The list of promising SICS are formulated in a rather general manner, mainly because SICS are site-specific and the crop rotations and agro-management techniques have to be optimized and integrated for site and farm specific conditions. The SICS concept presented here basically is a tool box of crop types, crop rotations and agro-management techniques. Depending on the local/regional environmental and socio-economic conditions, the farmer (with or without advisors) will select the appropriate combinations of crop types, crop rotations and agro-management techniques. The effectiveness of the selected combinations has to be assessed on the basis of monitoring programs of profitability, sustainability and soil quality indicators. Since the concept of SICS is not yet fully worked out, the following table lists some recommendations for scientists, stakeholders and policy to take into consideration in further developing the concept of SICS.

In order to make a proper choice of which SICS to use best under what conditions, this topic is further elaborated in Chapter 17.

16.6 References

Abdalla, M., A. Hastings, J. Truu, M. Espenberg, Ü. Mander, and P. Smith. 2016. Emissions of methane from northern peatlands: a review of management impacts and implications for future management options. Ecology and Evolution 6 (19): 7080-7102.

Agenbag, B.A. & Villiers, O.T. 1989. The effect of nitrogen fertilizers on the germination and seedling emergence of wild oat (Avena fatua l.) seed in different soil types. Weed Research 29, 239-245.

Amenumey, S., G. Sands, B. Wilson, D. Mulla, J. Nieber, and J. Swenson. 2009. Meta-analysis as a statistical tool for evaluating the hydrologic effects of water table management. Paper read at American Society of Agricultural and Biological Engineers Annual International Meeting 2009, ASABE 2009.

Amossé, C., Jeuffroy, M.H., Mary, B. & David, C. 2014. Contribution of relay intercropping with legume cover crops on nitrogen dynamics in organic grain systems. Nutr. Cycl. Agroecosyst. 98, 1-14.

Anderson, R.L. 2015. Integrating a complex rotation with no-till improves weed management in organic farming: A review. Agron. Sustain. Dev. 35, 967-974.

Andert, S., Bürger, J., Stein, S., Gerowitt, B., 2016. The influence of crop sequence on fungicide and herbicide use intensities in North German arable farming. European Journal of Agronomy 77, 81-89.

Andrew, I.K.S. Storkey, J, & Sparkes, D. L 2015. A review of the potential for competitive cereal cultivars as a tool in integrated weed management. Weed Research 55, 239–248

Andrey, A., J.Y. Humbert, C. Pernollet, R Arlettaz. 2014. Experimental evidence for the immediate impact of fertilization and irrigation upon the plant and invertebrate communities of mountain grasslands. Ecology and Evolution 4: 2610-2623.

Angus, J.F., Kirkegaard, J.A., Hunt, J.R., Ryan, M.H., Ohlander, L., Peoples, M.B., 2015. Break crops and rotations for wheat. Crop and Pasture Science 66(6), 523-552.

Armengot, L. Berner, A. Blanco-Moreno, J.M. Mader, P. & Sans F.X. 2015. Long-term feasibility of reduced tillage in organic farming. Agron Sustain. Dev. 35, 339-346.

Aronsson, H., Hansen, E.M., Thomsen, I.K., Liu, J., Øgaard, A.F., Känkänen, H. & Ulén, B. 2016. The ability of cover crops to reduce nitrogen and phosphorus losses from arable land in southern Scandinavia and Finland. Journal of Soil and Water Conservation 71 (1), 41-55.

Aronsson, H., Liu, J., Ekre, E., Torstensson, G. & Salomon, E. 2014. Effects of pig and dairy slurry application on N and P leaching from crop rotations with spring cereals and forage leys. Nutrient Cycling in Agroecosystems 98:281-293.

Bargaz A., Noyce, G.L., Fulthorpe, R., Carlsson, G., Furze, J.R. Jensen, E.K., Dhiba, D. & Issac, M.E. 2017. Species interactions enhance root allocation, microbial diversity and P acquisition in intercropped wheat and soybean under P deficiency. Applied Soil Ecology 120, 179–188

Bedoussac, L., Journet, E.P., Hauggaard-Nielsen, H., Naudin, C., Corre-Hellou, G., Jensen, E.S., Prieur, L., Justes, E. 2015. Ecological principles underlying the increase of productivity achieved by cereal-grain legume intercrops in organic farming. A review. Agronomy for Sustainable Development 35(3), 911-935.

Behan, V. M., S. B. Hill, and D. K. M. Kevan. 1978. Effects of nitrogen fertilizers, as urea, on Acarina and other arthropods in Quebec black spruce humus. Pedobiologia: 249-263.

Bergkvist, G., Stenberg, M., Wetterlind, J., Båth, B. & Elfstrand, S. 2011. Clover crops under-sown in winter wheat increase yield of subsequent spring barley—Effect of N dose and companion grass. Field Crops Research 120:292-298.

Bergstrom, L. & Kirchmann, H. 2004. Leaching and crop uptake of nitrogen from nitrogen-15-labeled green manures and ammonium nitrate. Journal of Environmental Quality 33, 1786-1792.

Blackshaw, R.E. Molnar, L.J. & Janzen, H.H. 2004. Nitrogen fertilizer timing and application method affect weed growth and competition with spring wheat. Weed Science 52, 614-622.

Bowles, T.M., Jackson, L.E., Loeher, M., Cavagnaro, T.R. 2016. Ecological intensification and arbuscular mycorrhizas: a meta-analysis of tillage and cover crop effects. Journal of Applied Ecology, doi: 10.1111/1365-2664.12815.

Buhler, D.D., Stoltenberg, D.E. Becker, R.L. & Gunsolus, J.L. 1994. Perennial weed populations after 14 years of variable tillage and cropping practices. Weed Sci.42, 205–209.

Butler, J., M. P. D. Garratt, and S. R. Leather. 2012. Fertilisers and insect herbivores: a meta-analysis. Annals of Applied Biology 161: 223-233.

Carroll, Z.L., Bird, S.B., Emmett, B.A., Reynolds, B., Sinclair, F.L. 2004. Can tree shelterbelts on agricultural land reduce flood risk? Soil Use and Management 20, 357-359

Casey, C. A., and M. J. Raupp. 1999. Supplemental nitrogen fertilization of containerized azalea does not affect performance of azalea lace bug (Heteroptera: Tingidae). Environmental Entomology 28: 998-1003.

Cayuela, M.L., E. Aguilera, A. Sanz-Cobena, D.C. Adams, D. Abalos, L. Barton, R. Ryals, W.L. Silver, M.A. Alfaro, V.A. Pappa, P. Smith, J. Garnier, G. Billen, L. Bouwman, A. Bondeau, and L. Lassaletta. 2016. Direct nitrous oxide emissions in Mediterranean climate cropping systems: Emission factors based on a meta-analysis of available measurement data. Agriculture, Ecosystems and Environment 238: 25–35.

Chardon, W.J. & Schoumans, O.F. 2007. Soil texture effects on the transport of phosphorus from agricultural land in river deltas of Northern Belgium, The Netherlands and North-West Germany. Soil Use Manage 23,16–24.

Christianson, L., J. Tyndall, and M. Helmers. 2013. Financial comparison of seven nitrate reduction strategies for Midwestern agricultural drainage. Water Resources and Economics 2-3: 30-56.

Cong, W.F., Hoffland, E., Li, L., Janssen, B.H., van der Werf, W. 2015a. Intercropping affects the rate of decomposition of soil organic matter and root litter. Plant and Soil 391(1-2), 399-411.

Cong, W.F., Hoffland, E., Li, L., Six, J., Sun, J.H., Bao, X.G., Zhang, F.S., Van Der Werf, W. 2015b. Intercropping enhances soil carbon and nitrogen. Global Change Biology 21(4), 1715-1726.

Constantin, J., Mary, B., Laurent, F., Aubrion, G., Fontaine, A., Kerveillant, P. & Beaudoin, N. 2010. Effects of catch crops, no till and reduced nitrogen fertilization on nitrogen leaching and balance in three long-term experiments. Agric. Ecosyst. Environ. 135, 268–278.

Cooper J., M. Baranski, G. Stewart, M. Nobel-de Lange, P. Bàrberi, A. Fließbach, J. Peigné, A. Berner, C. Brock, M. Casagrande, O. Crowley, C. David, A. De Vliegher, T.F. Döring, A. Dupont, M. Entz, M. Grosse, T. Haase, C. Halde, V. Hammerl, H. Huiting, G. Leithold, M. Messmer, M. Schloter, W. Sukkel, M.G.A. van der Heijden, K. Willekens, R. Wittwer, Paul Mäder 2016. Shallow non-inversion tillage in organic farming maintains crop yields and increase soil C stocks: a meta-analysis. Agron. Sustain. Dev. 36, 22.

Dalgaard, T., Bienkowski, J.F., Bleeker, A., Dragosits, U., Drouet, J.L., Durand, P., Frumau, A., Kedziora, A., Magliulo, V., Olesen, J.E., Theobald, M.R., Maury, O., Akkal, N. & Cellier, P. 2012. Farm nitrogen balances in six European landscapes as an indicator for nitrogen losses and basis for improved management. Biogeosciences 9, 5303–5321.

Daryanto, S., L. Wang, and P.A. Jacinthe. 2017. Meta-Analysis of Phosphorus Loss from No-Till Soils. Journal of Environmental Quality 46, 1028–1037. doi:10.2134/jeq2017.03.0121

Daugovish, O. Lyon, D.J. & Baltensperger D.D. 1999. Cropping systems to control winter annual grasses in winter wheat (Triticum aestivum). Weed Technology 13, 120-126.

