

Soil Constraints on Crop Production

Edited by
Yash Dang
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FOREWORD

Agriculture is an undoubted success in continuing to feed and clothe the world and is increasingly mitigating the societal and environmental concerns that stem from global change. This continual adaptation of agriculture, fit for the future, is underwritten by research innovation, is manifest in bred varieties and modern agronomic practices, and is driven by dedicated and innovative farmers supported by agricultural science.

Sustainable agriculture is essential to meeting the Sustainable Development Goals (SDGs) to end poverty and contribute to economic growth at farm, national, and global levels. Critical to the SDGs is the continued increase in agricultural productivity to meet demands from population growth without expanding agricultural land.

The soil resource defines agricultural land and practices and, consequently, soil must be the foundation on which improved productivity is focused. If soils are constrained, naturally or through management, then plant growth is restricted, and agricultural productivity is constrained. Recent estimates suggest that only 24% of agricultural land worldwide is currently without any major soil constraint – e.g., from salinity, sodicity, erosion, nutrient deficiencies, acidity, and/or compaction. Improved understanding and management of soil constraints is critical for future productivity, growth, and food security.

This book provides a comprehensive compendium of global research on soil constraint management, with contributors from around the world providing insight into the identification of soil constraints, the types of soil constraints currently affecting agricultural production, and the latest techniques available to ameliorate constraints and/or support agricultural production in constrained areas. Importantly, it also contains a series of chapters detailing the nature of soil constraints and their management across different geographical and socioeconomic regions, authored by expert practitioners.

There is no doubt that how we farm in the future will continue to evolve. I firmly believe that this book is an invaluable compilation of expert opinions from around the world and a great resource for the future management of constrained soils worldwide. I congratulate the editors, contributors, and publishers for this most worthwhile initiative.

Dr Peter Carberry
General Manager, Applied Research, Development and Extension
Grains Research and Development Corporation
Australia

PREFACE

Over three quarters of the world's agricultural land is affected by some type of soil constraint, which leads to significant yield loss globally. The effective management of soil constraints is thus required if the world is to rise to the challenge of feeding our rapidly increasing population.

The type and severity of soil constraints varies from location to location. Constraints may occur singly or in combination and tend to vary both spatially across the landscape and within soil profiles, occurring in both top-soil and subsoil layers. In addition, appropriate management practices will be dependent on geographical location, the type of farming system employed, and farmer access to resources. Consequently, it is not possible to use a "one size fits all" approach to soil constraint management.

To help improve the resources available to researchers and land managers, we have collated global information regarding the latest developments in soil constraint identification, characterisation, and management. The contents of this book are divided into six sections and 25 chapters (see below). The introductory chapters outline the problem and impact of soil constraints globally and the challenges associated with their identification and mapping. The characteristics and management of the specific types of physical, chemical, and biological soil constraints most prevalent worldwide are then outlined in detail. This is followed by case studies from different geographical regions worldwide. Finally, we conclude with perspectives on the management of multiple soil constraints, and the socioeconomic and environmental impacts of soil constraints.

Introduction	• Introduction and soil constraint mapping
Physical Constraints	• Compaction, waterlogging, and erosion
Chemical Constraints	• Sodicity, salinity, acidity, nutrient deficiencies, and nutrient toxicities.
Biological Constraints	• Soil borne diseases, and soil fertility decline
Regional Case studies	• Southern Africa, Central Asia, South-east Asia, Southern Asia, the Middle East, Europe, North America, South America, and Australia
Perspectives and Conclusion	• Multiple constraints, socioeconomic impacts, environmental impacts, and conclusion

We hope that this book is a valuable resource for both scientific researchers and land managers. We would like to thank all the volunteer contributors to this book for their time and valuable scientific input, particularly at a time when the impact of the Covid-19 pandemic has made working conditions very challenging for many individuals.

Finally, we thank Dr Kathryn Page. Without her persistence, patience, and passion, it would have been a challenge to deliver this book on schedule.

Yash Dang, Ram Dalal, and Neal Menzies

PART A:

INTRODUCTION AND MAPPING

CHAPTER 1

SOIL CONSTRAINTS TO CROP PRODUCTION: AN OVERVIEW

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Abstract

Soil constraints limit crop production worldwide and are responsible for significant yield losses each year. Constraints can be defined as any soil characteristics that limit crop growth and thus negatively impact agricultural production. These can be physical (e.g., surface seals, compaction, waterlogging), chemical (e.g., acidity, salinity, nutrient deficiencies), or biological (e.g., soil borne diseases, declines in vesicular-arbuscular mycorrhizae). Soil constraints may occur singly or in combination and tend to vary both spatially across the landscape and within soil profiles, occurring in both top-soil and subsoil layers. This variability increases management complexity and improvements in our ability to identify what constraints occur, and where these are located, are essential to effectively target management and increase production. Constraint management approaches can generally be grouped into three broad categories, namely a) amelioration, which focuses on the removal of constraints or reduction in their severity; b) agronomic management, which uses techniques to maintain crop production despite the presence of soil constraints; and c) land use change, which is used when amelioration or agronomic management is not logistically or economically feasible. This chapter provides a brief overview of the different types of constraints affecting cropping systems, the impact of these on crop production, and management options available to improve farm productivity and profitability. This acts as an introduction to the more detailed chapters to follow in the remainder of the book.

Keywords: Acidity, alkalinity, salinity, sodicity, compaction.

Introduction

The world's population is expected to increase by around 2 billion people over the next 30 years (United Nations 2019). To feed these people will require us to boost food production, and this must occur even though our capacity to expand the area used for agriculture is limited and production on existing land is threatened by land degradation, water resource scarcity, and climate change. This conundrum means that our existing agricultural systems will need to evolve to produce more food, more efficiently.

Soil constraints currently limit agricultural production worldwide (Bot, Nachtergaele, and Young 2000). A soil constraint can be defined as any soil characteristic that restricts plant growth and lowers crop production. Constraints can be naturally occurring or induced by agricultural management and can either be:

- Physical – for example, surface seals, compaction, waterlogging, and erosion.
- Chemical – for example, salinity, sodicity, acidity, alkalinity, nutrient deficiencies, and the presence of toxic elements.
- Biological – for example, an increase in the incidence of harmful biological agents, such as soil-borne diseases, and a decrease in the activity of beneficial ones, such as vesicular-arbuscular mycorrhizae or earthworms.

Soil constraints can reduce crop growth a) by reducing the movement of water and air into and within the soil profile e.g., due to the development of surface seals or compacted layers that reduce hydraulic conductivity; or b) by restricting a plant's ability to grow and function, e.g., due to the presence of physically or chemically hostile layers that prevent root growth and/or water and nutrient uptake (Dang et al. 2006). This can have a major impact at a farm level by decreasing agricultural output, increasing management costs, and thus decreasing farm income (Orton et al. 2018). It can also impact society at a broader level by decreasing country wide productivity and food security. Many negative environmental problems can also arise from soil constraints. These can include increased runoff and erosion, and thus pollution of surrounding waterways, increased production of greenhouse gasses, and decreased soil carbon storage (Figure 1.1).

This chapter provides a brief outline of the impact of soil constraints on crop production worldwide and the broad management principles and considerations that can be used to develop effective management options.

This outline provides a concise summary that acts as an introduction to the more detailed analysis in the chapters to follow.

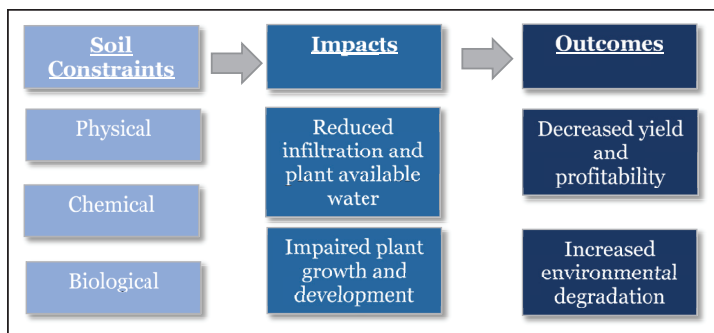


Figure 1.1 Summary of the major types of soil constraints, their impacts, and subsequent outcomes.

Distribution of soil constraints and their impact on crop production worldwide

Recent estimates suggest that only 24% of agricultural land worldwide is without any major soil constraint, and the areas affected are likely to increase by five to seven million hectares annually (Bot, Nachtergaele, and Young 2000). In many regions this can have a large impact on agricultural production. For example, much of Africa and Eastern Europe produce less than 40% of their potential yields, in part due to soil constraints (Pradhan et al. 2015). Similarly, in Australia the constraints of sodicity, salinity, and acidity are estimated to be responsible for annual production losses of nearly A\$ 1,900 million dollars in the wheat crop alone (Orton et al. 2018).

The type and severity of the constraints present vary from location to location and are affected by factors such as climate, soil type, topography, and farming system. This is clearly illustrated in Table 1.1, which shows the areas of land affected by seven common soil constraints and how these vary between major regions worldwide. Constraints may occur singly or in combination and tend to change not only spatially across the landscape but also within soil profiles, occurring in both top-soil and subsoil layers. This increases management complexity and highlights the importance of understanding exactly what constraints occur, and where these are located, to effectively target management and increase production. On a national level this allows management agencies to effectively identify those constraints

Table 1.1 Area of land affected globally by seven common soil constraints. Data reproduced in modified form from Bot, Nachtergaele, and Young (2000)

CONSTRAINT	Sub-Saharan Africa	AREA AFFECTED ('000 KM)					World Total
		North Africa & Near East	Asia and Pacific	North Asia east of Urals ¹	South and Central America	North America	
POOR DRAINAGE	1,903	79	3,083	5,702	2,086	3,388	17,382
LOW CEC	3,716	292	1,105	11	982	0	6,151
AL TOXICITY	4,371	1	3,906	783	8,019	2,219	19,867
HIGH P FIXATION	1,009	0	1,395	0	3,016	1	5,421
SODICITY AND SALINITY	884	780	3,043	2,137	1,115	191	8,369
SHALLOW SOILS	3,007	2,854	4,829	2,796	2,313	2,497	19,133
EROSION HAZARD	3,627	1,185	4,655	3,349	3,923	3,851	21,975

¹ Including Kazakhstan, Kyrgyzstan, the Russian Federation, Tajikistan, Turkmenistan, Uzbekistan

having the most impact on a region's agricultural production and prioritise resources appropriately. On the farm and paddock scale it allows us to identify appropriate management on a site-specific basis. This allows for better resource use to both improve production and the sustainability of the food supply, and maintain environmental quality (Gebbers and Adamchuk 2010).

Management of soil constraints

Once it is understood what constraints occur, and where these are located, knowledge of the most effective techniques to manage crop production is required. Soil constraints rarely occur in isolation and interact in a site-specific way to create growing environments that vary from location to location. Consequently, it is not possible to use a “one size fits all” approach to management, and different combinations of techniques will be appropriate in different situations. However, the management options available can generally be grouped into three broad categories: amelioration, agronomic management, and land use change (Figure 1.2).

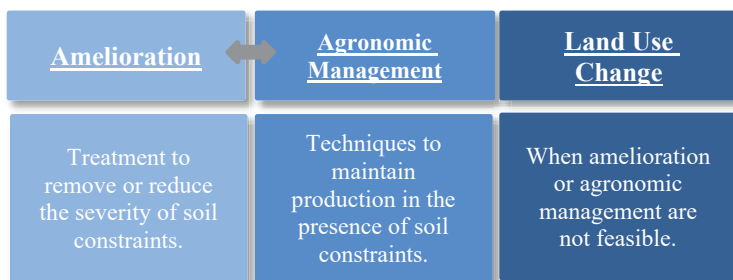


Figure 1.2 Illustration of the three broad approaches available for the management of soil constraints.

Amelioration

Amelioration is one of the most common approaches to the management of soil constraints and typically involves the use of some input to correct or improve a constraint. Common examples can include the application of lime to treat soil acidification, gypsum to treat soil sodicity, fertiliser to treat nutrient deficiencies, and cultivation to treat compaction. Because soil amelioration has been used extensively for many years, a broad knowledge base exists regarding how it can be best applied. However, significant gaps

in our understanding still exist, particularly regarding a) the most appropriate and efficient use of ameliorants where multiple constraints exist, and each constraint needs to be treated before maximum yield can be achieved; and b) how best to use amelioration when constraints occur in a subsoil environment and it is difficult to apply ameliorants.

Agronomic management

Agronomic management refers to the manipulation of farming practices to help crops grow better despite the presence of a soil constraint. This can be used either in conjunction with amelioration, or as an alternative option where amelioration is uneconomic or likely to take an extended time. However, while agronomic management has the potential to improve production on constrained soils, there is often less emphasis on this approach compared to amelioration and improvement in our understanding of its potential is required. Examples of agronomic management can include:

- Accurate identification of constrained areas so that these can either be excluded from production or managed differently where there is a high risk of negative returns. For example, where farmers have a solid understanding of how constraints change across a farm/paddock they can vary the application of inputs (fertilisers, pesticides, ameliorants) to better match soil conditions and expected plant responses.
- The identification and use of plant species or crop cultivars that are best suited to growth in a particular soil. For example, in soils affected by salinity the use of salt tolerant species, such as barley or canola, in preference to more salt sensitive ones, such as chickpea or durum wheat, can help maintain production (Dang et al. 2010). Work on wheat has also indicated that a 10% increase in yield can be observed on lower yielding sodic sites when growing more tolerant cultivars (Schilling et al. 2019).
- The manipulation of cultural practices such as sowing times, row spacings, seeding densities, and tillage management to best manage constraints. For example, raised beds can help increase yields where crop growth is limited by waterlogging (Armstrong, Eagle, and Flood 2015; Robertson et al. 2016). Alternatively, the use of no-tillage with stubble retention can help retain organic matter and improve structural stability (Li et al. 2019; Blanco-Canqui and Ruis 2018).

Land use change

In some instances, soil constraints may be so severe or difficult to manage that sustaining crop production using either amelioration or agronomic management is not logistically or economically feasible. Under these circumstances a move to an alternative land use may be the most appropriate management option. For example, in severely saline soils, converting from cropping to agroforestry or pasture production may be more a more successful option to maintain production (Dang et al. 2010). In the most severe cases, the return of affected areas to native vegetation may also be required.

Conclusions

The successful identification and management of soil constraints is clearly required to maximise agricultural productivity in a world where there is increasing demand for food due to a growing population, but limited opportunity to expand the area used for agriculture. However, the complexity of this task is great, and sophisticated and coordinated management responses are required. This book will provide a comprehensive overview of the types of physical, chemical, and biological constraints that impact crop production worldwide. The extent of these constraints, their effect on crop production, and the most promising approaches to manage their impact will be discussed. In addition, comprehensive case studies of important cropping regions worldwide will be presented to outline the unique management challenges and current state of the art in soil constraint management in different geographical and climatic regions. The social, economic, and environmental impacts of soil constraints will also be discussed and assessed.