De Kraker, J., R. Rabbinge, A. van Huis, J.C. van Lenteren & K.L. Heong. 2000. Impact of nitrogenous-fertilization on the population dynamics and natural control of rice leaffolders (Lep.: Pyralidae). International Journal of Pest Management 46: 225-235.

De Wit, C.T. 1992. Resource use efficiency in agriculture. Agricultural Systems, 1992, vol. 40, issue 1-3, pages 125-151.

Devlaeminck, R., Bossuyt, B., Hermy, M. 2005. Seed dispersal from a forest into adjacent cropland. Agriculture, Ecosystems and Environment 107, 57-64

Edenius, L., G. Mikusinski, J. Witzell, J. Bergh. 2012. Effects of repeated fertilization of young Norway spruce on foliar phenolics and arthropods: implications for insectivorous birds' food resources. Forest Ecology and Management 277: 38-45.

Evans L.T. and R.A. Fisher. 1999. Yield Potential: Its Definition, Measurement, and Significance. doi:10.2135/cropsci1999.3961544x

Finn, J.A., Kirwan, L., Connolly, J., Sebastià, M.T., Helgadottir, A., Baadshaug, O.H., Bélanger, G., Black, A., Brophy, C., Collins, R.P., Čop, J., Dalmannsdóttir, S., Delgado, I., Elgersma, A., Fothergill, M., Frankow-Lindberg, B.E., Ghesquiere, A., Golinska, B., Golinski, P., Grieu, P., Gustavsson, A.M., Höglind, M., Huguenin-Elie, O., Jørgensen, M., Kadziuliene, Z., Kurki, P., Llurba, R., Lunnan, T., Porqueddu, C., Suter, M., Thumm, U., Lüscher, A. 2013. Ecosystem function enhanced by combining four functional types of plant species in intensively managed grassland mixtures: A 3-year continental-scale field experiment. Journal of Applied Ecology 50(2), 365-375.

Follain, S., Walter, C., Bonté, P., Marguerie, D., Lefevre, I. 2009. A-horizon dynamics in a historical hedged landscape. Geoderma 150, 334-343

Follain, S., Walter, C., Legout, A., Lemercier, B., Dutin, G. 2007. Induced effects of hedgerow networks on soil organic carbon storage within an agricultural landscape. Geoderma 142, 80-95

Follett, R.F. & Hatfield, J.L. 2001. Nitrogen in the environment: sources, problems, and management. In Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection: Proceedings of the 2nd International Nitrogen Conference on Science and Policy. TheScientificWorld 1(S2), 920–926.

Forcella, F. 1984. Wheat and ryegrass competition for pulses of mineral nitrogen. Australian Journal of Experimental Agriculture and Animal Husbandry 24, 421- 425.

Fullen, M.A. 1983. Wind erosion of arable soils in east Shropshire (England) during spring 1983. Catena 12, 111-120

Garratt, M.P.D., D.J. Wright, and S.R. Leather. 2011. The effects of farming system and fertilisers on pests and natural enemies: a synthesis of current research. Agriculture, Ecosystems & Environment 141: 261-270.

Govaerts B., N. Verhulst, A. Castellanos-Navarrete, K.D. Sayre, J. Dixon & L. Dendooven. 2010. Conservation Agriculture and Soil Carbon Sequestration: Between Myth and Farmer Reality. Critical Reviews in Plant Sciences 28 (3), 97-122

Grant, C.A. & Lafond, G.P. 1994. The effects of tillage systems and crop rotations on soil chemical properties of a black chernozemic soil. Canadian Journal of Soil Science 74, 301-306.

Haddad, N. M., J. Haarstad, and D. Tilman. 2000. The effects of long-term nitrogen loading on grassland insect communities. Oecologia 124: 73-84.

Hansen, E.M. & Djurhuus, J. 1997. Nitrate leaching as influenced by soil tillage and catch crop. Soil & Tillage Research, 41, 203–219

Hansen, E.M., Eriksen, J. & Vinther, F.P. 2007. Catch crop strategy and nitrate leaching following grazed grass-clover. Soil Use and Management 23, 348-358

Hartley, M.K., W.E. Rogers, E. Siemann and J. Grace. 2007. Responses of prairie arthropod communities to fire and fertilizer: balancing plant and arthropod conservation. The American Midland Naturalist 157: 92-105.

Hauggaard-Nielsen, H., Ambus, P. & Jensen, E.S. 2001. Interspecific competition, N use and interference with weeds in pea–barley intercropping. Field Crops Res. 70, 101–109.

Hauggaard-Nielsen, H., Gooding, M., Ambus, P., G. Corre-Hellou, Y. Crozat, C. Dahlmann, A. Dibet, P. von Fragstein, A. Pristeri, M. Monti, E.S. Jensen. 2009. Pea-barley intercropping for efficient symbiotic N2-fixation: soil N acquisition and use of other nutrients in European organic cropping systems. Field Crops Res. 113, 64–71.

Hedlund K, Jørgensen HB, Jackson LE, Kätterer T, Lugato E, Thomsen IK, Isberg P, Haddaway N 2016. SR4 Database, included articles. See: http://www.eviem.se/en/projects/Soil-organic-carbon-stocks/ and the direct link to download at: http://www.eviem.se/Documents/projekt/2015/Additional%20File%207%20Systematic%20Map%20Database.xlsx

Hofman, G., Van Meirvenne, M. & Demyttenaere, P. 1992. Stikstofbemesting. Toediening van N-meststoffen in de rij: potenti:ele voordelen. Landbouwtijdschrift, 45, 341-353.

Holland, J.M. 2004. The environmental consequences of adopting conservation tillage in Europe: reviewing the evidence. Agriculture Ecosystems & Environment 103, 1-25.

Hucl, P. 1998. Response to weed control by four spring wheat genotypes differing in competitive ability, Canadian Journal of Plant Science 78, 171–173.

Ivarsson, J., Gunnarsson, A., Hansson, E., Fogelfors, H., Folkesson, 0., Lundkvist, A.& Anderson, I.L. 2001. Forsok med konventionella och ekologiska odlingsformcr 1987- 1998. Meddelande frdn sodra jordbruksforsoksdistriktet 53.

Jagannath, P., H. Pullabhotla, and N. Uphoff. 2013. Meta-analysis evaluating water use, water saving, and water productivity in irrigated production of rice with SRI vs. standard management methods. Taiwan Water Conservancy 61 (4): 14-49.

Känkänen, H. & Eriksson, C. 2007. Effect of undersown crops on soil mineral N and grain yield of spring barley. Eur. J. Agron. 27, 25–34.

Känkänen, H., Kangas, A., Mela, T., Nikunen, U., Tuuri, H. & Vuorinen, M. 1998. Timing incorporation of different green manure crops to minimize the 63-74. risk of nitrogen leaching. Agricultural and Food Science in Finland 7, 553-567.

Karlsson-Strese, E. M., Rydberg, I., Becker, H.C. & Umaerus, M. 1998. Strategy for catch crop development II. Screening of species undersown in spring barley (Hordeum vulgare L.) with respect to catch crop growth and grain yield. Acta Agric. Scand. 48, 26–33.

Kort, J., 1988. Benefits of windbreaks to field and forage crops. Agriculture, Ecosystems and Environment 22/23, 165-190

Kristensen, I.S. & Halberg, N. 1995. Markens nettoudbytte, næringsstofforsyning og afgrødetilstand på økologiske og konventionelle kvægbrug. (ed. E.S. Kristensen, Økologisk landbrug med udgangspunkt i kvægbedriften.). Intern Rapport 42, 33-49.

Kristensen, I.S. & Hermansen, J.E. 1986. Handlingsprogram for gennemførelse af høst og konservering af græs og helsæd til ensilage. se http://web.agrsci.dk/pub/sh_beretning_615.pdf Statens Husdyrbrugsforsøg, Beretning 615 (Kap 14), 159-198

Kristensen, I.S. & Hermansen, J.E. 1988. Handlingsprogram for produktion af bederoer. Udvikling, analyse og anvendelse. Statens Husdyrbrugsforsøg.Beretning. 641, 1-115.

Krooss, S., and M. Schaefer. 1998. The effect of different farming systems on epigeic arthropods: a five-year study on the rove beetle fauna (Coleoptera: Staphylinidae) of winter wheat. Agriculture, Ecosystems & Environment 69: 121-133.

Kuemmel, B. 2003. Theoretical investigation of the effects of field margin and hedges on crop yields. Agriculture, Ecosystems and Environment 95, 387-392

Lamm, F.R. 2016. Cotton, tomato, corn, and onion production with subsurface drip irrigation: A review. Transactions of the ASABE 59 (1): 263-278.

Langeveld, J.WA., Verhagen, A., Neeteson, J.J., van Keulen, H., Conijn, J.G., Schils, R.LM. & Oenema, J. 2007. Evaluating farm performance using agrienvironmental indicators: recent experiences for nitrogen management in The Netherlands. J. Environ. Manage. 82, 363– 376.

Latati, M., Bargaz, A., Belarbi, B., Lazali, M., Benlahrech, S., Tellah, S., Ghiles, K., Drevon, J.J., Ounane, S.M. 2016. The intercropping common bean with maize improves the rhizobial efficiency, resource use and grain yield under low phosphorus availability. Eur. J. Agron. 72, 80–90.

Li, G.C., Mahler, R.L. & Everson, D.O. 1990. Effects of plant residues and environmental factors on phosphorus availability in soils. Communications in Soil Science and Plant Analysis 21,471-491.