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CHAPTER 2

SOIL CONSTRAINT DIAGNOSIS AND MAPPING

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Abstract

Soil constraints significantly impact crop growth and yields. The ability to accurately measure and map these constraints has the potential to greatly improve on-farm productivity. Soil constraints have traditionally been assessed using farmer observations or conventional soil sampling and analysis techniques. Technological advancements have enabled the assessment of constraints using more formal sampling strategies, and proximal and remote sensing technologies. While regional and national soil databases and products contain valuable information about the soil environment, many lack information specific to soil constraints and have limited use on-farm due to their broad extent. The collection of field-scale data via point-based or spatial techniques provides an abundance of raw soil data. However, this information is often not presented in a useful or meaningful way for applied decision-making on-farm. Digital soil mapping (DSM) presents an opportunity to integrate the plethora of grower-sourced and publicly available soil data into useful decision-making tools. As techniques used to measure and map soil constraints continue to improve, the translation of this information into useful decision-making tools remains imperative.

Keywords: Digital soil mapping, soil sampling, soil constraints.

Introduction

Soil constraints have the potential to significantly impact crop growth and yield. Data on the soil landscape is becoming increasingly more accessible across a range of spatial and temporal extents. However, information specific to soil constraints and their impact on crops is either lacking or not portrayed in a way that is useful for guiding informed management decisions in the field. Growers must be able to measure and quantify soil constraints on farm to ameliorate or manage them accordingly.

This chapter will explore the range of soil sampling strategies and techniques available to understand soil physical, chemical, and biological constraints in both the surface and/or subsoil. The University of Sydney farm “L’lara” will be used as a case study throughout to demonstrate the range of soil sampling and mapping techniques available (Figure 2.1 to Figure 2.7). A range of sampling strategies, from arbitrary and ad hoc approaches, through to more formal systematic and stratified sampling methods will be discussed. Sample extraction techniques, including profile pits, augers, and cores, and analysis techniques such as proximal and remote sensing will be explored. Digital soil mapping (DSM) has been developing in recent decades and is an opportunity to integrate grower-sourced and publicly available data. The application of DSM to better understand the nature and extent of soil constraints will be discussed, and future directions will be explored.

Identifying a soil constraint to crop production

There are several ways to identify whether a soil constraint to crop production exists in a particular area. These include informal approaches, such as farmer experience and knowledge, as well as more systematic approaches that use yield data or satellite imagery. In the absence of more temporary constraints (pests, diseases, drought, frost), areas of poor and variable crop growth and yields over multiple seasons often indicate the presence of soil constraints (Lobell et al. 2017; Dang et al. 2011). While farmer observations play a key role in the initial identification of lower yielding areas, these visual assessments are often subjective, and a more formal assessment of this variability is increasingly being sought.

Many growers have access to yield monitor data and produce maps of yield variability (Bramley and Ouzman 2019). This may complement farmer observations and can be used to formally identify lower yielding areas within and between fields. Remote sensing, using satellite or airborne sensors and imagery, also presents an additional source of information to

assess crop variability. For example, Normalised Difference Vegetation Index (NDVI) maps can be used to assess variability in crop vigour, canopy health, and may be used as an indirect indicator of soil condition. This may complement existing yield maps and on-the-ground observations. Multiple years of remotely sensed surrogate yield data may also be used to identify areas affected by soil constraints, where consistently low-performing yields over multiple seasons may infer the presence of a soil constraint when the impacts of more temporary constraints are filtered out (Lobell et al. 2007; Dang et al. 2011).

However, while yield maps may suggest the presence of at least one soil constraint, they cannot identify the nature of the constraint or separate the interactions and relationships between multiple soil attributes. Ground-truthing of visual observations, yield maps, and satellite imagery through the collection of soil data is thus required to identify the exact nature and extent of soil constraints. Attributing crop variability to a single constraint is often difficult due to the interactions between the physical and chemical properties of soil (Adcock et al. 2007). Interrelationships between soil properties also make it difficult to identify the relative effect of individual or multiple constraints, and antagonistic interactions between soil properties may result in yield or growth-limiting conditions, even if each individual soil property does not exceed defined threshold values (Adcock et al. 2007).

Soil sampling and analysis for ground-truthing constraints

Soil sampling strategies

Soil analysis is important for ground-truthing observations of suspected soil constraints. The accuracy with which a soil sample represents the broader soil landscape depends on the number of sampling units, the sampling technique, and the inherent variability of the soil (Cline 1944). Thus, it is important to know both where and how to sample to identify soil constraints.

Since the early 20th century, the inherent spatial variability of soils has been increasingly recognised and is now a key consideration when identifying where to sample. Over time, sampling approaches have become increasingly sophisticated, supported by emerging academic research, extension publications, and agronomic advice (Lawrence et al. 2020). Soil sampling approaches range from ad hoc or arbitrary techniques, through to more formal approaches including systematic and stratified sampling schemes. The ad hoc selection of sampling points “at random” over an area is not considered an effective method of randomisation as it may introduce

strong personal bias and fail to effectively capture variability of the soil (Cline 1944). Prior observations and existing data, such as farmer knowledge, yield maps, or satellite imagery, may be used to arbitrarily guide the location of sampling points to capture sufficient variability. Ad hoc or arbitrary sampling approaches are commonplace due to their convenience and ease of use and depending on the experience and knowledge of the surveyor, may be time and cost-effective, particularly for exploratory investigations. However, unintended bias remains a concern.

Systematic sampling strategies select sample sites based on organised grids, patterns or transects and aim to objectively capture as much variability as possible across the study area and reduce personal bias. However, in practice, systematic sampling approaches using arbitrarily located transects or zig-zag patterns, for example, are likely to produce biased results as they fail to capture a truly randomised sample (Lawrence et al. 2020). The advent of Global Positioning System (GPS) and Geographic Information System (GIS) technologies have facilitated the use of grid-based sampling strategies to geo-locate soil information and produce digital soil maps. Grid soil sampling aims to capture soil variability most objectively by sampling based on a regular pattern of cells (Figure 2.1). However, soils are highly variable through space and some parts of a field may be more inherently variable than others. Grid sampling ignores this variability and has a uniform sampling density irrespective of the underlying variation present. While grid-based sampling can capture sufficient detail required for DSM, the process is time consuming and expensive (Dang et al. 2010). The variability of soil parameters, time, and financial constraints are important considerations when deciding upon grid sizes (Flowers, Weisz, and White 2005). While smaller grid cells can capture more variability (Flowers, Weisz, and White 2005), grid cell sizes may range anywhere from 10's to 100's of metres depending on logistical constraints. For inexperienced or non-technically trained individuals, grid sampling is generally preferred over sampling at random as it can limit potential bias (Cline 1944).

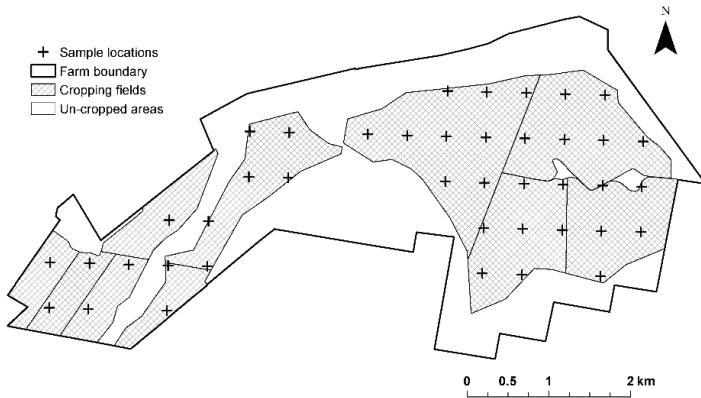


Figure 2.1. The distribution of sampling locations based on a regular-spaced grid pattern across “L’lara”.

Statistical or stratified sampling separates the study area into sub-areas (i.e., strata). Strata may be derived by separating the study region into equal-area strata or by subdivision into more homogenous regions through a range of statistical methodologies including Latin hypercube sampling (Minasny and McBratney 2006) or k-means clustering (Taylor et al. 2003; Filippi et al. 2019) (Figure 2.2). Depending on the complexity of the stratification technique, heterogeneity of the study area, and number of desired strata, information used to delineate strata may include farmer observations (Fleming et al. 2000), yield data (Flowers, Weisz, and White 2005), satellite imagery (Filippi et al. 2019), or a combination of this information alongside other ancillary data. However, there is a trade-off between increased precision and increasing cost and complexity that must be considered when choosing the optimum number of strata.

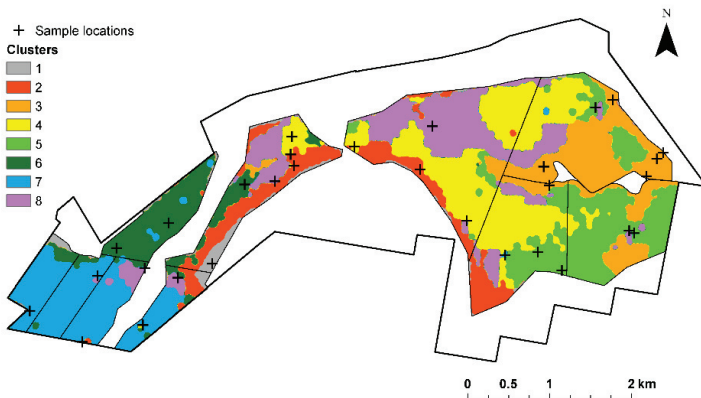


Figure 2.2 The distribution of eight clusters derived from a K-means clustering analysis, and the locations soil sampling sites randomly selected within each cluster across “L'lara” (Filippi et al. 2019).

Soil extraction techniques

Traditional soil sampling approaches treat fields or land management areas as homogenous regions (Dawson and Knowles 2018). As such, bulk or composite samples are created to represent the area of interest. Composite soil sampling is still widely used by farmers and agronomists in the field and is an approach best suited to more homogenous areas where there is existing knowledge about the soil condition (Dawson and Knowles 2018). Composite sampling operates under the assumption that if each individual sample was analysed and averaged, it would give the same result as for the larger composite sample (Tan 2005). The number of soil samples recommended to capture a sufficiently representative composite sample varies for different soil properties measured, although logistical constraints often determine the number of samples that can feasibly be collected (Lawrence et al. 2020). A general rule of thumb is that the more homogenous an area is, the larger the area the composite sample can represent (Dawson and Knowles 2018). Composite sampling is limited by its inability to capture the inherent spatial variability of soils, however, and sampling error will likely increase as sampling areas become more heterogeneous (Tan 2005).

Point-based sampling strategies, including digging profile pits, hand augering, and soil core samples, can be used to capture spatially referenced soil information. Soil profile pits can provide detailed information about a soil's physical, chemical, and biological properties. When using ad hoc or arbitrary sampling approaches, soil profile pits may be dug based on

observations of suspected constraints, historical yield maps, vegetation indices, or pre-existing soil surveys (Taylor et al. 2003). However, the time and labour required to dig and analyse soil profile pits means that they are rarely used on-farm (Earl et al. 2003), as they are impractical for spatial analysis and cannot be used to capture the inherent spatial variability of soils (Taylor et al. 2003).

Soil augers and cores offer cheaper, faster alternatives to soil pits that are highly suited to systematic or statistical sampling strategies. Hand auger sampling is typically used for the assessment of surface-soil or shallow subsurface properties, while mechanised soil cores enable the extraction of a high density of soil samples to depths typically up to 2 m. The mechanised extraction of soil cores is less time and labour intensive compared to hand augering (Earl et al. 2003), however, the vehicle access to fields required for mechanical sampling may be limited at certain crop growth stages and due to agronomic concerns, such as compaction. By geo-referencing the locations of samples, soil maps can be created through digital soil mapping (DSM) procedures.

Analysis of soil samples

Conventional soil analysis techniques continue to be a key contributor to our understanding of the soils physical, chemical, and biological condition. Soil samples may be analysed in situ or be transferred to a laboratory for analysis. Conventional analysis methods include the use of indicator dyes via a colourimetric method such as the Raupach and Tucker (1959) method for the in-field assessment of soil pH, the use of electrodes to measure pH and EC in soil solutions (Tan 2005), or a cylindrical core sampler to measure bulk density (Hao et al. 2008). There is an extensive body of literature available detailing local and national guidelines and recommendations for conventional soil analysis techniques (e.g., Tan 2005).

Proximal soil sensing may also be used to compliment, or replace, conventional techniques by providing quantitative results via portable and/or handheld sensors. Viscarra Rossel et al. (2011) offer a comprehensive review of proximal soil sensing. Here we explore their application for the analysis of soil constraints. Proximal soil sensors can facilitate the (near instantaneous) collection of vast amounts of data using simpler and less time and labour-intensive techniques than conventional approaches (Viscarra Rossel et al. 2011). For example, spectral sensors, such as portable X-Ray Fluorescence (pXRF), use elemental data as a proxy for the analysis of various soil attributes and have been used to assess soil pH (Sharma et al. 2014), CEC (Sharma et al. 2015), and salinity (Aldabaa et al. 2015). Neutron

probes, nuclear density gauges, and low-activity nuclear density gauges have also been used to measure soil moisture and bulk density (Dep et al. 2021). Similarly, Visible-Near Infrared spectroscopy has been used to measure a suite of soil properties including salinity and soil organic carbon (Ahmadi et al. 2021). Today, proximal soil sensing techniques are increasingly shifting from the research and development phase into practical application.

However, while handheld or portable proximal soil sensors have been used with some success, the output from individual sensors is not as accurate as output derived from a combination of multiple sensors and ancillary data (Ji et al. 2019; Aldabaa et al. 2015). Proximal soil sensors do not directly measure soil constraint or attribute values. Rather, sensor outputs serve as a proxy that must be correlated via mathematical processing and modelling. As correlations are often made with calibration samples unique to a particular location or soil type, the use of proximal sensors is restricted by the soil condition(s) it has been calibrated for (Stenberg and Viscarra Rossel 2010). In addition, interrelationships between soil properties requires multivariate calibrations to account for the non-specificity of proximal soil sensor output for individual constraints or attributes (Stenberg and Viscarra Rossel 2010).

Mapping soil constraints

Changes in soil sampling and analysis techniques, from point-based measurements *in situ* to the use of spatial sensors, has rapidly expanded the volume and complexity of information collected about the soil landscape. In its raw form, this data gives minimal insight into the extent and distribution of soil constraints and must be converted into a mapping format before it can provide information about the spatial distribution of soil properties and the nature and extent of constraints across the landscape in a readily interpretable, easy to use format.

Conventional soil maps typically rely on qualitative inferences of soil information, including field observations and aerial imagery, to classify soil properties and features. While these maps served their purpose throughout much of the 20th century, conventional soil surveys and maps have been criticised for their subjectivity and non-replicability (Arrouays et al. 2020). The rise of precision agriculture and increasingly intensive broadacre cropping systems is generating more and more quantitative information and spatial data. DSM has emerged in recent decades following rapid growth and advancements in computational and information technologies (Arrouays et al. 2020). Early DSM simply represented point observations without

spatial interpolation (e.g., Webster et al. 1979), or involved the digitisation of conventional soil maps (Thomlinson 1978). Modern DSM techniques use mathematical or statistical techniques to derive relationships between soil observations and ancillary environmental information for the quantitative prediction of soil properties across a landscape (Arrouays et al. 2020). McBratney et al. (2003) proposed the scorpan framework as a methodology to derive these quantitative soil-landscape relationships and produce digital soil maps. While this is not a standardised procedure, this framework has been widely adopted (Arrouays et al. 2020). In the last decade, DSM has shifted from a research and development phase into operational use, emphasising the need for DSM outputs to present information that is relevant to end-users (Carré et al. 2007).

Publicly available digital soil maps

Access to relevant and accurate soil information is important for making on-the-ground management decisions. Globally, two DSM projects have emerged in response to a demand for up-to-date, spatially referenced soil information with global coverage - SoilGrids (Hengl et al. 2014; 2017) and GlobalSoilMap (Arrouays, Grundy et al. 2014). SoilGrids provides complete, globally consistent gridded maps of soil properties. Released in 2014, the first SoilGrids product mapped soil properties on a 1 km grid using a top-down approach, which uses a global model to predict properties using data derived from legacy soil surveys. Although the coarse spatial resolution fails to capture sufficient variability of soil properties at the local level, the first SoilGrids product demonstrated the ability to map soil properties at a global scale using legacy soil data (Arrouays et al. 2017). In using a top-down approach, SoilGrids can be used to highlight regions where data may be sparse or missing, thus potentially encouraging ongoing collaborative efforts to fill these gaps through bottom-up approaches (Arrouays et al. 2017). An update to SoilGrids was released in 2017, which mapped soil properties across a 250 m grid (Hengl et al. 2017).

The GlobalSoilMap project aims to provide a collaborative, standardised framework for the provision of publicly accessible, continuous global soil information at a high spatial resolution (~90 m) (Arrouays, McBratney et al. 2014). In using a bottom-up approach (i.e., from country to globe), the framework specifies that soil information is included for a minimum of twelve key soil attributes (depth to rock, plant exploitable (effective) depth, organic carbon, pH, clay, silt, sand, coarse fragments, CEC, bulk density, available water capacity, and electrical conductivity) at six standard depth intervals (i.e., 0–0.05, 0.05–0.15, 0.15–0.30, 0.30–0.60, 0.60–1.00, 1.00–

2.00 m), mapped at a 3 arcsecond (roughly 90 m) spatial resolution with estimates of uncertainty (5th and 95th percentile prediction limits) (Arrouays, McBratney et al. 2014). Numerous contributions have been made to the GlobalSoilMap project already at the continental and national level, including for Sub-Saharan Africa (Hengl et al. 2015), Australia (Grundy et al. 2015), France (Mulder et al. 2016), and Nigeria (Akpa et al. 2014).

While digital soil maps from SoilGrids and the GlobalSoilMap project may be useful for regional soil assessments and other land-use applications, the large spatial resolutions of these maps (i.e., 90 m, 250 m, 1 km) is generally too coarse for use at the within field scale. As such, these maps are rarely used by growers on farm. Also, understanding the depth at which a constraint is reached in the soil profile is important for guiding management decisions and agronomic practices. However, the vertical resolution of the standard depth intervals of the GlobalSoilMap increases in coarseness with depth (i.e., 0.30–0.60, 0.60–1.00 m), which makes it difficult to accurately determine the depth to a constraint.

While several agronomically important soil constraints are included in the GlobalSoilMap project, including EC for salinity, pH for acidity/alkalinity (Figure 2.3), and bulk density for compaction, other attributes, such as ESP for sodicity, are not. Other soil attributes, such as Available Water Capacity (AWC), are estimated via the use of pedotransfer functions (PTFs). Region-specific PTFs are being developed and applied to GlobalSoilMap projects across the globe. However, errors derived from the accuracy of PTF input variables and PTF coefficients may result in uncertainty in AWC estimates (Román Dobarco et al. 2019). The incorporation of uncertainty estimates into GlobalSoilMap products and predictions is important for the application of this information in ecological and agricultural modelling and decision making (Román Dobarco et al. 2019).

By presenting soil attribute information at six depth intervals, end-users are provided with an abundance of soil information. However, these soil maps do not indicate the presence or absence of a constraint. Rather, they only represent the spatial distribution of attribute values at specified depth intervals. This abundance of data may overwhelm end-users, and additional processing may be required to understand the nature and extent of a constraint with respect to the application of this data.

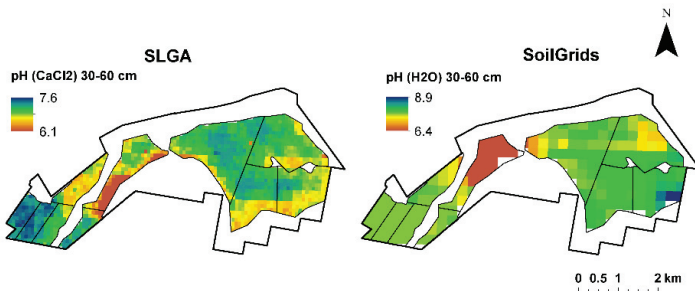


Figure 2.3 Digital soil map soil pH products for the 30–60 cm clay layer at their provided resolution from the Soil and Landscape Grids of Australia (SLGA) (~90 m) and SoilGrids (~250 m) across “L’lara”.

Proximal spatial sensing

Logistical constraints limit the volume of data that can be obtained via conventional soil sampling strategies and handheld/portable proximal soil sensors. Proximal spatial sensing enables the measurements of soil information at high densities, thus filling in these gaps and facilitating soil sampling for high-resolution soil mapping (De Gruijter, McBratney, and Taylor 2010).

Output from proximal spatial sensors (PSS), such as electromagnetic induction (EMI) instruments and ground-based gamma ray spectrometers (GRS), often have good relationships with soil constraints, but are indirect measures. Both EMI and GRS surveys are commonly used PSS for research purposes and commercial application. By sampling above the soil surface, EMI and GRS surveys measure apparent soil electrical conductivity (EC_a) and broad elemental radioactive emissions, respectively, at depth within the soil profile. Measurements are taken at fixed intervals across the study area by sampling on-the-go in swathes, forming a regular transect pattern which is then used to create a map of soil variability (De Gruijter, McBratney, and Taylor 2010) (Figure 2.4). Early uses of EMI surveys were for the assessment of soil salinity (Rhoades and Corwin 1981). Today, EMI surveys have been used for the assessment of numerous soil constraints, including sodicity (Paz et al. 2020), pH (Dunn and Beecher 2007), and bulk density (Al-Gaadi 2012), and have been used at the field (Triantafilis, Ahmed, and Odeh 2002) and landscape scale (Paz et al. 2020).

Other PSS have also been developed for the assessment of soil constraints, including mechanical and electrochemical sensors. Tine-based sensors have been used to measure soil strength to estimate compaction

(Lapen et al. 2002), and the Veris® pH meter is a commercially available proximal spatial sensor that has been used with considerable accuracy to produce maps for the variable rate application of lime to manage soil acidity (Trengeve and Sherriff 2019).

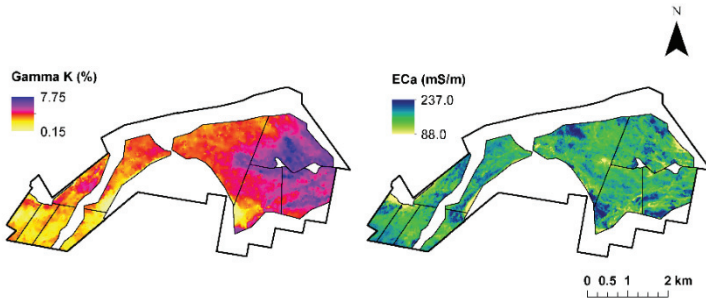


Figure 2.4 Gamma radiometrics K and Soil ECa (0–3.0 m) maps derived from GRS and a Dual EM-21S, respectively, across “L’lara”.

The management of soil constraints involves complex decision-making processes that require multiple layers of (complex) information (Adamchuk and Viscarra Rossel 2010). However, a single sensor can only measure certain soil properties (Ji et al. 2019). A selection of complementary sensors is often required to provide a concurrent quantitative measurement of multiple soil constraints (Grunwald, Vasques, and Rivero 2015). Output from PSS can be used with DSM techniques to create soil maps to inform management decisions. However, few growers actually collect proximally sensed soil data. This limited adoption may be attributed to a poor understanding and perception of the use of these technologies (Bramley and Ouzman 2019), or a lack of integration with other data to enable meaningful interpretation.

Overall, the application of PSS for the assessment of soil constraints is limited in much the same ways as for handheld or portable proximal soil sensors. Readings from PSS are generally an indirect measure of the soil constraint being examined, interrelationships between soil properties limit the number of constraints that can be measured with a single sensor (Ji et al. 2019), and calibration models and equations tend to be site-specific and time-dependent (Doolittle and Brevik 2014).

Publicly available data layers

Publicly available data layers, including remotely sensed satellite imagery, terrain attributes, and other environmental covariates, can also be used in

conjunction with DSM techniques to create maps of soil constraints. Remote sensing via satellite or aerial sensing platforms enables the collection of space-time dense data in a cost and time-effective manner (Grunwald, Vasques, and Rivero 2015). Over time, more and more data layers obtained via remote sensing platforms have been made publicly available as technologies and spatial, temporal, and spectral resolutions have improved. Databases of freely available, medium resolution (i.e., ≥ 10 m pixel) data layers are available, including Landsat, Shuttle Radar Topography Mission (SRTM), Sentinel (from the European Space Agency), and Advanced Very High-Resolution Radiometer (AVHRR). Other databases include higher spatial resolution data layers (i.e., ≤ 5 m) available at a cost, such as RapidEye and GeoEye-1. The recent proliferation of open-access databases has resulted in an increase in the number of studies utilising satellite remote sensing data (Khanal et al. 2020). Both Mulla (2013) and Khanal et al. (2020) provide comprehensive summaries of the applications of satellite remote sensing platforms to (precision) agriculture for the last 20–25 years. Here, we will focus on the application of publicly available data layers to measure and map soil constraints.

Many important soil constraints have been mapped using DSM approaches that utilise environmental covariates obtained from publicly accessible databases. Environmental covariates include a range of terrain attributes such as elevation and slope, and reflectance data in a range of forms such as direct single-band imagery (e.g., red-band) or multi-band processed imagery (e.g., NDVI) (Figure 2.5). By building relationships between available soil data and publicly available environmental covariates, soil constraints including pH, sodicity, and others have been mapped at the national (Roudier et al. 2020), catchment (Filippi, Jones, and Bishop 2020) and farm scales (Dang et al. 2011). In addition, publicly available data layers may also be used to directly measure and map the distribution and impact of soil constraints such as soil salinity (Abbas et al. 2013). There is increasing recognition of the value in combining publicly available and farm-sourced data to create “bespoke digital soil maps” unique to fields, farms, and regions (Filippi, Jones, and Bishop 2020). However, not all soil constraints can be accurately identified from publicly available data layers. Satellite or aerial remote sensing-based studies of soil compaction have yielded poor results, for example (Wells, Stombaugh, and Shearer 2013), and some remotely sensed indices, such as NDVI, are sensitive to the effects of canopy shading, soil colour, and brightness. Post-hoc adjustments of remote sensing products may be required to more accurately assess some soil or vegetation attributes (Xue and Su 2017).

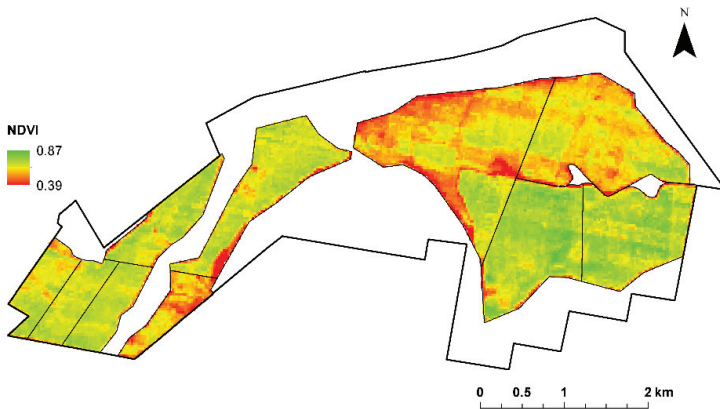


Figure 2.5 Average 95th percentile NDVI imagery for 2000–2018 from Landsat 7 across “L’lara”.

Data layers obtained via remote sensing platforms have been criticised for having spatial and temporal resolutions that are too coarse for application at the field scale for in-season use (Omran and Negm 2020). While this may limit the application of data from some platforms, such as Landsat (with a spatial resolution of 30 m pixels and a 16-day observation cycle), technological advancements have meant that sensors can now measure with sub-meter accuracy with relatively short re-observation times. For example, satellite imagery obtained from high spatial-resolution platforms, including Pleiades (with a 0.5 m spatial resolution and daily observation cycle) and IKONOS (~1 m resolution and a revisit time of 3 days), have been used to assess soil salinity (Laishanov et al. 2016; Allbed, Kumar, and Aldakheel 2014). While significant improvements in the spatial and temporal resolution of satellite remote sensing platforms and publicly available data layers requires higher capacities of data storage and processing requirements (Mulla 2013), the ability to capture a vast amount of high-resolution data has significantly improved soil constraint assessment capabilities.

Techniques used to map soil constraints at the field, farm, and regional scales

Traditional DSM has generally mapped soil attributes at different depth increments in an attempt to capture both the horizontal and vertical variability of soil. Conventional soil sampling approaches divide the soil

profile into horizons. These horizons vary in their number and position within the profile, which may differ between sampling locations. Other sampling strategies sample at specified depth increments down the profile. These increments often vary between different publications; however, more and more studies are producing maps at depths in line with the standard depth intervals of the GlobalSoilMap project (i.e., 0–0.05, 0.05–0.15, 0.15–0.30, 0.30–0.60, 0.60–1.00 m) (Arrouays, McBratney et al. 2014) (Figure 2.6). In the past, inconsistencies in sampling depths between new and/or legacy soil surveys meant that it was often difficult to use this soil data to map soil properties down the profile. Equal-area quadratic smoothing splines (EAQSS) were proposed to overcome this challenge and make horizon data more continuous (Bishop, McBratney, and Laslett 1999). Today, EAQSS are used to map soil properties at a range of depth layers, ranging from 1 cm increments (Filippi et al. 2019) through to coarser intervals (Akpa et al. 2014). While mapping soil constraints at different depth intervals offers a wealth of information about the soil landscape, this may also overwhelm end-users and over-complicate decision making. Attempts to condense this information include simplifying map layers to only the top and/or subsoil (Filippi et al. 2018), or by producing fewer maps at coarser depth intervals. However, these options may fail to capture the true extent of the variability of soil constraints depending on the thickness of each depth layer.