Li, L., Li, S.M., Sun, J.H., Zhou, L.L., Bao, X.G., Zhang, H.G. & Zhang, F.S. 2007. Diversity enhances agricultural productivity via rhizosphere phosphorus facilitation on phosphorus-deficient soils. Proc. Natl. Acad. Sci. U. S. A. 104, 11192–11196.

Li, M. Jordan, N.R. Koide, R.T. Yannarell, A.C. & Davis A.S. 2016. Meta-analysis of crop and weed growth response to Arbuscular Mycorrhizal Fungi: Implication for Integrated Weed Management. Weed Science 64 (4), 642-652

Liang, W.L., Kosugi, K., Mizuyama, T. 2009. A three dimensional model of the effect of stemflow on soil water dynamics around a tree on a hillslope. Journal of Hydrology 366, 62-75

Liebman, M. & Dyck, E. 1993. Crop rotation and intercropping strategies for weed management. Ecol. Appl. 3, 92.

Lin, Y., P. Lu, X. Yang, and F. Zhang. 2013. Soil insect diversity and abundance following different fertilizer treatments on the loess plateau of China. Journal of Integrative Agriculture 12: 1644-1651.

Lundekvam, H. 2007. Plot studies and modelling of hydrology and erosion in southeast Norway. Catena 71, 200–9

Luo, Z., E. Wang, O.J. Sun. 2010. Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. Agriculture, Ecosystems and Environment 139, 224–231

Marshall, E.J.P., Arnold, G.M. 1995. Factors affecting field weed and field margin flora on a farm in Essex, UK. Landscape and Urban Planning 31, 205-216

Marshall, E.J.P., Moonen, A.C. 2002. Field margins in northern Europe: their functions and interactions with agriculture. Agriculture, Ecosystems and Environment 89, 5-21

Mazoyer M. and L. Roudart. 2006. A history of world agriculture from the neolithic age to the current crisis, Earthscan, London, United Kingdom.

McDaniel, M., Tiemann, L., Grandy, A. 2014. Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics? A meta-analysis. Ecological Applications 24(3), 560-570.

Menalled F.D., K.A. Kohler, D.D. Buhler, M. Liebman. 2005. Effects of composted swine manure on weed seedbank. Agriculture, Ecosystems and Environment 111, 63–69

Metzger, M.J., R.G.H. Bunce, R.H.G. Jongman, C.A. Mücher and J.W. Watkins. 2005. A climatic stratification of the environment of Europe. DOI: 10.1111/j.1466-822X.2005.00190.x

Miguez, F.E., Bollero, G.A. 2005. Review of corn yield response under winter cover cropping systems using meta-analytic methods. Crop Science 45(6), 2318-2329.

Miñarro, M., X. Espadaler, V.X. Melero and V. Suárez-Álvarez. 2009. Organic versus conventional management in an apple orchard: effects of fertilization and tree-row management on ground-dwelling predaceous arthropods. Agricultural and Forest Entomology 11: 133-142.

Mitchell, C.C., Westerman, R.L., Brown, J.R. & Peck, T.R. 1991. Overview of long-term agronomic research. Agronomy Journal 83: 24-29.

Mohler, C.L. 2001. Enhancing the competitive ability of crops. In: M. Liebman, C.L. Mohler & C.P. Staver (eds.). Ecological management of agricultural weeds. Cambridge University Press, Cambridge, UK. pp. 269-321.

Moonen, A.C., Marshall, E.J.P. 2001. The influence of sown margin strips, management and boundary structure on herbaceous field margin vegetation in two neighbouring farms in southern England. Agriculture, Ecosystems and Environment 86, 187-202

Moss, S. Storkey, J. Cussans, J.W. Perryman Sam. & Hewitt, M.V. 2004. The Broadbalk long-term experiment at Rothamsted: what has it told us about weeds? Weed Science 52, 864–873.

Munkholm, L.J., Heck, R.J., Deen, B. 2013. Long-term rotation and tillage effects on soil structure and crop yield. Soil and Tillage Research 127, 85-91.

Murphy, K.M., J.C. Dawson, and S.S. Jones. 2008. Relationship among phenotypic growth traits, yield and weed suppression in spring wheat landraces and modern cultivars, Field Crops Research 105, 107–115.

Nevens, F. & Reheul, D. 2001. Crop rotation versus monoculture; yield, N yield and ear fraction of silage maize at different levels of mineral N fertilisation. Netherlands Journal of Agricultural Science 49, 405-425.

Nicholas, V., N. Verhulst, R. Cox, and B. Govaerts. 2015. Weed dynamics and conservation agriculture principles: A review. Field Crops Research 183, 56-68.

O'Donovan, J.T. de St. Remy, E.A. O'Sullivan, P.A. Drew, D.A. & Sharma, A.K. 1985. Influence of the relative time of emergence of wild oat (Avena fatua) on yield loss of barley (Hordeum vulgare) and wheat (Triticum aestivum). Weed Science 33, 489-503.

Olsen, J. Kristensen, L. Weiner, J. & Griepentrog, H.W. 2005. Increasing density and spatial uniformity increase weed suppression by spring wheat. Weed Research 45, 316-321

Pandey, A., Li,F., Askegaard, M. & Olesen, J.E. 2017. Biological nitrogen fixation in three long-term organic and conventional arable crop rotation experiments in Denmark. European Journal of Agronomy 90, 87-95.

Patrick, L. B., M. W. Kershner, and L. H. Fraser. 2012. Epigeal spider responses to fertilization and plant litter: testing biodiversity theory at the ground level. Journal of Arachnology 40: 309-324.

Paustian, K., Parton, W.J. & Persson, J. 1992. Modeling soil organic matter in organic-amended and nitrogen-fertilized long-term plots. Soil Science Society of America Journal 56, 476-488.

Peigné, J. Ball, B. Roger-Estrade, J. & David, C. 2007. Is conservation tillage suitable for organic farming? A review, Soil Use and Management, 23, 129-144

Peoples, M.B., Herridge, D.F. & Ladha, J.K. 1995. Biological nitrogen fixation: an efficient source of nitrogen for sustainable agricultural production? Plant and Soil 174, 3-28.

Pittelkow, C.M.,B.A. Linquist, M.E. Lundy, X. Liang, K.J. van Groenigen, J. Leed, N. van Gestel, J. Six, R.T. Venterea, C. van Kessel. 2015a. When does no-till yield more? A global meta-analysis. Field Crops Research 183, 156–168

Pittelkow, C.M.,B.A. Linquist, M.E. Lundy, X. Liang, K.J. van Groenigen, J. Leed, N. van Gestel, J. Six, R.T. Venterea, C. van Kessel. 2015b. Productivity limits and potentials of the principles of conservation agriculture. Nature 517, 365-368. doi:10.1038/nature13809

Plaza-Bonilla, D., Nolot, J.M., Raffaillac, D. & Justes, E. 2015. Cover crops mitigate nitrate leaching in cropping systems including grain legumes: Field evidence and model simulations. Agriculture, Ecosystem and Environment 212, 1-12.

Poeplau, C., Don, A. 2014. Soil carbon changes under Miscanthus driven by C4 accumulation and C3 decompostion - toward a default sequestration function. GCB Bioenergy 6(4), 327-338.

Poeplau, C., Don, A. 2015. Carbon sequestration in agricultural soils via cultivation of cover crops - A meta-analysis. Agriculture, Ecosystems and Environment 200, 33-41.

Posner, J.L. Baldock, J.O. & Hedtcke, J.L. 2008. Organic and conventional production systems in the Wisconsin Integrated Cropping Systems Trials: I. Productivity 1990–2002. Agronomy Journal 100, 253–260.

Prado, J., C. Quesada, M. Gosney, M.V. Mickelbart, and C. Sadof. 2015. Effects of nitrogen fertilization on potato leafhopper (Hemiptera: Cicadellidae) and maple spider mite (Acari: Tetranychidae) on nursery-grown maples. Journal of Economic Entomology 108: 1221-1227.

Preissel, S., Reckling, M., Schläfke, N., Zander, P. 2015. Magnitude and farm-economic value of grain legume pre-crop benefits in Europe: A review. Field Crops Research 175, 64-79.

Qin, W., C. Hu and O. Oenema. 2015. Soil mulching significantly enhances yields and water and nitrogen use efficiencies of maize and wheat: a meta-analysis. Scientific Reports 5: 16210. DOI: 10.1038/srep16210.

Qin, W., F.B.T. Assinck, M. Heinen, and O. Oenema. 2016. Water and nitrogen use efficiencies in citrus production: A meta-analysis. Agriculture, Ecosystems and Environment 222: 103-111.

Quemada, M., Baranski, M., Nobel-de Lange, M.N.J., Vallejo, A. & Cooper, J.M. 2013. Meta-analysis of strategies to control nitrate leaching in irrigated agricultural systems and their effects on crop yield. Agric. Ecosyst. Environ. 174, 1–10.

Råberg, T.M., Carlsson, G. &Jensen, E.S. 2017. Productivity in an arable and stockless organic cropping system may be enhanced by strategic recycling of biomass. Renewable Agriculture and Food Systems. https://doi.org/10.1017/S1742170517000242

Rasmussen, K. Rasmussen, J. & Petersen, J. 1996. Effects of fertiliser placement on weeds in weed harrowed spring barley. Acta Agric. Scand. Sect. B Soil Plant Sci. 3:192–196.