Although maps of soil properties may be useful for identifying relationships between crop yields and soil condition, they are unable to identify the presence or absence of a constraint. The threshold values at which a soil property becomes constraining differ between crop types, growth stages, and other environmental factors. There are published thresholds for the effect of soil properties on crops, usually based on reductions in crop growth and yield. Filippi et al. (2019) proposed the production of a “depth to constraint” map to simplify the plethora of information describing the variability of soils (Figure 2.7). By mapping the depth-to a specific constraint threshold value at a 1 cm vertical resolution, multiple layers of soil information can be integrated into a single map. In recognising both the impact of constraints on crop growth and yields, and mapping constraints at a high vertical resolution, these maps may be of significant benefit for informing management decisions and assessing the efficacy of potential amelioration strategies via a simple and informative decision-making tool.

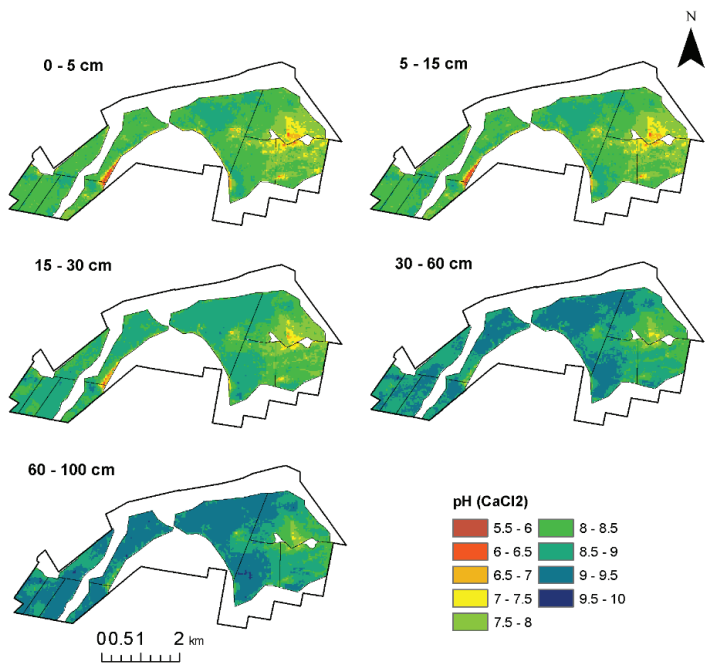


Figure 2.6 Digital soil map products for pH (CaCl₂) at GlobalSoilMap standard depth intervals for “L’lara”.

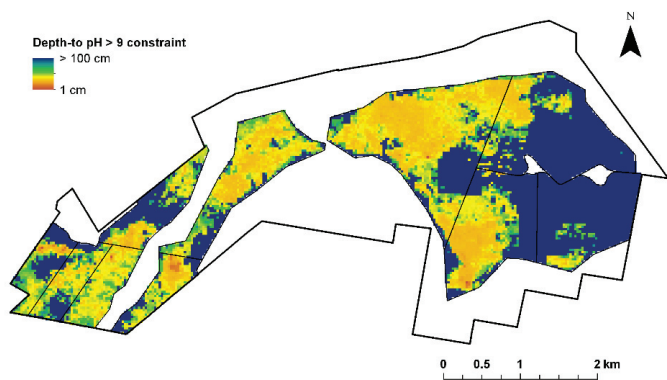


Figure 2.7 Depth-to pH > 9 constraint map for "L'lara".

Soils are highly variable, however, and more than one constraint may be present in the soil environment at a particular location. This may make it difficult to attribute crop growth or yield responses to a particular constraint. Also, several maps showing the distribution of different constraints may present an excess of information that is difficult to interpret. Quantifying the impact of constraints may highlight those that most significantly impact crop growth and yields or identify constraints requiring additional amelioration or management strategies to better optimise crop production. These impacts have been considered in terms of estimating potential yield gaps (Hajkowicz and Young 2005), economic impacts (Orton et al. 2018), or the effect of constraints on soil available water capacity (Sadras et al. 2003). However, most efforts have focussed on spatially mapping the impacts constraints have on potential yield gaps.

In using the boundary-line analysis method, Shatar and McBratney (2004) modelled relationships between crop yields and soil data to identify soil constraints that resulted in the largest yield gaps. These were identified to be the most-limiting constraints to yield and were presented on a patchwork-styled map at the field-scale. Mapping the spatial distribution of most-limiting constraints enables potential amelioration or management strategies to be directed towards those constraints that result in the greatest quantifiable impact on crop growth and yield for a particular area. However, although these maps may be easy to interpret, these may also over-simplify soil-plant relationships and ignore the antagonistic interactions and interrelationships between soil constraints and other properties (Adcock et al. 2007).

Future directions

The measurement and mapping of soil constraints relies on the collection of useful, appropriate, and accurate data. A plethora of data is currently available to growers and farm managers for the on-farm assessment of soil constraints. Moving forward, several key hurdles must be overcome to continue to support informed decision making into the future.

Firstly, the ongoing collection of additional soil data through on-farm soil surveys should be encouraged. There is significant value in using on-farm data in conjunction with publicly available digital soil maps and data layers to create bespoke field, farm, and region-specific maps of soil properties (Filippi, Jones, and Bishop 2020). Through the standardisation of sampling procedures, this farm-sourced data can be integrated with existing soil databases to increase the information available and improve mapping capabilities at a range of scales. However, the return on investment of soil

sampling surveys must be realised. The translation of soil data into useful decision-making tools and frameworks, such as maps of soil constraints, an understanding of yield gaps (Dang and Moody 2016), and a calculation of the return on investment from soil constraint amelioration (Bennett et al. 2021), can highlight the importance of data collection and investment in these strategies. As such, the collection of soil data should not be the goal of technological developments for measuring and mapping soil constraints. Rather, this data should be collected with the purposes of guiding informed decision making. Through the effective translation and communication of this information to end-users, the ongoing sustainability and productivity of agricultural systems can be ensured moving into the future.

Secondly, the benefits of collecting additional soil data cannot be appreciated if the quality and accuracy of this information is poor. Direct sampling methods are time-consuming and expensive (Dang et al. 2010), while proximal and remote sensing options are indirect measures that rely on proxies to reflect soil information. Improving the accuracy of on-the-go sampling and analysis methods and improving direct measurement techniques may improve the accuracy and ease with which data is collected. Further, the ongoing calibration of proximal and remote sensing technologies to different soil types and regions will improve the accuracy of proxy-based sensing systems. Improvements in uncertainty reporting of DSM products is also important for communicating the reliability of this information for decision-making by a range of end-users including farmers and consultants, policy makers, and the broader community (Kidd et al. 2020).

Finally, improving the understanding of the values at which soil properties become constraining to crop growth and yields should be considered. Describing the presence or absence of a soil constraint via the use of arbitrarily defined threshold values can be beneficial (e.g., Filippi et al. 2019). However, crop growth and yield responses to soil constraints are not defined at a single set value. Rather, the magnitude of the impact may vary between different soil types, crops, seasons, and regions. Also, antagonistic interactions between multiple soil properties may promote crop growth and yield-limiting conditions, even if each individual property does not exceed a defined threshold limit. Given that there is abundant yield monitor data relative to soil constraint data, further work should examine using this data source for refining constraint thresholds that are season, site, and crop specific.

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PART B:

PHYSICAL CONSTRAINTS

CHAPTER 3

EFFECTS OF HIGH SOIL DENSITY ON SOIL FUNCTIONS AND CROP GROWTH

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Abstract

Soils can be compact, i.e., have a high dry bulk density, due to natural causes, land use change, or unintended soil management effects (e.g., soil compaction caused by vehicle traffic). Soil compaction decreases pore volume, connectivity, and continuity, adversely affecting many soil properties and soil conditions, which negatively impacts crop productivity and other soil functions. Crop growth is impeded in compact soils, amongst others, due to increased soil mechanical resistance for root growth resulting in decreased root elongation rates and reduced accessibility to water and nutrients, poor soil aeration causing oxygen limitation, and reduced water infiltration and storage capacity resulting in water stress. This chapter gives a brief overview of the causes and consequences of soil compaction, discusses how crop growth is affected in high bulk density soils, and outlines strategies to minimise and alleviate soil compaction.

Keywords: Soil compaction, hard-setting soils, root physiology, root-soil interactions, resource accessibility.

Nature, causes and extent of soil compaction

Soils can be dense, i.e., compact, due to either anthropogenic (e.g., vehicle-induced compaction) or natural reasons (e.g., poorly sorted soils). *Soil compaction* refers to any process that involves an increase in soil bulk

density, and conversely, a decrease in soil total porosity. The state of soil in terms of how dense it is, is described by *soil compactness*. It is important to note that absolute values of soil bulk density are not suitable to assess the state of compactness of a soil, because the textural composition strongly influences bulk density. A certain value of soil bulk density, e.g., 1.5 Mg m^{-3} , has different implications in a sandy soil than in a clay soil. For comparisons across soil textures, it is more meaningful to use a relative bulk density. One such example is the “degree of compactness” (Håkansson 1990), which is given as the ratio of actual bulk density to a reference density. The latter equals Proctor density, or is obtained in a uniaxial compression test at a given stress (Carter 1990; Håkansson and Lipiec 2000).

The main reasons for compact soils are (Daniells 2012; Schneider and Don 2019):

- Natural causes: poorly sorted particle size distribution, overburden under geological time scales (e.g., thick ice sheets, as during the last ice age), or naturally occurring chemical characteristics (e.g., naturally sodic and hard-setting soils).
- Land use change: loss of soil organic carbon resulting in an increase in bulk density (e.g., conversion of grassland to arable land).
- Unintended effects of soil use and management: vehicle traffic (agricultural and forest vehicles, construction machines, and other types of off-road vehicle traffic), grazing animals, and changes to soil salt contents leading to dispersion and destruction of soil structure (e.g., irrigation induced sodic/hard-setting soils).

Compact soils cause problems in different ecosystems including arable lands, grasslands, and forests. Most research related to compact soils in arable systems concerns the unintended effects of soil management, mainly because knowledge of how and when soil management-induced compaction occurs can be used to derive management practices to avoid compaction in the first place (Schjønning et al. 2015). Another reason may be because compaction due to vehicle traffic or trampling animals is a fast process (seconds), while soil structure degradation due to land use change is slow (decades), and geological or pedogenic causes of compaction even slower (millennia) (Or, Keller, and Schlesinger 2021).

Trampling can seriously degrade the structure of soil, but is usually restricted to topsoil layers (Drewry 2006; Hu et al. 2021), while machinery traffic also affects deeper soil layers, which is of great concern because of the very limited recovery potential of subsoils (Schjønning et al. 2015). The

reason for soil compaction due to vehicle traffic is in principle simple and well known: compaction occurs when induced mechanical stresses exceed soil mechanical strength (Horn and Lebert 1994). A main driver of increasing compaction problems, particularly subsoil compaction, are increasing vehicles weights (Keller et al. 2019). For details on how soil stress varies with applied load and soil conditions, and on how soil strength varies with soil properties and conditions, we refer the reader to Keller and Lamandé (2010) and Schjønning et al. (2015).

The mechanisms that create hard-setting soil have received attention primarily in regions where this phenomenon occurs (Daniells 2012; Mullins et al. 1987). Hard-setting soils are characterised by a strong increase in soil strength upon drying. These soils have a hard, “structure-less” mass, and are often associated with high concentrations of exchangeable sodium or low levels of organic matter; the latter can be a cause of poor soil management – for further reading we refer the reader to Daniells (2012).

It is important to differentiate between harmful compaction and the creation of optimum seedbed conditions by re-compaction. The latter is also referred to as firming, and is deliberately done by farmers (e.g., using a roller) to achieve an optimum degree of compactness for crop growth (Håkansson et al. 2011). It has been shown that maximum crop yields are achieved at an intermediate soil compactness (Håkansson 1990), probably because unsaturated hydraulic conductivity and nutrient supply to crops is reduced in soil that is too loose, while mechanical impedance and reduced soil aeration impedes crop development in soil that is too dense. Target values, optimum ranges and thresholds for various soil properties of tilled topsoil were recently compiled by Or et al. (2021).

By nature, subsoils are denser than topsoils, and subsoil compaction is almost exclusively considered to have negative impacts on soil functioning. Less work has been done to establish target and threshold values for subsoil compaction than for identifying ideal seedbed conditions for optimum crop growth. However, the importance of the subsoil for crop productivity as well as other soil ecosystem services (e.g., flood regulation) is increasingly being (re-) realised (Kautz et al. 2013; Alaoui et al. 2018; Keller et al. 2019). Threshold values for the identification of harmful subsoil compaction have been developed in Germany and possibly other countries: Lebert et al. (2007) propose that the subsoil is harmfully compacted when saturated hydraulic conductivity is $< 10 \text{ cm day}^{-1}$, air capacity (air-filled porosity at -60 hPa matric potential) $< 5\%$, and visual soil structure assessment indicates that soil is “dense” or “very dense”. Schneider and Don (2019) evaluated soil compactness based on the so-called packing density (PD), which is given as: $PD = BD + 0.005 \text{ clay} + 0.001 \text{ silt}$, where BD is bulk density in

Mg m⁻³ and clay and silt in mass %. Threshold values for PD have been suggested in the range of 1.75-1.82 Mg m⁻³ (cited in Schneider and Don 2019).

The spatial extent and severity of soil compaction, i.e., how much of the total area of arable land is compacted and how severely, is not well known. Estimates include: 4 Mha in the Australian wheat belt; 25-40% of the total arable land in European countries; and 68 Mha worldwide (see references in Hamza and Anderson 2005 and Keller et al. 2019). However, some of these estimates are outdated. Schneider and Don (2019) estimated the extent, severity, and nature of soil compactness in arable land in Germany based on the packing density, using managed grassland as a reference (i.e., it was assumed that grasslands were not anthropogenically compacted). Their study classified about 70% of all agricultural soils as compact to an extent that restricted root growth. About 30% of the causes of a high degree of soil compactness were associated with land use and soil management. Although data on the extent and status of soil compactness at large scales are rare or uncertain (Hu et al. 2021), there is little doubt that the costs of compaction in terms of loss of soil functioning and reduced productivity are substantial (Graves et al. 2015; Sonderegger, Pfister, and Hellweg 2020). Consequently, soil compaction is considered one of the main threats to soil functioning and productivity by the FAO and ITPS (2015).

While an increased bulk density is the *per se* definition of a compacted soil, it is the effect of this increased bulk density on physical and mechanical soil properties and their consequences for soil condition and soil processes that is the real problem of soil compaction (Horn et al. 1995; Nawaz, Bourrié, and Trolard 2012; Shah et al. 2017; Hu et al. 2021). In general, compaction not only increases bulk density (and conversely, decreases total porosity), but also changes the pore size distribution. There is a preferential loss of macropores, a decrease in pore connectivity and continuity, and an increase in pore tortuosity and volume of isolated and dead-end pores (Horn et al. 1995). These changes reduce water infiltration, saturated hydraulic conductivity, and water storage. This results in poorer soil drainage and increased risk of flood generation (Rogger et al. 2017), and reduced air permeability and gas diffusivity resulting in poorer soil aeration, changed redox conditions (Reiser et al. 2020) and potentially increased greenhouse gas emissions (Ball 2013). Any change in soil pore structure also modifies the physical habitat for soil organisms (Young et al. 1998). Enhanced bulk density implies increased soil mechanical resistance, which slows down root growth (Dexter 1987; Bengough et al. 2006) and the bioturbation rates of burrowing animals such as earthworms (Ruiz, Or, and Schymanski 2015). Moreover, compaction reduces the number of, and accessibility to, surfaces

for nutrient absorption (Horn et al. 2018), and modifies the conditions for organic matter turnover and thus soil organic carbon amounts and distribution (Brevik, Fenton, and Moran 2002). A recent review by Hu et al. (2021) provides a comprehensive overview of how soil compaction affects various soil processes and functions, and identifies knowledge and data gaps (although the review focuses on New Zealand, its content applies to other regions as well).