Riksen, M., Brouwer, F., de Graaff, J. 2003. Soil conservation policy measures to control wind erosion in northwestern Europe. Catena 52, 309-326

Ryan, M.R., D.A. Mortensen, L. Bastiaans, J.R. Teasdale, S.B. Mirsky, W.S. Curran, R. Seidel, D.O. Wilson, P.R. Hepperly. 2010. Elucidating the apparent maize tolerance to weed competition in long-term organically managed systems. Weed Research 50, 25–36

Sainju, U.M. 2016. A global meta-analysis on the impact of management practices on net global warming potential and greenhouse gas intensity from cropland soils. PLoS ONE 11(2).

Sauer, T.J., Cambaradella, C.A., Brandle, J.R. 2007. Soil carbon and tree litter dynamics in a red cedar-scotch pine shelterbelt. Agroforestry Systems 71, 163-174

Schoumans, O.F. (ed.), Chardon, W.J.(ed.), Bechmann, M., Gascuel-Odoux, C., Hofman, G., Kronvang, B., Litaor, M.I., Lo Porto, A., Newell-Price, P. Rubæk, G. 2011. Mitigation options for reducing nutrient emissions from agriculture. A study amongst European member states of Cost action 869 Alterra-Report 2141

Sharpley, A.N., Chapra, S.C.,Wedepohl, R., Sims, J.T., Daniel, T.C. & Reddy, K.R. 1994. Managing agricultural phosphorus for protection of surface waters: issues and options. J Environ Qual 23, 437–51.

Siemann, E. 1998. Experimental tests of effects of plant productivity and diversity on grassland arthropod diversity. Ecology 79: 2057-2070.

Skaggs, R.W., M.A. Youssef, J.W. Gilliam, and R.O. Evans. 2010. Effect of controlled drainage on water and nitrogen balances in drained lands. Transactions of the ASABE 53 (6): 1843-1850.

Smith, R.G. Mortensen, D.A. & Ryan, M.R. 2010. A new hypothesis for the functional role of diversity in mediating resource pools and weed–crop competition in agroecosystems. Weed Research 50, 37–48.

Sørensen, P. & Thomsen, I.K. 2005. Separation of pig slurry and plant utilization and loss of nitrogen-15-labeled slurry nitrogen. Soil Science Society of America Journal, 69, 1644–1651.

Stahlman, P.W. & Miller, S.D. 1990. Downy brome (Bromus tectorum) interference and economic thresholds in winter wheat (Triticum aestivum). Weed Science 38, 224-228

Stenberg, M., Aronsson, H., Lindén, B., Rydberg, T. & Gustafson, A. 1999. Soil mineral nitrogen and nitrate leaching losses in soil tillage systems combined with a catch crop. Soil Tillage Res 50, 115-125.

Strauss, P. & Smid, G. 2004. Einfluss von Saattechnik und Zwischenfrucht auf den Oberflächenabfluss und die Bodenerosion im Zuckerrübenbau. Schriftenreihe BAW, 20, 91–109

Thomsen, I.K. 2005. Nitrate leaching under spring barley is influenced by the presence of a ryegrass catch crop: Results from a lysimeter experiment. Agriculture, Ecosystems and Environment 111, 21-29

Tomasoni, C., Borrelli, L., Brambilla, M. 2011. Energy balance of five fodder cropping systems in the irrigated lowlands of northern Italy. Italian Journal of Agronomy 6(1), 50-54.

Tonitto, C., David, M.B., Drinkwater, L.E. 2006. Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: A meta-analysis of crop yield and N dynamics. Agriculture, Ecosystems and Environment 112(1), 58-72.

Treadwell, D.D. Creamer, N.G. Schultheis, J.R. & Hoyt, G.D. 2007. Cover crop management affects weeds and yield in organically managed sweet potato systems. Weed Technology 21, 1039–1048.

Trost, B., A. Prochnow, K. Drastig, A. Meyer-Aurich, F. Ellmer, and M. Baumecker. 2013. Irrigation, soil organic carbon and N_2O emissions. A review. Agronomy for Sustainable Development 33 (4): 733-749.

Ulén, B., Aronsson, H., Bechmann, M., Krogstad, T., Øygarden, L. Stenberg, M. 2010. Soil tillage measures to control phosphorus loss and potential side-effects: A Scandinavian review. Soil Use and Management 26:94-107.

Uusi-Kamppa, J. & Mattila, P.K. 2010. Nitrogen losses from grass ley after slurry application –surface broadcasting vs. injection. Agricultural and Food Science 19, 327-40

Valkama, E., Lemola, R., Känkänen, H. & Turtola, E. 2015. Meta-analysis of the effects of undersown catch crops on nitrogen leaching loss and grain yields in the Nordic countries. Agriculture, Ecosystem and Environment 203, 93-101.

Van Dijk, W. & Van Geel, W. 2012. Advice base for the cultivation of arable crops. (Fertilizer recommendations for arable crops). PPO, Lelystad, the Netherlands.

Van Groenigen, J.W., O. Oenema, K.J. van Groenigen, G. Velthof, and C. van Kessel. 2010. Best Nitrogen Management Practices to Decrease Greenhouse Gas Emissions. Better Crops/Vol. 95 (2011, No. 2)

Van Ittersum, M.K. and R. Rabbinge. 1997. Concepts in production ecology for analysis and quantification of agricultural input-output combinations. Field Crops Res., 52 (1997), pp. 197-208

Van Kessel, C., R. Venterea, J. Six, M.A. Adviento-Borbe, B. Linquist and K. J. van Groenigen. 2013. Climate, duration, and N placement determine N2O emissions in reduced tillage systems: a meta-analysis. Global Change Biology 19, 33–44, doi: 10.1111/j.1365-2486.2012.02779.x

Van Vooren, L., Reubens, B., Broekx, S., Pardon, P., Reheul, D., van Winsen, F., Verheyen, K., Wauters, E., Lauwers, L. 2016. Greening and producing: an economic assessment framework for integrating trees in cropping systems. Agricultural Systems 148, 44-57

Veihe, A., Hasholt, B., Schiøtz, I.G. 2003. Soil erosion in Denmark: processes and politics. Environmental Science & Policy 6, 37-50

Venter, Z.S., Jacobs, K., Hawkins, H.J. 2016. The impact of crop rotation on soil microbial diversity: A meta-analysis. Pedobiologia 59(4), 215-223.

Verret, V., A. Gardarin, E. Pelzer, S. Médiène, D. Makowski, M Valantin-Morison. 2017. Can legume companion plants control weeds without decreasing crop yield? A meta-analysis. Field Crops Research 204, 158-168.

Vicente-Vicente, J.L., García-Ruiz, R., Francaviglia, R., Aguilera, E., Smith, P. 2016. Soil carbon sequestration rates under Mediterranean woody crops using recommended management practices: A meta-analysis. Agriculture, Ecosystems and Environment 235, 204-214.

Weiner, J. Griepentrog, H.W. & Kristensen, L. 2001. Suppression of weeds by spring wheat (Triticum aestivum) increases with crop density and spatial uniformity. Journal of Applied Ecology 313, 33-51.

West, T.O., Post, W.M. 2002. Soil organic carbon sequestration rates by tillage and crop rotation: A global data analysis. Soil Science Society of America Journal 66(6), 1930-1946.

Wiesenborn, W. D. 2011. Biomasses of arthropod taxa differentially increase on nitrogen-fertilized willows and cottonwoods. Restoration Ecology 19: 323-332.
Wivstad, M., Dahlin, A.S. & Grant, C. 2005. Perspectives on nutrient management in arable farming systems. Soil Use and Management 21, 113-121.
Wolfe M.S., J.P. Baresel, D. Desclaux, I. Goldringer, S. Hoad, G. Kovacs, F. Löschenberger, T. Miedaner, H. Østergård, and E.T. Lammerts van Bueren. 2008. Developments in breeding cereals for organic agriculture. Euphytica 163 (3), 323-346.
Wu, H., J. Pratley, D. Lemerle and T. Haig. 2000. Laboratory screening for allelopathic potential of wheat (Triticum aestivum) accessions against annual ryegrass (Lolium rigidum), Australian Journal of Agricultural Research 51, 259–266.
Yang, X., M. Warren, and X. Zou. 2007. Fertilization responses of soil litter fauna and litter quantity, quality, and turnover in low and high elevation forests of Puerto Rico. Applied Soil Ecology 37: 63-71.
Yang, Y., K. Liu, H. Han, H. Xu, F. Zhang, X. Zheng, J. Tian, G. Wang, G. Chen, Z. Lu. 2016. Impacts of nitrogen fertilizer on major insect pests and their predators in transgenic Bt rice lines T2A-1 and T1C-19. Entomologia Experimentalis et Applicata 160: 281-291.
Yardım, E. N., and C. A. Edwards. 2003. Effects of organic and synthetic fertilizer sources on pest and predatory insects associated with tomatoes. Phytoparasitica 31: 324-329.
Yu, Y., Stomph, T.J., Makowski, D., Zhang, L., van der Werf, W. 2016. A meta-analysis of relative crop yields in cereal/legume mixtures suggests options for management. Field Crops Research 198, 269-279.
Zhou, L., X. Zhou, J. Shao, Y. Nie, Y. He, L. Jiang, Z. Wu, and S. Hosseini Bai. 2016a. Interactive effects of global change factors on soil respiration and its components: a meta-analysis. Global Change Biology 22 (9): 3157-3169.
Zhou, X., L. Zhou, Y. Nie, Y. Fu, Z. Du, J. Shao, Z. Zheng, and X. Wang. 2016b. Similar responses of soil carbon storage to drought and irrigation in terrestrial ecosystems but with contrasting mechanisms: A meta-analysis. Agriculture, Ecosystems and Environment 228: 70-81.