Effects of increased soil bulk density on root physiology and crop yield

Biophysical interactions at the root-soil interface under high soil bulk density

The physical properties of compact soil, in particular increased soil penetration resistance and decreased water and gas transport capacity, significantly affect the biophysical interactions at the root-soil interface (Figure 3.1). Due to the reduced void space available for the displacement of soil particles, cavity expansion pressure increases in dense soil (Jin et al. 2013; Bengough et al. 2011; Iijima et al. 2003). Moreover, the friction at the root-soil interface may also increase with increasing soil bulk density (Bengough et al. 2011; Kirby and Bengough 2002). Thus, roots are exposed to higher mechanical stress when growing through compact soil (Ruiz et al. 2016). Apart from these biomechanical interactions, soil bulk density strongly governs gas exchange between roots and the surrounding soil. The flux of oxygen (O_2) from the atmosphere through the soil into roots is impeded in soil with high bulk density because of low air-filled porosity and poor continuity and connectivity of the pore space (Keller et al. 2021; Kuncoro et al. 2014). Therefore, cellular O_2 concentrations can decrease below the levels that are needed to maintain aerobic root respiration. Similarly, the flux of carbon dioxide (CO_2) and the gaseous plant hormone ethylene is reduced from the root into the soil in compact environments and these gases may therefore accumulate in roots and the surrounding soil (Pandey et al. 2021; Conlin and Van den Driessche 2000; Fernández et al. 2017).

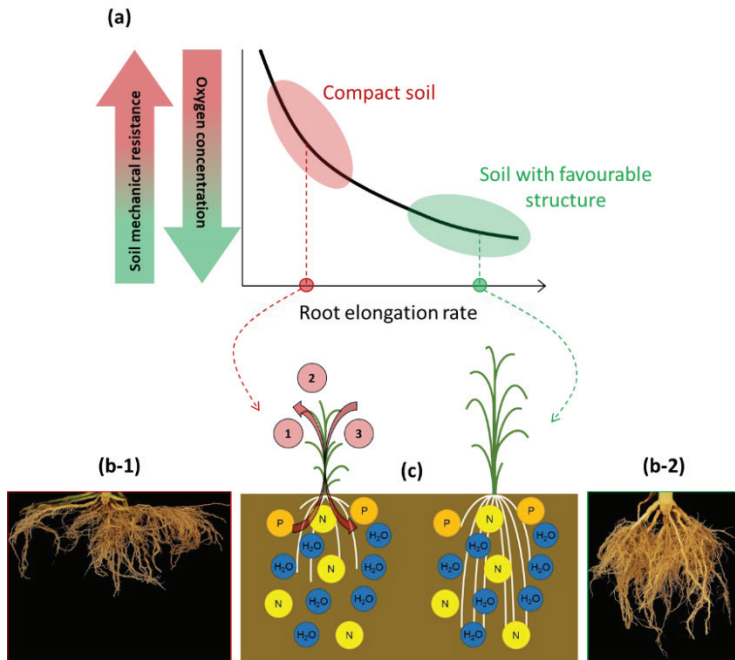


Figure 3.1 Schematic illustration of the impacts of compact soil on crop development: Higher soil mechanical resistance and lower soil oxygen concentration in compact soil reduces root elongation rates (a), which reduces soil exploration and thus limits access to water and nutrients (b and c). Limited access to nutrients and water reduces above-ground development (①), which reduces leaf growth and thus photosynthetic capacity (②). Increased mechanical resistance for root growth and lower oxygen levels increase carbon demand for root growth (③), which in turn reduces the carbon available for above-ground plant development.

High soil bulk density reduces root system expansion and resource accessibility

Root system expansion is slowed down in compact soil (Figure 3.1). It has been shown that in numerous crop species, root elongation rate is reduced and the initiation of lateral roots is delayed when soil penetration resistance is high (Colombi and Walter 2016; Chen et al. 2014; Grzesiak et al. 2014; Pfeifer et al. 2014; Bengough and Mullins 1991). Similarly, a

reduction in soil gas transport capacity and the resulting changes in the root cellular concentration of O₂, CO₂, and ethylene decreases root elongation rates (Saglio et al. 1984; Qi, Marshall, and Mattson 1994; Pandey et al. 2021; Thomson et al. 1992; Malik et al. 2002; Blackwell and Wells 1983). Recent studies suggest that ethylene plays a key role in the regulation of root elongation rate in compact soils. These studies showed that ethylene accumulation in roots may reduce root elongation rate independently from cellular O₂ and CO₂ concentrations, and the mechanical stress that the root tip perceives (Vanhees et al. 2021; Pandey et al. 2021).

Reduced root elongation rate and the delayed initiation of lateral roots limits the volume of soil that crops can explore to acquire water and nutrients (Figure 3.1). Several studies have highlighted that increased soil bulk density limits the access of plants to water and nutrients in soil, resulting in decreased acquisition of these vital resources (Araki and Iijima 2005; Colombi et al. 2018; Lynch and Wojciechowski 2015; Valentine et al. 2012; Nosalewicz and Lipiec 2014). An increase in soil bulk density may also affect water and nutrient availability, although study results are inconsistent and may strongly depend on soil type (Archer and Smith 1972; George et al. 2011; Kristoffersen and Riley 2005; Atwell 1990a; Pfeifer et al. 2014). However, overall, limited resource accessibility is considered to be the major constraint of increased soil bulk density on crop productivity (Bengough et al. 2011; Valentine et al. 2012; Colombi and Keller 2019).

Carbon requirements for soil exploration increase in compact soil

Compact soil also interferes with carbon partitioning patterns between above- and belowground plant organs. High soil penetration resistance and poor soil aeration often induce root thickening, which is mainly driven by an enlargement of the root cortex (Lipiec et al. 2012; Striker et al. 2007; Atwell 1990a). Increased root diameters are associated with greater root porosity and thus improved root aeration in soils with poor aeration (Broughton et al. 2015; Pedersen et al. 2020). Thicker roots are less likely to buckle and root thickening reduces the mechanical stress at the growing root tip (Kirby and Bengough 2002; Chimungu, Loades, and Lynch 2015; Materechera et al. 1992). Moreover, plants may increase sloughing of root cap cells and excretion of mucilage into the rhizosphere to decrease the friction at the root-soil interface (Bengough and McKenzie 1997; Iijima, Griffiths, and Bengough 2000). These root phenotypic adjustments improve the ability of plants to withstand the physical conditions occurring in dense soils. However, the carbon requirements for root growth also increase in

response to the root adjustments (Iijima, Griffiths, and Bengough 2000; Colombi et al. 2019; Atwell 1990b). Furthermore, poor soil aeration may induce a shift in root metabolism from aerobic respiration to anaerobic fermentation, which contributes to increased root carbon consumption (Bailey-Serres, Lee, and Brinton 2012; Colombi and Keller 2019). Hence, plants need to invest greater amounts of carbon into root growth and soil exploration in compact soil, and as a result, the carbon available for aboveground plant growth and crop yield formation decreases (Figure 3.1).

Plants may preferentially exploit soil macropores in soils with high bulk density

Plants may use biopores and other soil macropores to avoid the adverse physical conditions of dense soil (Colombi, Braun et al. 2017; White and Kirkegaard 2010; Atkinson et al. 2020; Stirzaker, Passioura, and Wilms 1996; Pfeifer, Kirchgessner, and Walter 2014). Macropores provide a pathway of least resistance and improve water infiltration and soil aeration. It has been shown that roots grow actively towards soil macropores (Colombi, Braun et al. 2017; Atkinson et al. 2020), but the mechanisms underlying such directed root growth are not yet fully understood. Oxygen concentration in soil air is higher in and around macropores than in bulk soil, while soil penetration resistance decreases in the vicinity of soil macropores. Thus, “oxytropism”, referring to the response to gradients in oxygen concentration (Colombi, Braun et al. 2017; Pfeifer, Kirchgessner, and Walter 2014), as well as “trematotropism”, referring to the response to gradients in penetration resistance (Atkinson et al. 2020; Dexter 1986), may explain preferential root growth towards macropores. Exploiting macropores as a pathway of least resistance, or as a source of oxygen, fosters root growth in dense soil, thereby improving soil exploration and aboveground plant growth (Colombi, Braun et al. 2017; Landl et al. 2019; White and Kirkegaard 2010). These studies highlight the crucial role of soil macropores, such as biopores, for whole plant growth and crop productivity in compact soil.

Managing compact soil: minimisation, alleviation, and adaptation

Minimising or restricting soil compaction

Evidence from field experiments shows that the functions of compacted soils, especially subsoils below annual tillage depth, are impaired for decades (Hakansson and Reeder 1994; Peng and Horn 2008; Berisso et al.

2012). In other words, in contrast to the formation of compaction, which may happen within seconds, recovery of soil structure and function is very slow. Therefore, avoiding soil compaction should always be the primary goal (Schjønning et al. 2015). Avoiding soil compaction is simple in principle and can be achieved by ensuring soil stress does not exceed soil strength for any field operation. In practice, any combination of high vehicle load and low soil strength should be avoided. Subsoil stress increases with increasing wheel or track load, while tyre (track) dimensions and tyre inflation pressure primarily affect stress in the upper soil layers (Keller and Arvidsson 2004). Subsoil stresses are also particularly high during conventional in-furrow mouldboard ploughing (Keller, Trautner, and Arvidsson 2002; Keller et al. 2019). Soil strength generally decreases with increasing soil moisture (Schjønning and Lamandé 2018; Culley and Larson 1987). However, complete avoidance of soil compaction may be illusory, and hence the aim may be to minimise (rather than avoid) soil compaction. Decision support tools such as Terranimo® [www.terranimo.world (Stettler et al. 2014)] can be used for estimating compaction risks. Another solution is to restrict compaction to small areas such as tramlines, as practised in controlled traffic farming. This has been successfully adopted in certain regions of the world (e.g., Australia), with positive effects on crop yields, the environment, and farm economics (Antille et al. 2019; McPhee et al. 2020). While the methods outlined here are management strategies to avoid, minimise and restrict future compaction, they do not solve problems related to already compacted or naturally compact soils.

Alleviating soil compaction

Tillage of topsoil has been practised for millennia, and one of the objectives of (top)soil tillage is to loosen the soil. Hence, compacted topsoil can be loosened by tillage. Nevertheless, it is important to note that tillage is not simply the “reverse function” of compaction (Keller et al. 2021): it has been shown that it takes several years to recover compacted topsoil even when the soil is annually tilled (Arvidsson and Håkansson 1996; Weisskopf et al. 2010).

Mixed responses of crop yields to subsoiling (subsoil loosening) have been reported (Chamen et al. 2015). Lack of, or negative effects, of subsoiling can be associated with the fact that the subsoiling itself was not successful, or that soils quickly re-compacted after subsoiling. Moisture conditions at subsoiling are critical for the success of the loosening operation: if soil is too wet at the time of subsoiling, no loosening or structural improvement will happen. In contrast, soil will be homogenised and

weakened when tilled under too wet conditions (Spoor and Godwin 1978). In many regions, the only way to adequately dry subsoils to a soil moisture suitable for subsoiling is with the help of plant roots (Dexter 1991).

Re-compaction after subsoiling is a frequent phenomenon because subsoiled soil structure is initially weak (Chamen et al. 2015). Therefore, gentle post-subsoiling management is key for long-lasting effects. This may include usage of light vehicles and adapted vegetation e.g., grass-legume mixtures for a few years before re-introducing the land to crop production [see e.g., Swiss guidelines for recultivation (BAFU und BLW 2013)]. Apart from using deep-rooting plants, liming or incorporation of organic material may also be options for stabilizing the loosened subsoil (Gill, Sale, and Tang 2008; Leskiw, Welsh, and Zeleke 2012; Getahun et al. 2018). Where subsoil compaction has formed due to sodicity, the stabilisation of soil structure using gypsum, or other ameliorants capable of preventing dispersion, is essential to maintain structural improvements (Hamza and Anderson 2005).

Because the rate of success is low for subsoiling, it has been suggested that creating slots or fissures could be a better alternative (Spoor, Tijink, and Weisskopf 2003). Hereby, only a limited volume of soil is disturbed, but the fissures or slots still improve infiltration capacity and soil aeration, and foster root growth. Similarly, the introduction of artificial vertical macropores into dense soil may improve the physical conditions for root growth (Mori, Fujihara, and Yamagishi 2014; Colombi, Braun et al. 2017). Since such vertical macropores are relatively stable under uniaxial compression (Schäffer et al. 2008), they could be a promising option to improve the functionality of dense soils and enhance crop productivity.

Another alternative, or complementary approach, to mechanical subsoiling is to include plant species with a high ability to penetrate dense soil layers into crop rotations (Dexter 1991). Cover crops, and perennial plants such as tall fescue, with deep and extensive root systems have received particular interest in recent years (Inagaki et al. 2021; Pulido-Moncada et al. 2020). Roots may reach deep into the soil profile, and after roots have decayed, a network of biopores remains that can be exploited by subsequent crops (White and Kirkegaard 2010; Athmann et al. 2013). While plant roots can create new pores, thereby improving the functional properties of soil such as soil aeration or water infiltration, they do not usually de-compact soil. De-compaction, i.e., a reduction in bulk density and an increase in soil total porosity, is only possible through soil expansion or upwards transport of soil (e.g., by earthworms) (Keller et al. 2021).

Root biomass and root exudates are also the major input source of organic matter in arable soils (Kätterer et al. 2011). Thus, root growth into dense layers may increase soil organic matter content (Inagaki et al. 2021),

which improves soil properties that are essential to plant growth such as water holding capacity and soil biological activity. Moreover, water uptake by plant roots may induce soil shrinkage and subsequent soil crack formation in the vicinity of roots (Materechera, Dexter, and Alston 1992; Colombi et al. 2021). These cracks improve soil water infiltration rates and increase the surface area available for water and nutrient uptake in dense soil (Horn et al. 2018; Zhang et al. 2014). Hence, plant species that can develop extensive root systems in dense soil may potentially be used to improve crop growth conditions in compact soil.

Adapting crops to the physical conditions of compact soil

Another approach to improve the productivity of compact soil is to adapt crop germplasm to the physical conditions occurring in dense soil through plant breeding. Exploiting root trait diversity and subsequently selecting for specific root traits has great, yet largely unused, potential to alleviate the adverse effects of compact soil on crop growth (Lynch and Wojciechowski 2015; Colombi and Keller 2019). Until now, most studies addressing the potential to adapt crops to high soil bulk density have focused on monocots such as wheat, maize, and rice. However, significant root phenotypic diversity also exists between different genotypes in dicot species such as common bean (Lynch 2019; 2015), suggesting that this potential is not limited to monocotyledonous crop species.

A sharp root tip opening angle decreases the mechanical stress acting on the root tip, which is related to increased root elongation rates under high penetration resistance (Colombi, Kirchgessner et al. 2017). Roots may also be anchored to the surrounding soil through root hairs, which enables roots to exert greater penetration force while growing through soil, which increases root elongation (Bengough, Loades, and McKenzie 2016; Haling et al. 2013). Furthermore, root anatomical traits can have potentially beneficial effects on root growth under high soil penetration resistance. The occurrence of multiseriate cortical sclerenchyma (referring to the small and lignified cells in the outer region of the root cortex) reduce the risk of root buckling, which facilitates root growth and soil exploration under high soil penetration resistance (Schneider et al. 2021). Fewer cortical cell files, larger average cortical cell size, and the increased occurrence of root cortical aerenchyma, have been associated with deeper rooting under high soil penetration resistance (Vanhees et al. 2020). Root cortical aerenchyma, which are air-filled structures in roots connecting roots to the aboveground atmosphere, can also act as an internal root aeration system. Therefore, root cortical aerenchyma facilitate gas (O_2 , CO_2 , ethylene) exchange between

roots and the atmosphere, which may alleviate the adverse effects of poor soil aeration on root growth (Pedersen et al. 2020; Thomson et al. 1992; Broughton et al. 2015).

Breeding efforts that aim to adapt crops to compact soil may also target root traits that reduce the carbon demands of soil exploration and thus decrease the respiratory requirements of roots. It has been shown that the carbon required for root elongation under high soil bulk density decreases with increasing cortical cell size and decreasing number of cortical cell files (Colombi et al. 2019). Similarly, the carbon requirement of roots decreases with root cortical senescence and the increased occurrence of root cortical aerenchyma (Schneider et al. 2017; Saengwilai et al. 2014). Due to their function as an internal root aeration system, cortical aerenchyma may increase O₂ levels in roots and aerobic root respiration can be maintained despite low soil gas transport capacity (Bailey-Serres, Lee, and Brinton 2012; Pedersen et al. 2020). Ultimately, ensuring aerobic root respiration helps to reduce the carbon requirements of soil exploration under high bulk density.

The diversity of root traits and the effect of these traits on root growth rate and the carbon required for root growth indicates that crop productivity on dense soil can potentially be improved through crop breeding. Plants can enhance the functioning of dense soil through bioturbation, organic matter inputs, and water extraction. It is therefore likely that novel crop cultivars that are adapted to compact soil will also improve soil structure in the long term.

Conclusions

Compact soils are a global phenomenon and can form due to land use change, agricultural management, as well as natural processes. Dense soils are characterised by high mechanical impedance, and poor gas and water transport capacity. These impaired soil physical conditions limit root growth, which limits the access of plants to water and nutrients in soil. Moreover, plants need to invest greater amounts of carbon into root growth in dense soil, which reduces the amount of carbon available for crop yield formation. The different options to manage compact soil can be grouped into three categories: i) avoidance/minimisation, ii) alleviation, and iii) adaptation. Avoidance refers to management systems that aim to reduce the risk of soil compaction (low wheel loads, wide low-pressure tyres) or constrain vehicle traffic to small areas (tramlines, controlled-traffic farming). Alleviation consists of approaches that improve conditions for root growth in dense soil, such as mechanical or “biological” subsoiling.

Lastly, adaptation refers to the selection of root traits through plant breeding to improve the tolerance of crops to the conditions occurring in compact soil. We suggest that combinations of options belonging to these three categories are needed to sustainably increase crop productivity and to improve the overall functioning of dense soil in arable systems.

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CHAPTER 4

SOIL WATERLOGGING: CAUSE, IMPACT, AND MANAGEMENT

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Abstract

The sensitivity of global food production to frequent weather anomalies (e.g., flooding, soil waterlogging, and drought) has emerged as a threat amid climate change. Among abiotic stresses, flooding is the most damaging stress, affecting 10-16% of agricultural lands worldwide. Flooding alone caused around two-thirds of the total crop losses between 2006-2016 and is predicted to increase considerably in the future. In this chapter, “flooding” refers to the condition where the whole plant or plant parts are submerged underwater, and “soil waterlogging” refers to the condition of complete soil pore saturation with moisture. In addition to direct crop damages, flooding and waterlogging also results in several environmental concerns, such as declines in surface and groundwater quality. It is, therefore, critical to manage flooding and soil waterlogging to mitigate adverse impacts on crop production systems by understanding the causes, impacts, and management strategies. Soil waterlogging or flooding can be caused and influenced by increased precipitation frequency, precipitation after irrigation, over-irrigation, soil compaction, shallow water tables, poorly drained soils, and land topography. Soil waterlogging or flooding stresses crops mainly through reduced oxygen levels near the rhizosphere, altered soil physicochemical properties, and reduced nitrogen uptake. Management for

soil waterlogging or flooding requires a complementarily integrated approach of agronomic management, genetic improvements, and engineering measures. Major soil management for waterlogging or flooding includes raised bed farming, nutrient management, drainage systems, and controlled farm traffic. The most used crop management techniques for waterlogging management are flood-tolerant varieties, cover crops, altering planting dates, and application of plant growth regulators. Despite the development of several management strategies, there is enormous scope for research to develop more flood-resistant varieties, integrate various precision agricultural techniques, and utilise remote sensing and crop modelling for mitigating waterlogging stress and losses in agricultural production systems.

Keywords: Drainage, flooding, waterlogged soils, raised bed farming, cover crops, nutrient management, plant growth regulators.

Introduction

The changing climate is creating significant constraints for agricultural crop production and threatens global food security (Aderonmu 2015). Crop production is highly sensitive to climate alterations, as different crops require different optimum weather conditions throughout the growing period. Frequent extreme weather events, such as drought and flooding, have been predicted due to climate change (Westra et al. 2014). Flooding is one of the most damaging abiotic stresses for plant growth and constrains crop production (Dat et al. 2006; Smith 2013).

“Flooding” mainly refers to the condition where the complete plant, or part of the plant, is underwater (Figure 4.1) (Bailey-Serres, Lee, and Brinton 2012). Flooding can be further classified into waterlogging, partial submergence, and complete submergence (Sullivan et al. 2001). “Waterlogging” is defined as a condition where soil pores are filled with water resulting in anaerobic conditions for the plant’s root system. The term “partial submergence” refers to a condition where an aboveground water layer forms in addition to soil pore saturation, resulting in submergence of the root zone and part of the above-ground plant parts. The term “complete submergence” is used to describe the condition where the complete plant is underwater. Globally, flooding affects around 10-16% of agricultural lands (Yaduvanshi et al. 2012) and was the cause of almost two-thirds of all crop damages and losses in the period between 2006 and 2016, costing billions of dollars (FAO 2017). Waterlogging affects 16% and 10% of the agricultural lands of the United States and Russia, respectively (Boyer

1982; Yaduvanshi et al. 2012). In Australia, waterlogging results in around AU\$180 M in lost grain production annually (Pang et al. 2004). It is predicted that crop production damages will result in a loss of around \$3 billion annually by 2030 due to increasing extreme weather events (both flooding and droughts) amid the changing global climate (Parry et al. 2007; Rosenzweig et al. 2002).

The changing global climate causes waterlogging events to be more severe, unpredictable, and frequent (Jackson and Colmer 2005; Westra et al. 2014). Saturated soil pores under waterlogged conditions reduce gas diffusion by several times, which leads to oxygen-limited conditions triggering negative impacts to plant growth and development (Colmer and Greenway 2010). Nitrogen losses are increased due to increased leaching, denitrification, and runoff, which can adversely impact crop production (Halvorson et al. 2014; Robertson and Vitousek 2009; Kopyra 2004). Nutrient runoff and leaching losses can also degrade surface and groundwater quality (Singh et al. 2017; Kaur et al. 2017; Kaur et al. 2020b; Valipour 2014). Therefore, it is crucial to understand the causes, impacts, and management of soil waterlogging for the development of regional and global mitigation strategies. This will require a complementary and integrated approach that includes agronomic management, genetic improvements, and engineering measures (Manik et al. 2019).

Causes and Impact of soil waterlogging

Causes

Soil waterlogging or flooding (Figure 4.1) can be caused by increased precipitation magnitude and frequency, along with multiple wet days, rainfall following irrigation, over-irrigation, soil compaction, shallow water tables, poorly drained soils with high clay content or restrictive sub-surface clay layers (e.g., claypan soils), sodicity, and land topography. Extreme precipitation events are likely to increase in the future, mainly due to an increase in greenhouse gas emissions (Cubasch et al. 2001). For example, the frequency of heavy precipitation has increased by 50% during the 20th century in the upper midwestern regions of the United States (Groisman, Knight, and Karl 2001). In general, regions receiving 1000 mm mean annual precipitation experience around 40% more precipitation during extreme wet years than in average years (Knapp et al. 2015). In poorly drained soils, runoff from upslope can result in waterlogged conditions in low-lying areas (Hanna, Harlan, and Lewis 1982; Singh, Williard, and Schoonover 2016). Though most waterlogging affected soils have excessive moisture due to

natural causes, anthropogenic perturbances in land use and modification of stream channels and hydrology further exacerbate soil waterlogging.

Identifying waterlogging, especially in waterlogging-prone areas, is crucial as waterlogging can be a potential problem even without water appearance on the surface. Heavy-textured (clayey) soils and duplex (coarse-textured topsoil with clayey subsoil) soils are most susceptible to waterlogging (Cox and McFarlane 1990; Shaw et al. 2013). Waterlogging in heavy-textured soils occurs when surface ponded water saturates the soil profile from the surface downwards, while in duplex soils, saturation occurs from the subsoil up towards the surface due to the inhibition of drainage by the clayey subsoil (McFarlane and Cox 1992; Bakker et al. 2005). For example, heavy clay soils, especially with shrink-swell capacity clays (e.g., Vertisols), are more susceptible to waterlogging during heavy irrigation or rainfall (Kaur et al. 2020a; Kaur et al. 2016). Heavy clay soils with naturally- or anthropogenically-induced poor soil structure (e.g., soil compaction from heavy farm traffic, puddling for rice) can reduce downward water movement because of reduced infiltration, permeability, and water flow in the soil profile. This can lead to a shallow perched water tables in which water accumulates near the soil surface, ultimately resulting in waterlogging (Batey 2009). Waterlogging can also be a result of natural restrictive layers in the soil profile (e.g., claypan and fragipan) that limit water infiltration and flow in the soil profile. Globally, around 290 million ha of soils have restrictive subsurface layers (USDA-NRCS 2006). In addition, waterlogging is also common in sodic soils as the dispersive nature of these soils results in restricted internal drainage (McIntyre et al. 1982).

Impacts

For optimum plant physiological performance, the soil should have adequate proportions of air and water throughout the plant growing period (Crawford 1982). Waterlogging leads to hypoxic or anoxic conditions that can reduce crop growth, mainly due to lower oxygen availability in the rhizosphere (Shaw et al. 2013; Lee et al. 2007, Drew, Sisworo, and Saker 1979). Hypoxic or anoxic conditions lead to several changes in soil physicochemical properties and biological activity due to changes in soil oxygen diffusion (Ponnamperuma 1984), soil redox potential (Eh) (Boivin et al. 2002), and soil pH (Greenway, Armstrong, and Colmer 2006). The Eh is considered the primary indicator of soil chemical alterations during waterlogging, and generally declines under waterlogged conditions (Pezeshki and DeLaune 1998; Kaur et al. 2016). The normal Eh range of soils varies from -300 to +900 mV as compared to -300 to +400 mV in

reduced soils (Sahrawat 2005). The Eh also affects the soil nutrient availability and concentration (Pezeshki 2001). Reduction in Eh leads to excess electron exchange (e.g., Fe^{3+} to Fe^{2+} and Mn^{4+} to Mn^{2+}), which increases the solubility of iron and manganese (Singh and Setter 2017) and can damage plant roots (Marashi 2018; Sharma et al. 2018; Jones and Etherington 1970). The production of ethanol, acetaldehyde, and some organic acids (e.g., acetic, formic, and lactic acids) are favoured under reduced conditions in soil (Parent et al. 2008). Soil pH alterations under waterlogging conditions depend on soil type (alkaline vs. acidic soils) (Sahrawat 2005) and can take a variable amount of time depending on organic matter content, soil chemistry, and temperature (Kongchum 2005). Soil pH decreases in alkaline soils due to CO_2 accumulation but increases in acidic soils due to proton consumption (Sahrawat 2005). Soil pH changes also influence many processes including soil organic matter turnover, mineralisation, and nitrification, affecting nitrogen transformation and availability (Probert and Keating 2000).

Waterlogging reduces stomatal conductance (Folzer et al. 2006; Parent et al. 2008) and leaf chlorophyll content (Mutava et al. 2015), which decreases transpiration and photosynthetic capacity, respectively, leading to substantial yield loss (Wollmer, Pitann, and Mühling 2018; Kaur et al. 2020a). Waterlogging also leads to reduced leaf growth and increased wilting (Reicosky et al. 1985). In addition to crop physiological and morphological responses, waterlogging also causes yield loss due to increased susceptibility to crop diseases and other stresses (Siddique, Walton, and Seymour 1993; Balerdi, Crane, and Schaffer 2004). Soil waterlogging results in lower plant nitrogen absorption and transportation (Huang et al. 1994) and increased soil nitrogen losses through denitrification, leaching, and runoff (Kopyra 2004). This reduced nitrogen availability and uptake negatively affect N-dependent crop productivity (Jaiswal and Srivastava 2018). Yield losses from soil waterlogging can vary from 1 to 100% depending on the plant growth stage in which waterlogging occurs, with different plant responses observed at different stages (Kaur et al. 2020a). For example, losses of 17-43% and 50-56% in soybean yields were observed when waterlogging occurred during vegetative versus reproductive stages, respectively (Scott et al. 1990; Oosterhuis et al. 1990).

The mechanisms for plant tolerance to soil waterlogging have been reviewed in detail by many recent studies including Sairam et al. (2008), Pan et al. (2020), Herzog et al. (2016), Kreuzwieser and Rennenberg (2014), and Shaw et al. (2013). Waterlogging-induced anoxia or hypoxia greatly inhibits plant growth but the responses and tolerance to waterlogging vary among plant type, species, and varieties. For example, wheat (*Triticum*

aestivum) roots are more tolerant to waterlogging than lupin (*Lupinus* sp.) roots (Bramley et al. 2011), while yellow lupin (*L. luteus*) has a higher tolerance to waterlogging than narrow-leaved lupin (*L. angustifolius*) (Davies, Turner, and Dracup 2000). Tolerance to waterlogging also differs with crop growth stage. In wheat, tolerance to waterlogging varies as booting stage < jointing stage < tillering stage < grain filling stage (Bao 1997; Setter and Waters 2003).