17 Decision support tool for the pre-selection of key soil-improving cropping systems

O. Oenema

17.1 Background

This chapter describes a simple decision support tool for the pre-selection of key soil improving cropping systems (SICS) on the basis of SICS that have been identified in the literature reviews (Chapters 6-16). The pre-selected SICS are termed 'key SICS' here. The key SICS will be discussed further with stakeholders in WP 3 of SoilCare, and following approval by stake-holders, will be tested in field experiments in WP 4 and further assessed in WP 5 and WP 6.

The tool has to be simple and transparent, to allow the end-users insight in the decision making process. The decision tool will have to identify SICS with relatively large positive impacts on soil quality, crop productivity and profitability, and minimal negative environmental and social side-effects, in terms of resource use efficiency, environmental impacts, and human health effects. Further, the tool considers the applicability of the SICS for different environmental zones, soil types, farming systems, and socio-economic conditions.

17.2 Purpose

To develop and apply a simple decision tool for the pre-selection of key soil improving cropping systems (SICS), and to make a pre-selection of key soil-improving cropping systems.

17.3 Decision support tool for SICS

Decision making relates to making a choice among several alternatives, based on preferences. The outcomes of decision making processes are in general not equally attractive and it is therefore important to examine these outcomes (or alternatives) in terms of their preference or desirability. The quality of a decisions can often be improved by the decomposition of a problem into simpler components that are well defined and well understood. Supporting decisions means in this case supporting the decision-making process so that better decisions are made. In general, better decisions can be expected to lead to better outcomes. However, it is important to distinguish between good decisions and good outcomes. A good decision can be followed by a bad outcome. Conversely, a poor decision can lead to a good outcome by a fortunate occurrence that could not have been predicted.

The decision making process starts with two questions, i.e. is there a specific soil threat, and is there a need for soil quality improvement? These questions relate to the two types of SICS that have been distinguished in this review, namely (i) soil-threat specific SICS and (ii) general SICS (Chapter 5). The distinction between these two depends on the presence of specific soil threats and soil quality issues (Figure 17.1). Both soil-threat specific and general SICS are composed of

crop rotations and specific agro-management techniques, which have to be prioritized in an optimization process (Chapter 5). If there are no soil threat(s) and no soil quality issue(s), cropping systems can be optimized for high yield, profitability and sustainability without *specific* consideration of soil threats and soil quality improvement needs. Alternatively, in case there are soil threats and/or soil quality issues, specific crop types, crop rotations and agro-management techniques will have to be prioritized in the optimization of cropping systems, so as to mitigate/prevent the soil threat(s) and/or enhance the soil quality in general.

The scheme presented in Figure 17.1 helps in making choices by raising questions and identifying the need for prioritization. The scheme is 'target-oriented' or "impact driven", i.e., the scheme leads to SICS with the greatest positive impact on soil quality and crop productivity, through the prioritizaiton of specific crop types, rotations and/or agro-management techniques, depending on the specific soil threats and/or need for soil quality improvement.

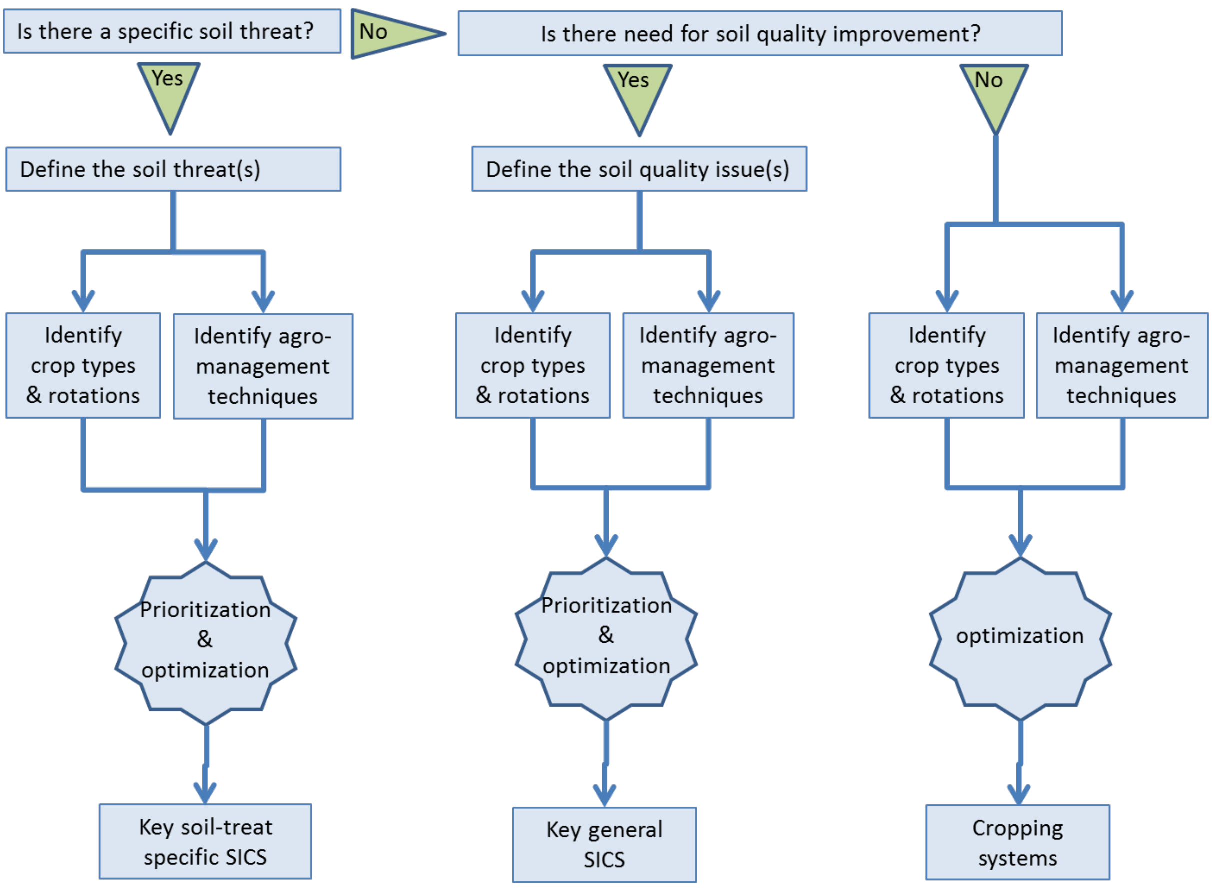


Figure 17.1. *General scheme of decisision making for the selection of key SICS. The presence of soil threats and/or soil quality issues defines the need for prioritization of specific crop types, crop rotations and agro-management techniques in the optimization of cropping systems (see also chapter 5).*

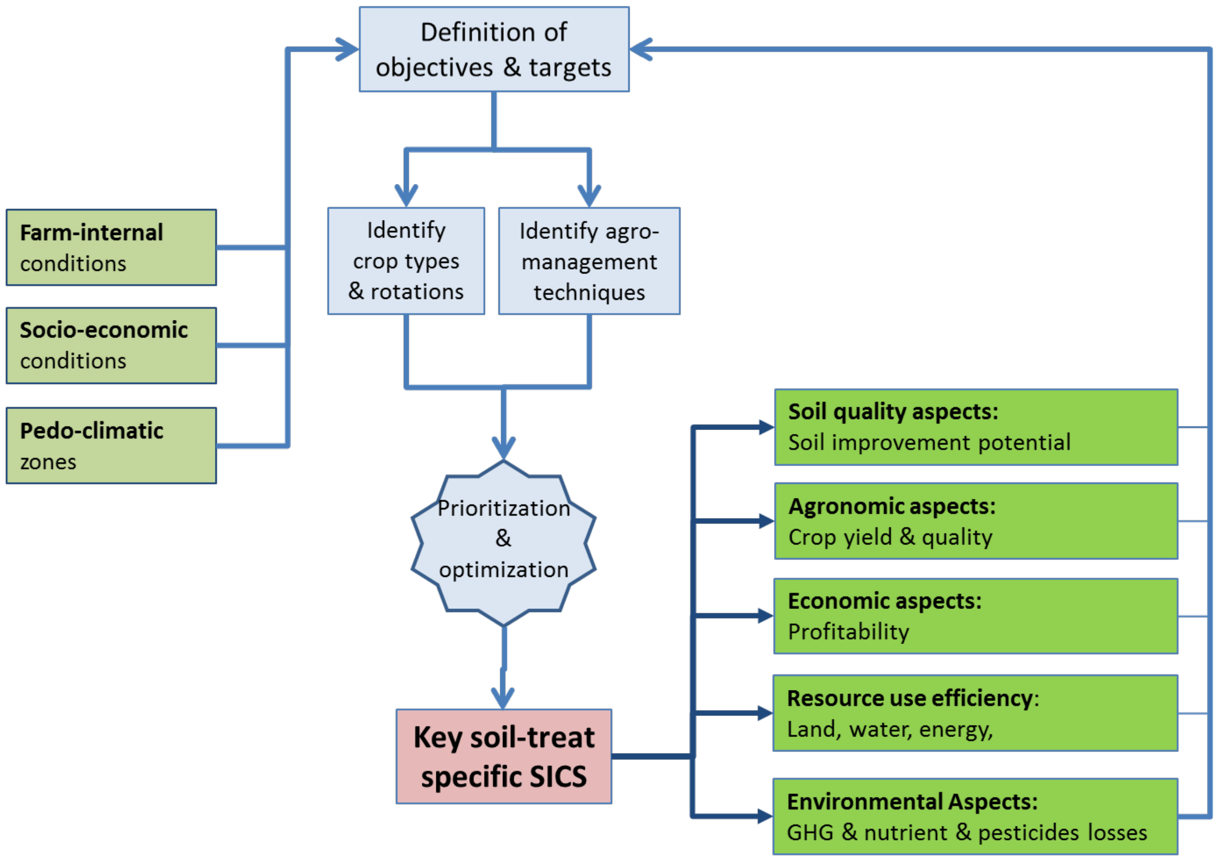


Figure 17.2. *Scheme for the identification of SICS and for the verification of outcomes; do the outcomes of the SICS meet the objectives/targets, in terms of soil quality, agronomy, economy, resource use and environmental impacts? If targets are not met, a new round of prioritization and optimization of crop types and agro-management techniques will be needed. The targets are a function of the farm internal conditions and preferences, and the external socio-economic (markets, infrastructure) and environmental conditions (pedo-climatic zones).*

Critical in the decision making process are (i) the definition of objectives and targets, based on the farm internal and external conditions, and (ii) the verification of the outcomes; do the outcomes of the SICS meet the set objectives/targets, in terms of soil quality, agronomy, economy, resource use and environmental impacts (Figure 17.2). The definition of the objectives/targets depends on the farm internal conditions (farm type, farmer's preferences, soil threats, etc) and the farm external conditions (markets, extension services, climate, etc.). The outcome of the decision making process depends in turn on the prioritization and optimization; if the outcomes of the SICS do not meet the set objectives and targets, the selections of the crop types and/or specific agro-management techniques will have to be re-considered, in a new round of optimization and prioritization.

Whether a new round of optimization and prioritization is needed, depends on the outcomes of the pre-selection of crop types and agro-management techniques. A so-called SWOT

(Strength, Weakness, Opportunities, Threats) analysis (Humphrey, 2005) can be helpfull to analyse the outcomes. Strengths are characteristics that contribute to achieving the main objectives of the SICS, based on the atrributes of the SICS (internal focus), while weaknesses are harmfull to achieving the main objectives based also on the attributes of the SICS (internal focus). Opportunities are characteristics that contribute to achieving the objectives of the SICS based on the attributes of the socio-economic and pedo-environmental zones (external focus), while threats are harmfull to achieving the objectives of the SICS based on the attributes of the socio-economic and pedo-environmental zones (Figure 17.3).

Figure 17.3. *Scheme for the SWOT analysis of the identified soil-improving cropping systems.*

	HELPFUL **to achieve the objectives of the SICS**	**HARMFUL** **to achieve the objectives of the SICS**
INTERNAL FOCUS **(attributes of the SICS)**	**STRENGTHS** • … • … •	**WEAKNESSES** • … • …
EXTERNAL FOCUS **(attributes of environment)**	**OPPORTUNITIES** • … • … • …	**THREATS** • … • … •

As an example, Figure 17.4 presents the SWOT analysis of the soil-threat-specific SICS "crop rotations with low wheel load - low tyre pressures" for reducing the risk of soil compaction. Harvesting machines and animal manure application machines with low wheel load - low tyre pressures are effective in reducing the risk of soil compaction, and are therefore strongly recommended. However, the adoption of machines with low wheel load - low tyre pressures in practice is still low in large-scale, intensive farming, mainly because of the lower operational capacity of these machine, which increases labour costs and increases possibly also the risk of having to conduct field work under less optimal wheather conditions, when the activities take too much time. This weakness may be overcome possibly by developing innovative machines

with low wheel load - low tyre pressures and high operational capacity. A possible threat for the adoption of machines with low wheel load - low tyre pressures is the increasing wish of consumers to have uniform and fresh products, and hence the risk that the processing industry increases the crop quality criteria for the fressness of harvested products, which may not be met by the current machines with low wheel load - low tyre pressures. Evidently, the SWOT analysis may help to optimize the further development of the agro-management techniques as well as help in the prioritization of the optimal combination of crop type and agro-management techniques. A possible (partial) alternative for machines with low wheel load - low tyre pressures may be controlled traffic in combination with adjusted crop rotations. This depends also on the pedo-climatic and socio-economic conditions.

Figure 17.4. *Example SWOT analysis of the soil-threat-specific SICS "crop rotations with low wheel load, low tyre pressures".*

	HELPFUL	**HARMFUL**
INTERNAL	**STRENGTHS** • Decreased risk of soil compaction	**WEAKNESSES** • Low operation capacity; increased labour cost and increased risk of working on less optimal conditions
EXTERNAL	**OPPORTUNITIES** • Machine companies developing new type of machines	**THREATS** • Processing industry increase crop quality criteria on fressness

17.4 Impact of climate, soil and socio-economic constraints for SICS

The feasibility/suitability of (components of) SICS depend in part also on climatic conditions, land and soil conditions, and socio-economic conditions. Conversely, the combination of the these conditions also determine the risk of soil threats (Stolte et al., 2016).

Governing climate factors are photosynthetic active radiation, rainfall, and temperature during the growing season, which together determine the length of the growing season (Table 17.1). These factors influence the choice of crop type and crop rotation, as well as the agro-management techniques. The factors vary from north to south and from west to central Europe, and show up in the map of environmental zones. Europe is divided in 13 main environmental zones, with clear differences in climatic conditions (Metzger et al., 2005). Main climate constraints for crop production are a short growing season for northern Europe, and low rainfall during the main growing season in the Mediterranean and central Europe. Photosynthetic

active radiation (PAR, in Wm^{-2}) is mainly determined by latitude and the inclination of the slope, and increases from north to south Europe. Rainfall during the growing season (Rain, total and distribution) is determined by a number of meteorological factors, including oceanity, latitude, altitude, and geomorphology. Rainfall distribution is as important as total rainfall, as dry spells during critical crop growth stages, or heavy rains can be damaging to crop yield and contribute to soil treats. Mean temperature during the growing season (Temp, °C) is also determined by meteorological factors, including oceanity, latitude, altitude, and geomorphology.

Table 17.1. *Main climate-related factors that influence crop growth and development, and thereby the choice of crop rotation and agro-management techniques.*

Abbreviation	Climate factors
GSL	Growing season length (days)
PAR	Photosynthetic active radiation (Wm^{-2})
Rain	Rainfall during the growing season (mm)
Temp	Mean temperature during the growing season (°C)

Main governing land and soil conditions are slope and relief, soil depth, stoniness, soil texture, soil structure, and soil organic matter content (Table 17.2). These factors also influence the choice of crop type and crop rotation, as well as the agro-management techniques. Slope and aspect (SA) influence the micro-climate and hence yield potential, mechanization options and labour demands. Slope and aspect also determine the risk of nutrient losses via overland flow and erosion. Soil depth (SD) determines the soil volume that can be explored by roots, as well as the soil water and nutrient storage and delivering capacity. Stoniness (St) also influences the soil volume that can be explored by roots, as well as the soil water and nutrient storage and delivering capacity. Stoniness will also influence crop choice and mechanization. Soil texture (ST) determines the soil water and nutrient storage delivering capacity, soil fertility, workability and hence may influence crop choice and mechanization. Soil organic matter (SOM) content influences the soil water and nutrient storage and delivering capacity, biodiversity, workability and hence may influence crop choice and mechanization; both a low and high SOM content is suboptimal. Soil structure (SS) influences the workability and hence crop choice and mechanization. Soil structure also influences the germination of plant seeds, as well as the soil water storage and delivering capacity of the soil.

Table 17.2. *Main land and soil conditions related factors that influence the choice of crop rotation and agro-management techniques.*

Abbreviation	Climate factors
SR	Slope, relief and inclination (towards the south) (%)
SD	Soil depth (cm)
St	Stoniness (wt %)
ST	Soil texture (%clay)
SS	Soil structure (qualitative score)
SF	Soil fertility (extractable nutrients)
SOM	Soil organic matter content (%)

Main socio-economic factors relate to access to markets, technology, labour, advice and financial capital (Table 17.3). These factors also influence the choice of crop type and crop rotation, as well as the agro-management techniques. Access to markets (AM) is of key importance, as it affects the ability to market products and thereby the price of the produce and hence farm income. Access to markets is also important for obtaining farm inputs. Access to (new) farm technology (AT) determines the modernization potential of the farm; this may both reduce the risk of some soil threats as well as form a barrier for the implementation of components of SICS (e.g., machines and equipment for controlled traffic). Access to labour (AL) is important for crops with high labour demand during the planting and harvest seasons; some crops may not be grown without sufficient qualified labour. Access to advice (AT; agronomic and economic) from specialists may hinder the modernization of the farm (cropping systems) and the improvement of farm performance (allocation of production factors may be suboptimal). Access to capital (AC) is important for investments and hence farm modernization and size. All these socio-economic factors are influenced by infrastructure and the distance to markets, cities and R&D centres. In addition to these external socio-economic factors, there are personal factors and preferences that influence farmers' behaviour and choices. These personal factors and preferences may have a background in culture and education, and may be influenced also by the local society.

Table 17.3. *Main socio-economic conditions related factors that influence the choice of crop rotation and agro-management techniques.*

Abbreviation	Climate factors
AM	Access to markets (km, network & costs of transport)
AT	Access to technology (km, network & costs of transport)
AL	Access to labour (km, network & costs of transport)
AA	Access to advice (km, network)
AC	Access to capital (km, network)

Evidently, the various components of SICS depend on the aforementioned conditions set by the site-specific climate, land and soil, and market and society (including policy, public opinion, public attitude). Not all crops can be grown under all environmental conditions. The growth and success of a cover crop depends on the harvest of the main crop and the length of the remaining growing season. Similarly, not all fertilization practices can be applied at all environmental conditions; fertilizers and manures have to be incorporated into the soil on sloping land to prevent the loss of the nutrients. However, main barriers for the implementation of SICS seem farm profitability (on the short-term) and lack of knowledge and awareness among farmers and land managers about soil threats and SICS. Benefits of SICS often emanate on the longer term, while farmers and land managers have to bear the costs upon implementation of the components of SICS.