Plants use several different mechanisms to adapt to waterlogging, including stem enlargement, adventitious root formation, quiescence, and aerenchyma formation in the root cortex (Bailey-Serres, Lee, and Brinton 2012). For example, winter wheat, soybean, and maize crops use aerenchyma and adventitious roots strategies (Rhine et al. 2010; Setter and Waters 2003; Zaidi et al. 2004), while lowland rice copes through shoot elongation and escape strategies (Fukao et al. 2006). Quiescence is utilised as a survival strategy under excess water stress by several plant species including *Arabidopsis species* (Vashisht et al. 2011), *Oenathe aquatic* (Mommer et al. 2006), and *Rorippa species* (Akman et al. 2012).



Figure 4.1 Flooded soybean trial in the Mississippi Delta.

Options available to manage waterlogging to improve farm productivity

There are various soil and crop management strategies that help mitigate soil waterlogging. Sustainable waterlogging management practices aim to maintain soil structure over the long term to maintain crop production and

environmental quality (Belmonte et al. 2018; Bogunovic et al. 2017). Typically, agronomic practices aim to increase aeration in the root zone, either by improving drainage or by elevating the planting beds (Nguyen 2018). In this section, agronomic management practices that help reduce the waterlogging stress on plants are discussed.

Soil management

Raised bed farming

Raised beds can improve drainage by maintaining a favourable soil moisture status that can increase crop yields. Raised beds reduce waterlogging via shallow drains, or furrows, that are installed at fixed intervals that are approximately 15-20 cm wide. The raised beds allow water to drain away from the plant root zone, which reduces the probability of waterlogging damage. Raised beds enhance soil hydro-physical properties, including macroporosity, infiltration, and aggregate stability (Hussain et al. 2018). Multiple research studies have reported significant yield increment with the use of raised beds compared to flat seedbed planting (Acuña, Dean, and Riffkin 2011; Bakker et al. 2007). However, raised beds also have some disadvantages, including: the cost and operation of machinery; difficulty in controlling seeding depth and seed placement on beds; problems with the management of drainage water; limited effectiveness under high water table conditions; livestock damage to raised beds; difficulties with stubble and fodder management; chemical contamination into waterways through leaching; and difficulties with weed management (Bakker et al. 2007). These disadvantages can reduce the suitability of raised beds for some farming systems.

Controlled farm traffic

This management aims to protect soil structure from inconsistent and extensive uncoordinated farm traffic (Hamza and Anderson 2005). Controlled farm traffic is a crop production system where the cropping area and the farm machinery traffic tracks are distinctly marked, divided, and followed (Taylor 1983; Raper 2005). Therefore, the cropping area always remains unaffected by vehicle tracks, while the traffic tracks become a compacted zone, which increases the efficacy of farm machinery operations (Taylor 1992). This helps prevent any adverse effect on a wide range of soil hydro-physical characteristics including porosity, water infiltration, drainage, amenability to crop sowing, establishment and growth, and soil

gaseous exchange (Gasso et al. 2013). The controlled farm traffic can reduce waterlogging occurrence, compaction, erosion, runoff and nutrient loss, and increase crop yield (Chamen 2015).

Need-based deep tillage and manure application

Deep tillage involves the strategic and need-based use of deep tillage equipment, such as ripper, rotary, moldboard, or disk ploughs, to help loosen compact soil layers, especially clay subsoils, to improve subsoil drainage and thereby reduce waterlogging (Renton and Flower 2015; Roper et al. 2015). The benefits of this practice may last for 3-10 seasons (Roper et al. 2015). However, this technique is not effective in hostile sub-soils such as sodic, acidic, or saline soils. In such cases, addition of gypsum or lime may help in stabilizing the soil (Henry et al. 2018; Matosic et al. 2018). Another practice to reduce waterlogging is by application of large volumes of organic matter in and above the clay layers, which is commonly called “subsoil manuring” (Gill et al. 2009). The addition of organic amendments in the form of manure can significantly improve soil hydro-physical characteristics and crop growth by improving soil structure and supplying carbon and nitrogen to soils (Gill et al. 2009). However, the adoption of this practice has remained low due to the high cost of application, variable responses to crop yields, and various logistical constraints such as limited or no availability of suitable organic matter sources and associated machinery to place organic matter at depth (Armstrong et al. 2017).

Nutrient management

Nutrient deficiency in plants is one of the major adverse impacts of waterlogging and leads to reduced photosynthesis, growth, and yield (Bange, Milroy, and Thongbai 2004). To overcome this, the application of certain essential nutrients (e.g., phosphorus, potassium, etc.) can help reduce the negative impacts of waterlogging and thereby, increase productivity (Noreen et al. 2018). For instance, nitrogen is lost via several pathways under waterlogged conditions, including denitrification, leaching, runoff, and erosion. Some studies provide evidence of the beneficial effects of nitrogen addition to waterlogged soils under various cropping systems (Kaur et al. 2017; Najeeb et al. 2015). Similarly, potassium fertilisers have also been reported to amend the detrimental impacts of waterlogging on crop yields (Ashraf et al. 2011; Cong et al. 2009). External fertiliser application also increases canopy duration and accelerates the production of photosynthetic assimilates, thereby, increasing the harvest index (Kisaakye

et al. 2015). Appropriate nutrient type, application method, and time and rate of application should be taken into consideration to prevent any negative effects of toxicities in plant tissues (Silva et al. 2017) and the environment. However, adoption of this practice is limited due to a lack of research and information on its potential use in improving crop growth and yield under waterlogged conditions.

Drainage systems

Drainage is one of the most important approaches to minimise the detrimental impacts of soil waterlogging and improve crop yields (Bos 1994). Draining excess water from soil improves root growth, crop emergence, and crop yields. The main aim of drainage is to reduce soil submergence and control soil salinity (Singh and Panda 2012). Surface drainage removes excess water from the land surface through constructed channels (Ritzema et al. 2008). There are various methods of surface drainage such as spoon drainage, W-drainage, reverse seepage, and these drainage systems have been reported to be cost-effective (Cox, McFarlane, and Skaggs 1994). However, the suitability of surface drainage is often limited by lateral movement of water or intrinsic soil properties (Saadat et al. 2018).

Poor subsurface water drainage occurs in waterlogged soils due to low or no movement of water. This could be due to soil compaction, heavy texture, or hardpans, or downward movement of water from slopes, which raises the groundwater table (Zhang et al. 2012). Subsurface drainage systems help in alleviating waterlogging problems and consists of pipe drains placed at various depths and spacing (Christen, Ayars, and Hornbuckle 2001). Tile drainage is also a subsurface drainage method used widely in agricultural soils (Williams, King, and Fausey 2015). The tile drainage system consists of small pipes made of either concrete or burnt clay that are installed at certain depths below the soil surface (King, Fausey, and Williams 2014). Along with waterlogging problems, subsurface drainage also controls soil salinity by leaching out concentrated salts (Christen and Skehan 2001).

There is another subsurface drainage practice referred to as mole drainage that is similar to tile drainage (Manik et al. 2019). This system relies on closely spaced channels and cracks in the subsoil to transfer excess water to the tile or pipe drainage systems (Hallard and Armstrong 1992). The installation of mole drainage is cost-effective, and these drainage systems can be used to efficiently balance water tables and manage drainage in floodplains (Tuohy et al. 2016).

Crop management

Flood tolerant crop varieties

One practice to reduce crop yield losses is the use of crop varieties that are tolerant to waterlogging stress. The mechanisms that plants use to counteract flood stress include the ability to transport oxygen to submerged roots, higher anaerobic respiration via higher glycolytic metabolism, and a longer period of the stomatal opening, which facilitates photosynthesis (Caudle and Maricle 2012). In addition, some plant species also adapt to flood stress by developing aerenchyma, adventitious roots, tolerance to secondary metabolites, maintenance of membrane potential, and stem hypertrophy (Bailey-Serres, Lee, and Brinton 2012). Additional plant traits for crop varieties that can be taken into consideration for developing flood-tolerant varieties include increased plant nitrogen uptake, retention of essential nutrients, lower susceptibility to waterborne pathogens, and improved crop quality characteristics (Kaur et al. 2020a). To summarise, the success of a breeding program is strongly dependent on the identification of genes and associated markers to various flood tolerance mechanisms (Manik et al. 2019).

Cover crops

Cover crops help improve soil health and can reduce waterlogging by improving soil structure and other soil hydro-physical characteristics (Blanco-Canqui et al. 2015). The root system of these cover crops can increase soil porosity and thus, ameliorate waterlogging. The higher water demand and transpiration rate of cover crops, especially during spring, can also potentially reduce soil water content in the waterlogged soils before the crop sowing. However, the impacts of cover crops on soil water depend on soil type, climate, the main crop, and cover crop species (Blanco-Canqui et al. 2015). Therefore, the utility of cover crops in waterlogged areas should be investigated (Kaur et al. 2020a) especially on a site-specific basis.

Changing planting dates

Planting dates of crops can be altered to increase crop water use and decrease the occurrence of waterlogging (Gardner, Nielsen, and Shock 1992). Planting dates should be adjusted so that extreme precipitation events and waterlogged conditions are prevented (Kucharik 2006). Other factors influencing the change in planting dates include the crops to be grown, timing, and duration of waterlogging.

Bio drainage

Bio drainage can be defined as soil water drainage using specific types of fast-growing trees with high evapotranspiration demand (Heuperman and Kapoor 2003). This also involves the incorporation of the herbaceous leguminous plants (e.g., clover) into cropping systems. These plants are adapted to waterlogging as they possess deep roots that can utilise the soil water to greater depths than most annual crops (Cocks 2001). The establishment of bio drainage plants has been demonstrated to lower the groundwater table adjacent to the root zone area (Sarkar et al. 2018). The appropriate choice of plant species with optimum population and suitable plant geometry will help in controlling the groundwater table (Singh and Lal 2018). Integration of bio drainage into agricultural systems could help improve land and water productivity. Also, the roots of bio drainage plants will help reduce soil compaction and enhance groundwater recharge capacity (Chowdhury et al. 2011). However, the adoption of this approach is limited due to a lack of proper knowledge, expertise, planting technique, and maintenance issues.

Addition of plant growth regulators

The adverse effects of waterlogging can be reduced by applying growth regulators (e.g., auxins and cytokinins) at appropriate growth stages (Ren et al. 2018). Growth regulators promote the stomatal conductance and photosynthetic capacity of flooded plants (Drew, Sisworo, and Saker 1979). For instance, pre-waterlogging foliar application of abscisic acid improved the tolerance to subsequent waterlogging induced injury in cotton plants by improving photosynthesis (Pandey et al. 2002). It is also reported that combined application of fertiliser and growth regulators can improve crop growth and yield (Li, Zhu, and Jiang 2013). In addition to plant growth regulators, treatment of seeds with hydrogen peroxide can also be an effective way to prevent flood stress (Jisha, Vijayakumari, and Puthur 2013). Pre-treatment of seedlings with hydrogen peroxide can protect crops from the oxidative damage caused by waterlogging due to the higher activity of antioxidant enzymes and reduced cell membrane damage. Treatment with hydrogen peroxide can also increase photosynthetic rate, pigments, root biomass, and stem diameter (Andrade et al. 2018). However, despite the benefits associated with hydrogen peroxide treatment, its adoption is still limited.

Areas requiring further research

There are significant knowledge gaps in our understanding of the pros and cons of the various waterlogging management practices under different climate conditions, soil types, landscape positions, and cropping systems. It is evident that while plants have various defence mechanisms to combat waterlogging stress (e.g., development of aerenchyma tissue, growth of adventitious roots), there is still a strong need for the development of resistant varieties that can withstand waterlogged soils. This should be the key strategy for overcoming flooding stress. In nature, genetic variants exist, but these may have variable tolerance to waterlogging stress and more efforts are required in breeding and screening crop cultivars. While several quantitative trait loci (QTL) pertinent to stress tolerance have been identified, most of these focus on oxygen availability and ignore other major waterlogging constraints. Therefore, further research is required to identify QTL pertinent to other stress constraints. Also, digging deeper into molecular mechanisms of flood tolerance in various crops by incorporating interdisciplinary approaches involving metagenomics, metaproteomics, and metabolomics should be considered. New flood-tolerant cultivars also need to be assessed for other agronomic, quality, and yield traits.

Further research on agronomic practices is also vital for flooding management. Conducting more advanced studies on soil science and hydrology will help improve our understanding of the various physical, chemical, hydrological, and biological soil properties and their interactions on different crops and their cultivars. Integrating the use of various digital data sources, crop modelling, geographic information systems, and remote sensing techniques for identifying areas in fields that are prone to waterlogging can help in the precision placement of crop or soil management practices to overcome waterlogging stress. Also, the adoption of various crop modelling techniques can be useful in simulating various management strategies and developing best management practices to evaluate the effectiveness of different management strategies. These models can be helpful for producers to make rational decisions on the adoption of various soil and crop management strategies in areas vulnerable to waterlogging problems.

For the wide adoption of these management strategies, it is necessary to evaluate the cost/benefit of these management practices. It is also imperative to understand the mechanisms of nutrient loss under waterlogged conditions under different cropping systems and quantify the benefits of the external application of nutrients such as nitrogen, potassium, phosphorus, and other macro-and micronutrients. There is still enormous scope for

improvement in managing waterlogged soil and minimizing its detrimental impacts on crop production.

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CHAPTER 5

EROSION

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Abstract

Soil erosion is the detachment of soil particles by an erosive agent (i.e., water, wind), and transportation of the sediment to another location. Sediment particles fall back to a state of rest on the soil surface (or other surface) in a sequential process called deposition or sedimentation. This chapter will discuss a) the types and causes of soil erosion and their impact on crop production worldwide; b) management options to reduce erosion rates and remediate cropland impacted by erosion; c) soil erosion assessment tools; and d) areas requiring future research.

Keywords: Wind, water, soil detachment, sediment transport, deposition.

Introduction

Water and wind erosion on the Earth's surface is the detachment of soil particles by a fluid (i.e., water or air) from an original soil body, and transport of the detached sediment away from its original location, where at some point, it is deposited. Each of these three processes (i.e., detachment, transport, and deposition) are part of the erosion process. Geologically, erosion is a natural process that can break particles from rocks and is critically important in the formation of soils and landscapes. However, in agricultural settings, loss of precious topsoil from cropland fields can be accelerated due to anthropogenic activities such as soil disturbance by tillage, reduced soil surface vegetative cover by harvest or tillage, overgrazing by animals, and burning of crop residues. Climate change that includes more variability in precipitation may increase drought and wet

periods, stressing plants and increasing the risk of crop failure and susceptibility to erosion. Also, more intense rainfall and wind speeds can increase water and wind soil erosion rates.

Erosion is a world-wide phenomenon that threatens not only a secure, sustainable food supply, but has environmental, human health, and economic consequences. Considering the constraints of soil erosion to agricultural crop management and production, we here discuss wind and water erosion processes, their extent around the world, detrimental effects, control and remediation practices, prediction tools, and future research needs. Understanding all these is essential for advancing our ability to lessen soil erosion constraints to crop production.