Crop rotations, fertilization, irrigation, drainage, and pest and weed control all have a large effect on farm income. Tillage, mulching, traffic management and landscape management have in general a modest effect on farm income. Fertilization, irrigation, drainage, and pest and weed control often have a negative effect on the environment, but the assessment differs when the effects are based on a product or area basis. The environmental effects often have a minimum at optimal inputs of fertilizers, irrigation, drainage, and pest and weed control when the environmental effects are expressed on a product basis (and not on an area basis) . The same holds for resource use efficiency. High (excessive) inputs generally have negative environmental effects, both expressed on a product and area basis. Hence, the definition of inputs depend on (i) the level (rate) of input, and (ii) the units chosen, i.e. area or product basis.

In summary, the choice of crop type, crop rotation and agro-management techniques depends on socio-economic conditions, climate and land and soil conditions (Tables 17.1, 17.2, 17.3). Some crop types and crop rotations may not be feasible on some soil types, and/or not possible in some environmental conditions. The growth of sunflower, olives, grapes, oranges is economically feasible in the southern part of Europe. Root crops may not be grown easily in heavy-textured soils. The assessments so far have not consider possible interactions between components, which can be positive (synergistic) and negative (antagonistic). For example, fertilization is most attractive when there are no other growth constraints than nutrient elements. The same applies to irrigation; it is economically most profitable when no other growth limiting and reducing factors occur.

17.5 A preselection of SICS

Based on the decision support schemes (Figures 17.1, 17.2) and the SWOT analyses of possible SICS (Figures 17.3, 17.4) *priority* crop types and rotations, and *priority* agro-management techniques have been selected for soil-threat-specific SICS and for general SICS. These so-called priority crop types and rotations, and priority agro-management techniques have been found to be effective in preventing and remediating specific soil threats and improve soil quality, as discussed in Chapters 6-16.

Soil-threat specific SICS target a specific soil threat through specific crop rotations and/or agro-management techniques (Chapters 5-15). Soil-threat specific SICS require specific adjustments of crop rotations and agro-management techniques, to reduce the threat and alleviate the effects of the threats, as a function of the environmental and socio-economic conditions. Table 17.4 presents priority crop types and priority agro-management techniques of soil threat-specific SICS; these priority crop types and priority agro-management techniques servie as building blocks for the pre-selection of soil threat-specific SICS. The actual selection will depend on the farm internal and external (socio-economic and environmental) conditions (Figure 17.2).

Table 17.4. *Prioritization of crop types and agro-management technique in soil threat-specific SICS.*

Nr	Soil threat-specific SICS	Priority crop types	Priority agro-management techniques
1	Acidification	No specific crop type	Liming, manuring
2	Erosion	Permanent groundcover, Deep-rooting crops Cereals with cover crops Alfalfa, Agroforestry	Zero-tillage, landscape management, contour traffic Proper timing of activities
3	Compaction	Deep-rooting crops, Cereals, perennial rye, alfalfa	Controlled traffic Low wheel load, low tyre pressures Proper timing of activities
4	Pollution	Biofuel crops Some fodder crops No leafy vegetables	No use of polluted inputs Tree lines to scavenge air-born pollution Restricted use of pesticides
5	Organic matter decline	Permanent groundcover, deep-rooting crops Cereals with cover crops, alfalfa	Minimum tillage, Residue return, Mulching Manuring
6	Biodiversity loss	Crop diversification	Manuring, minimum tillage, residue return, No pesticides, Restricted fertilization
7	Salinization	Salt-tolerant crops	Drainage Targeted irrigation Ridging Proper timing of activities
8	Flooding	Flooding-tolerant crops	Drainage Landscape management
9	Landslides	Deep-rooting crops, trees	Landscape management, No arable cropping
10	Desertification	Deep-rooting C4 crops	Landscape management Irrigation

General SICS improve soil quality and soil functions in general. The main soil function in cropping systems is crop production, which is mainly determined by the 6 crop yield limiting and reducing factors:

1) Water retention and delivery to crops, i.e. soil depth and water holding capacity
2) Nutrient retention and delivery to crops, fertility indices,
3) Control of pathogens and weeds, and improve soil biodiversity,
4) Soil structure and tilth,
5) Control of pollutants, and
6) Control of organic matter content and quality

General SICS are also composed of crop rotations and specific agro-management techniques, which have to be prioritized in the optimization process (Figure 17.5). A pre-selection of priority crop types and priority agro-management techniques of general SICS are presented in Table 17.5. These priority crop types and priority agro-management techniques servie as building blocks for the pre-selection of general SICS; the actual selection will depend on the farm internal and external (socio-economic and environmental) conditions.

Table 17.5. *Prioritization of crop types and agro-management technique in general SICS.*

Nr	Targets of general SICS	Priority crop types	Priority agro-management techniques
a	Soil structure improvement	Permanent groundcover, Deep-rooting crops Cereals with cover crops Alfalfa, clovers	Minimum tillage, Proper timing of activities Manuring Liming Proper timing of activities
b	Balanced nutrition	No specific crops	Fertilization based on soil fertility and plant leaf analyses, targeted manuring
c	Increasing crop yield	High-yielding crop varieties	Proper timing of activities, in-depth soil analyses, frequent field observation, targeted irrigation, fertilization, pest management and weed control
d	Coping with and benefiting from spatial variations in soil quality	No specific crops	Establishing relationships between spatial variations in soil quality and spatial variations in crop yield, Variable rate tillage, liming, manuring, irrigation seeding, fertilization, and crop management.
e	Improving soil quality, farm profitability and cropping system sustainability	Wide crop rotations with high values crops, leguminous crops, cover crops	Site-specific optimization of the agro-management techniques

Tables 17.4. and 17.5 list the priority crop types and agro-management techniques, because these have been shown in the literature studies and meta-analyses reviewed in Chapters 6-16 to prevent and/or alleviate soil threats, and improve soil quality. The listed crop types and agro-management techniques have to be combined / optimized with other crop types and agro-management techniques, so as to further increase farm income and the sustainability of the cropping system. Common to most soil threat-specific SICS are crop rotations with cereals, green manures, cover crops and catch crops so as to have groundcover, deep-rooting crops and crops with relatively large amounts of crop residues. Such crops may increase soil organic matter content, increase biodiversity, suppress soil-borne pathogens (depending on crop species), improve soil structure, and decrease nutrient leaching, run-off and erosion. The feasibility of green manures, cover crops and catch crops depends on the date of harvest of the main crop, the planting date of the next crop, climatic conditions during autumn and winter seasons, and the characteristic of the green manures, cover crops and catch crops (susceptibility for soil borne diseases, winter hardness, etc.).

Agroforestry systems are especially useful in hilly and mountaineous areas, to prevent and minimize soil erosion. They may be considered a subsystem of permanent cropping systems or of landscape management elements (tree lines, hedges, riparian zones). Agroforestry may contribute to biodiversity and landscape diversity; it modifies the micro-climate and reduces erosion. Intercropping, mixed cropping, alley cropping, strip cropping, double cropping all may have specific benefits for enhancing total crop yield, soil organic matter input, increasing biodiversity, and improving soil structure under certain conditions, but often have disadvantages in terms of mechanization and labour efficiency. They have not been considered here.

17.6 Monitoring of SICS

Indicators are defined as measurable phenomena with a specific function. Soil quality, soil functions and soil threats are defined in terms of indicators. These indicators are important for monitoring changes in soil quality and soil threats, and hence also for assessing the effectiveness and efficiency of SICS.

There are many possible indicators, but to ease effective communication and comparision, it is important to have a common terminology and a set of common approved/agreed indicators. A glossary of terms has been defined in this review, on the basis of literature review (Heinen, 2016), and has been made available through the website of SoilCare. A minimum data set of soil quality indicators has been defined by Doran and Parkin (1996). An extensive list of soil indicators will be compiled in the ongoing EU iSQAPER project. Here, we have made use of these list of soil indicators, and have slightly modified the indicators where feasible, so as to obtain a list of key indicators (see below).

Indicators can be defined at different spatial and temporal scales, and also at different functional scales. Further, a distinction is often made between single-issue and integrated

indicators. The six indicators identified for assessing the capacity of the soil to deliver high crop yields are so-called combined or integrated indicators, i.e. they are based on a number of different measurements. This section provides lists of

1. Indicators for the profitability and sustainability of SICS (in Table 17.6),
2. Soil quality indicators and properties that have to be measured for a proper monitoring of changes in soil quality (in Table 17.7), and
3. Soil properties that have to be measured for a proper monitoring of the effectiveness of soil threat-specific SICS (in Table 17.8).

Table 17.6. *General indicators for the profitability and sustainability of SICS. Last column indicates the frequency of the measurements/observations.*

Indicators	Unit	Frequency, yr^{-1}
Crop yield	kg ha^{-1} yr^{-1}	1
Crop quality	Contents of starch, protein, fatty acids, oils, minerals, vitamins, form, shape, colour, etc.	1
Marketable yield (gross)	Euro ha^{-1} yr^{-1}	1
Land cost	Euro ha^{-1} yr^{-1}	1
Labour costs	Hours ha^{-1} yr^{-1} and Euro ha^{-1} yr^{-1}	1
Building and infrastructural costs	Euro ha^{-1} yr^{-1}	1
Nutrient management costs	Euro ha^{-1} yr^{-1}	1
Irrigation costs	Euro ha^{-1} yr^{-1}	1
Drainage costs	Euro ha^{-1} yr^{-1}	1
Soil cultivation costs	Euro ha^{-1} yr^{-1}	1
Pest management costs	Euro ha^{-1} yr^{-1}	1
Weed control costs	Euro ha^{-1} yr^{-1}	1
Crop residue /mulching costs	Euro ha^{-1} yr^{-1}	1
Machine costs	Euro ha^{-1} yr^{-1}	1
Landscape management costs	Euro ha^{-1} yr^{-1}	1
		1
Carbon sequestration	kg CO_2-eq ha^{-1} yr^{-1}	1
Methane emissions	kg CO_2-eq ha^{-1} yr^{-1}	1
Nitrous oxide emissions	kg CO_2-eq ha^{-1} yr^{-1}	1
Fuel use	kg CO_2-eq ha^{-1} yr^{-1}	1
Electricity use	kg CO_2-eq ha^{-1} yr^{-1}	1
	-	
Nitrogen balance	kg N ha^{-1} yr^{-1}	1
Phosphorus balance	kg P ha^{-1} yr^{-1}	1
Potassium balance	kg K ha^{-1} yr^{-1}	1
Nitrate concentration in waters	mg NO_3-N L^{-1}	1
Phosphate concentration in waters	mg P L^{-1}	1
Total nitrogen in waters	mg N L^{-1}	1
Ammonia emissions to air	kg NH_3-N ha^{-1} yr^{-1}	1

Table 17.7. *Possible soil quality indicators and related soil properties for assessing the quality of the soil for crop production. The last column indicates the frequency of the measurements.*

#	Indicator of soil quality	Measurable soil properties	Frequency, yr^{-1}
1	Soil water retention and delivery	Soil depth (m)	0.1
		Mean groundwater level (m)	2-12
		Soil moisture retention curve	0.1
		Bulk density ($g\ cm^{-3}$)	0.2
2	Soil nutrient retention and delivery (rating; low-high)	pH	0.2
		Bulk density ($g\ cm^{-3}$)	0.2
		SOM (%)	0.2
		Extractable N, P, K, Ca, Mg, Na, S, Cl, Cu, Zn, Co, Mn, Fe, Mo ($mg\ kg^{-1}$)	0.2
		Texture: clay, silt, sand (%)	0.1
3	Soil-borne pathogens & soil biodiversity	Earthworms diversity (number per species)	1
		Collembola (springtails) diversity (number per species)	1
		Microbial respiration (mg CO_2-C m^{-2} day^{-1})	1
		Parasitic fungi,	1
		Parasitic nematodes	1
		DNA sequencing	0.1
4	Soil-borne weeds	Germination of weeds (number m^{-2})	1
		Stubborn weeds	1
5.	Soil structure and tilth	Size of soil aggregates (mm)	
		Shape and stability of aggregates	
		Water infiltration rate (cm hr^{-1})	
6.	Soil pollutants	Extractable (in $\mu g\ kg^{-1}$) - Heavy metals - Organic micro pollutants - Oil residues - Metals from actinide series	0.2
		Plastics	0.1
		Antibiotics	0.2
7	SOM content and quality	Total C (%)	0.2
		Mineralizable C ($g\ kg^{-1}\ yr^{-1}$)	0.2
		Extractable C & N (DOC, DON) ($mg\ L^{-1}$)	0.2
		C/N ratio	0.2

Table 17.8. *Main soil quality indicators and related soil properties for assessing the effectiveness of soil threat specific SICS.*

#	Soil threat specific SICS	Soil Quality indicator	Measurable soil properties
1	Acidification	Change of acid neutralizing capacity (mol_c ha^{-1} yr^{-1})	Sum of basic cation minus sum of anions
		Change of soil pH	pH (H_2O), pH (KCl), pH ($CaCl_2$)
2	Erosion	Loss of soil (ton ha^{-1} yr^{-1})	Mass of soil (via wind / water)
		Soil surface phenomena	Visual observation
		Aggregate stability of surface soil (%)	Aggregate stability
		Soil cohesion	Shear strength
3	Compaction	Bulk density (g cm^{-3})	Bulk density
		Water infiltration rate (mm day^{-1})	Water infiltration rate
		Penetration resistance (MPa cm^{-2})	Penetration resistance
4	Pollution	Metal content (mg kg^{-1})	Cd, Pb, Cr, Zn, Cu, As contents
		Organic pollutants (µg kg^{-1})	PACs, PCBs
		Radiation pollution (beq kg^{-1})	Actinides
		Oil (mg kg^{-1})	Oil
		Plastics (mg kg^{-1})	plastic
		Antibiotics (µg kg^{-1})	Antibiotics
5	Organic matter decline	Total organic C (g kg^{-1})	Organic C
		Mineralizable C (g kg^{-1} yr^{-1})	Mineralizable organic C
		C/N ratio	C/N ratio
6	Biodiversity decline	Earthworms diversity (number per species)	Number per species
		Collembola (springtails) diversity (number per species)	Number per species
		Microbial respiration (mg CO_2-C m^{-2} day^{-1}	Respiration
		Parasitic fungi (m)	
		Parasitic nematodes (number per species)	Number per species
7	Salinization	Extractable salt contents (mg kg^{-1})	Na, K, Cl, SO_4^{2-}, HCO_3^-
		EC (mS)	Electric conductivity
		pH	pH (H2O), pH (KCl), pH (CaCl2)
		Soil structure (descriptive)	Soil structure
8	Flooding	Period and number of days $year^{-1}$	Flooding
		Regional drainage (canals, dams, pumping stations)	Descriptive
9	Landslides	Tree density (number m^{-2})	Number of trees
		Drainage (canals, rivers)	Descriptive
10	Desertification	Change in green cover (ha yr^{-1})	Surface mapping
		Water infiltration rate (mm day^{-1})	Infiltration rate

17.7 Conclusions

Soil improving cropping systems (SICS) are "*cropping systems that improve soil quality (and hence its functions), prevent and/or minimize soil threats, and have positive impacts on the profitability and sustainability of cropping systems*". The SICS concept is rather new; possible measures/components have been reviewed, the concept has been further elaborated and a preselection of SICS has been made. Indicators needed for its monitoring have been defined.

Soil improving cropping systems (SICS) are a combination of crop rotations and 9 agro-management techniques. Specific components of the SICS have to be prioritized to address soil quality concerns, which likely depend on site-specific conditions and socio-economic drivers. Next, crop rotations and the 9 agro-management techniques have to be integrated and optimized for site-specific environmental and socio-economic conditions, so as to both address soil quality, farm profitability and the sustainability of the cropping system.

Two categories of SICS have been distinguished, (i) soil threat specific SICS, which mitigate the threat and alleviate its effects, and (ii) general SICS, which enhance soil quality and soil functions in general.

The concept of SICS is still somewhat theoretical, i.e., there are ideas and partial proofs of its applicability, effectiveness and efficiency, but there are no comprehensive descriptions of a framework, handbook, guidance document, and/or results of the concept in practice yet. These have to be developed, tested and refined further in SoilCare. The current report forms the start for defining the concept and setting up such a framework.

The list of promising SICS are formulated in a rather general manner, mainly because SICS are site-specific and the crop rotations and agro-management techniques have to optimized and integrated for site and farm specific conditions. The SICS concept presented here basically is a tool box of crop types, crop rotations and agro-management techniques. Depending on the local/regional environmental and socio-economic conditions, the farmer (with or without advisors) will select the appropriate combinations of crop types, crop rotations and agro-management techniques. The effectiveness of the selected combinations has to be assessed on the basis of monitoring programs of profitability, sustainability and soil quality indicators.

Recommendations	For whom
• Define hypotheses and treatments to test soil-improving cropping systems • Test the concept and usefulness of general SICS and soil threat-specific SICS	Science
• Test the SICS concept in practice, and consider the options and possible barriers for its implementation. • Make use of demonstration fields to show the importance of SICS	Practice
• Raise awareness on the importance of soil quality in society and practice • Consider to include priority crop types and agro-management techniques (section 5.2) in the CAP and/or Rural Development Regulation.	Policy

17.8 References

Doran, J.W. and T.B. Parkin. 1996. Quantitative indicators of soil quality: a minimum data set. In: J.W. Doran and A.J. Jones (eds.), Methods for Assessing Soil Quality. SSSA Special Publication Number 49, pp. 25-37, Madison, Wisconsin, USA.

Heinen, M. 2016. Glossary of soil terms. Compiled on the basis of the EU Landmark project, the EU project Recare and other sources. https://www.soilcare-project.eu/glossary.

Humphrey, A.S. 2005. SWOT Analysis for Management Consulting. SRI Alumni Association Newsletter, December 2005, pp. 7-8. https://www.sri.com/sites/default/files/brochures/dec-05.pdf (last access: 10-June-2016).

Metzger, M.J., R.G.H. Bunce, R.H.G. Jongman, C.A. Mücher and J.W. Watkins. 2005. A climatic stratification of the environment of Europe. DOI: 10.1111/j.1466-822X.2005.00190.x

Stolte et al. (eds.). 2016. Soil Threats in Europe. JRC Technical Report JRC 98673, EUR 27607 EN; doi:10.2788/488054 (print); doi:10.2788/828742 (online); available at: https://ec.europa.eu/jrc/en/publication/soil-threats-europe?search (last accessed: 19-05-2016).