Erosion processes

Wind erosion

Soil erosion by wind occurs when the wind velocity at the soil surface exceeds a critical threshold value, and soil particles detach and move in response to wind shear forces (Tatarko et al. 2019). A soil surface that is rough or protected with non-erodible material, such as vegetation, requires a stronger wind to initiate particle movement than a bare, smooth surface. This means there is no single threshold velocity, but rather a range of velocities depending on the soil surface condition, including aggregation, roughness, crop status, and soil moisture.

Modes of sediment movement under wind erosion are surface creep, saltation, and suspension (Figure 5.1) and each has specific characteristics and effects (Hagen 2017). Surface creep involves soil particles approximately 0.5 to 1 mm in diameter that roll along the surface because they are too large to be lifted into the air. Saltation involves smaller particles of about 0.1 to 0.5 mm in diameter that are detached and lifted from the soil surface into the wind, subsequently dropping back down to the surface in a relatively short distance, typically less than 8 m. If these particles strike the soil surface with enough impact force, they can detach additional soil particles in a process known as abrasion. Suspension is where the smallest soil particles with diameters less than 0.1 mm are detached and lifted higher up into the air stream and can be suspended for long times and distances, even crossing oceans and continents.

Eventually the wind velocity decreases, and soil particles are deposited (Hagen et al. 2007). Within-field deposition typically occurs in furrows or vegetated areas. Deposition also occurs along the edge of fields, in ditches, fence rows, or barriers such as windbreaks. Suspension sized particles may

not be deposited until they have travelled hundreds or thousands of kilometres.

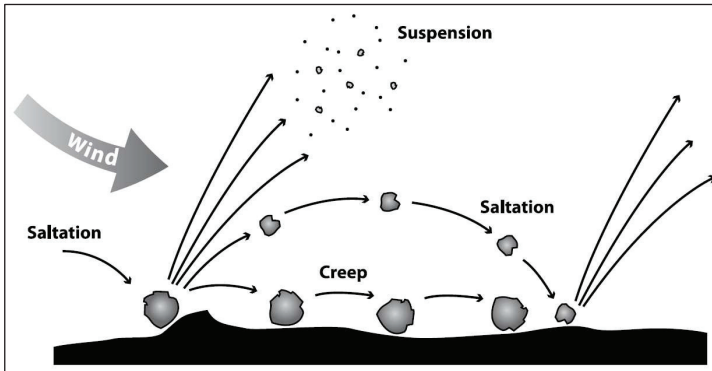


Figure 5.1 Soil particles can move during wind erosion by surface creep, saltation, and suspension. Particles in water erosion move similarly, though sediment movement near the soil surface is called bedload, and suspended particles are restrained to the height of the water surface (Figure credit: USDA-ARS).

On a wind eroding field, the amount of soil movement will tend to increase with distance downwind due to the impact of saltating grains breaking up clods and initiating movement in other particles (Hagen et al. 2010). If the field is long enough, the creep and saltation flow will reach a maximum that a wind of a given velocity can sustain, while the amount of suspension particles can keep increasing as they diffuse up into the atmosphere.

Water erosion

Soil erosion by water at the smallest upland scale occurs through two main processes: 1) soil detachment by raindrop impacts with shallow flow, rainfall-enhanced sediment transport (Kinnell 2005); and 2) soil detachment and sediment transport by concentrated flow in rill channels (Foster et al. 1985). Raindrops falling from heights of thousands of metres reach terminal velocity and strike the soil surface with extreme force that can detach and splash soil particles many centimetres into the air (Figure 5.2). Destruction of soil aggregates (Truman et al. 1990) releases fine particles (i.e., silt, clay) that can move into soil pores and clog them, decreasing infiltration rates and

increasing runoff potential (also known as surface sealing). Once runoff begins, detached soil particles move in the shallow overland flow towards downslope flow concentrations through rolling, saltation, and suspension. Continuous rainfall onto these shallow flows enhances the transport of all sizes of sediment particles (Toy et al. 2002; Gholizadeh et al. 2018).



Figure 5.2 Water and soil particles being splashed into the air after a raindrop has impacted a soil surface (Photo credit: USDA-NRCS).

The shallow water flow will eventually coalesce into small channels in local areas of lower elevation (e.g., in tillage rows). These small channels are called “rills” and may appear as somewhat regularly spaced on a field (Figure 5.3), though with sufficient length they will often coalesce into larger flow channels, such as gullies (Toy et al. 2002). Rills are small, typically only 4 to 20 cm in width and 2 to 10 cm in depth, and once obliterated by tillage will re-form in new, random locations (Nearing et al. 1997).

Gullies are larger channels that are much wider and deeper (Toy et al. 2002; Bennett and Wells 2019). In small agricultural fields, ephemeral gullies (Casali et al. 2000) form in regions of natural landscape flow concentrations (Figure 5.3). They may be obliterated from view by tillage but will reform in the same locations in subsequent storm runoff events. Ephemeral gullies often detach soil through downward cutting until a soil layer more resistant to detachment is reached (e.g., a plowpan), and then will widen. Moving up in scale, classical gullies (Figure 5.4) are larger channels that destroy entire areas in fields and are too large to cross with farm equipment and difficult to manage and remediate (Poesen 2011).



Figure 5.3 Ephemeral gully channels in a field in western Iowa, U.S. Extensive rill erosion is also visible (Photo credit: USDA-NRCS).



Figure 5.4 Classical gully in northeast China. The eroded channel has destroyed a road and is encroaching into adjacent fields (Photo credit: USDA-ARS).

As with wind erosion, sediment detached by water will move for some distance, until conditions change and decrease the ability of the water flow to carry the particles further (Foster et al. 1981; Toy et al. 2002). Decreases

in sediment transport capacity may be due to reductions in slope gradient in a landscape, or an increase in flow resistance due to a rougher vegetated surface (e.g., a grass buffer). When the water flow can no longer carry all the sediment, deposition occurs (also known as sedimentation). This dropping of sediment particles out of the flow onto the soil surface is usually selective, with the larger and denser particles (e.g., sands, large aggregates) depositing first, followed by smaller particles (silts, clays, small aggregates) (Davis et al. 1983).

Erosion on agricultural lands worldwide

Soil erosion is a global problem that threatens food security and the functioning of ecosystems. When considering erosion rates, they are usually in the context of the acceptable or tolerable rate of loss. Rates of tolerable soil loss are calculated using soil formation rates that range from 0.2 to 2.2 t ha⁻¹ yr⁻¹. Tolerable rates are considered those that maintain crop production and range from approximately 1 to 11 t ha⁻¹ yr⁻¹ (FAO 2019). Wind and water erosion rates from cultivated fields average 10–40 times greater than rates of soil formation and approximately 30% of the world's arable land has been lost to erosion in the last 40 years (Pimentel and Burgess 2013). Borrelli et al. (2017) found a 2.5% increase in global water erosion driven by land use change to cropland. The least developed economies experienced the greatest estimated soil loss and the greatest predicted erosion increases were in Sub-Saharan Africa, South America, and southeast Asia (Figure 5.5).

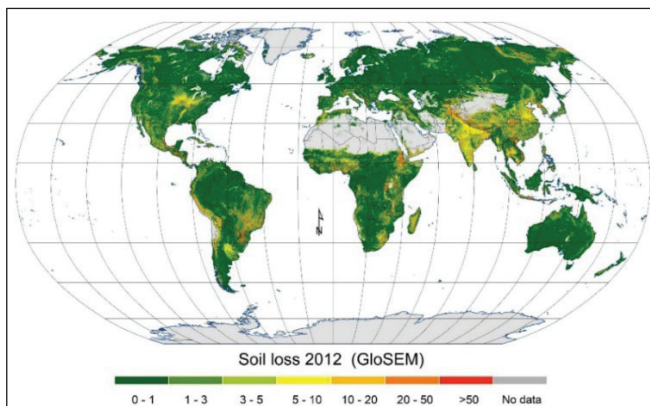


Figure 5.5 Estimated 2012 global (sheet and rill) soil erosion by water (Mg ha⁻¹ yr⁻¹) (Map courtesy of Pasquale Borrelli, 2/12/2021).

Soils, in general, are vulnerable to wind erosion if they a) are bare of vegetation, b) exist on large, flat areas, c) lack roughness or aggregation, and d) experience strong wind forces. Areas where one or more of these conditions exist, resulting in a high potential for wind erosion, are illustrated in Figure 5.6. For example, vegetation may be sparse in arid and semi-arid regions; broad flat areas exist in coastal plains; and poor aggregation may exist in sandy areas along coasts and fluvial deposition areas.

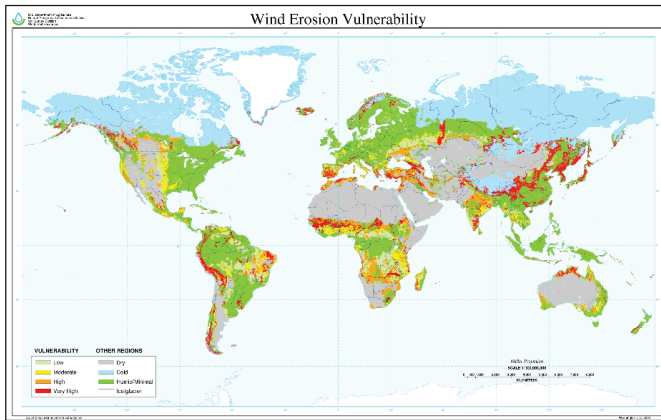


Figure 5.6 Map of areas worldwide vulnerable to wind erosion (USDA-NRCS 2002).

Estimates of global wind erosion have been made using both ground observations and modelling (FAO 2019). Regions with the highest wind erosion rates (>1.0 to $7.0 \text{ t ha}^{-1} \text{ yr}^{-1}$) are North Africa, the border between Iran and Afghanistan, and the Gobi Desert. Areas with $<1.0 \text{ t ha}^{-1} \text{ yr}^{-1}$ wind erosion include the mega-region of drylands through Iran and Afghanistan, the Arabian Peninsula, and across northern Africa, as well as dry areas of the western United States, Mexico, Australia, Argentina, and Chile (FAO 2019). Areas with the greatest loss of soil organic carbon (SOC) are those with intermediate wind erosion where the relative SOC contents are higher than drier areas such as deserts.

Detrimental effects of soil erosion

Loss of topsoil through wind or water erosion can have considerable detrimental effects. Wind erosion is known to affect agroecosystem services

including soil, air, and water quality (Tatarko et al. 2019). Wind erosion reduces soil productivity by physically removing the most fertile part of the soil (i.e., clays, organic matter). It lowers water-holding capacity, degrades soil structure, and increases soil variability across a field (Lyles 1977). Wind eroded sediments deposited in water systems such as road ditches, streams, and lakes can lower water quality and is one of the costliest off-site damages from wind erosion (Pimentel et al. 1995; Pimentel and Burgess 2013). Wind erosion damages plants by abrasion, blowouts, and deposition.

Blowing dust is a significant traffic safety concern in dry regions (Li et al. 2017). Soil blowing across roadways causes traffic accidents with numerous deaths and injuries each year (Van Pelt et al. 2020). For example, the U.S. National Weather Service reported that dust caused highway accidents in Arizona resulted in at least 157 fatalities and 1324 injuries statewide in 50 years (Lader et al. 2016). Fine dust from wind erosion also contributes to global climate change. Dust fall on mountain snowpacks increases the adsorption of solar energy causing earlier snow melting and altering regional and continental hydrology (Painter et al. 2017). In the U.S. Great Plains, windblown dust levels have risen at rates of up to 5% per year over the last 20 years (Lambert et al. 2020), similar to trends in the expansion of cropland. With increasing drought and land use change in the Great Plains, as possible under climate change, an increased trend in dust from wind erosion can be expected.

Fine particulates from wind erosion are also known to be a human respiratory hazard (Goudie 2014; Kjølgaard et al. 2004). Very small dust particles of less than 2.5 and 10 microns in aerodynamic diameter, known as PM-2.5 and PM-10 respectively, can penetrate deep into the lungs and affect pulmonary systems and human health. Both PM-2.5 and PM-10 are regulated as health hazards by many countries (e.g., US-EPA 1999; NOM 2014).

Water erosion also removes productive topsoil, reducing soil productivity and increasing the need for additional fertiliser. Losses of organic matter with the topsoil decrease soil aggregation, water holding capacity, and general soil tilth, increasing the need for supplemental irrigation, aeration, and subsurface tile drainage. Many of the most notable impacts of soil erosion by water are the sediment and pollutants that affect off-site streams, rivers, and oceans (US-EPA 2021). Sediment is the major pollutant of water and reduces the capacity of water supply and flood control reservoirs, clogs navigation channels and harbours, and transports nitrogen (N) and phosphorus (P) that are critical nutrients affecting eutrophication, hypoxia (Rabalais and Turner 2019), and formation of harmful algal blooms (HAB; Watson et al. 2015). For example, Lake Erie in the northern U.S. annually

experiences algal blooms due to the P loading in runoff from the surrounding agricultural areas. In August 2014, an HAB episode near the Lake Erie water intake for the city of Toledo resulted in a cyanobacteria algal bloom. Its microcystin toxin contaminated the city's drinking water supply causing a 3-day ban on water use for nearly 500,000 people (Fitzsimmons 2014).

Worldwide, research into the economic costs of erosion has been limited. One study estimated annual loss to global GDP at ~\$8 billion (2021 U.S. dollars), reducing crop yields by 33.7 million tons, and increasing water withdrawal by 48 billion m³ (Sartori et al. 2019). In the U.S., an estimated 4×10^9 tons of soil are lost from cropland each year, translating to total on-site economic losses of \$50 billion, of which 74% is lost nutrient replacement (Pimentel et al. 1995). Additional annual off-site costs exceeded \$932 million for dredging of waterways, increased water treatment facility operating costs, and loss of ecosystem services. On-site and off-site costs associated with wind erosion exceed \$15 billion (Huszar and Piper 1986). In 1992, the combined off-site and on-site costs of erosion in the U.S. were estimated to be \$82 billion (2021 dollars), or about \$182 ha⁻¹ for cropland and pasture (Pimentel et al. 1995). This erosion increases production costs by about 25% each year. The study estimated that it would take an investment of \$11.9 billion yr⁻¹ for conservation practices to reduce U.S. erosion rates from about 17 t ha⁻¹ y⁻¹ to a sustainable rate of about 1 t ha⁻¹ y⁻¹ on most cropland.

Practices used to control and minimise erosion

While soil erosion is a critical problem, there are numerous practices available that can control or minimise its impacts. Conservation practices are designed to either reduce the erosive force of the fluid at the soil surface or create a soil surface more resistant to these forces. Some practices also trap particles to encourage deposition.

For wind erosion, most practical conservation practices preserve crop residues or keep growing vegetation on the field (Siddoway et al. 1965; Woodruff et al. 1977). This results in many beneficial effects, such as reducing the wind force on the soil, creating a wind-resistant surface, and trapping saltating particles. Crop rows perpendicular to the prevailing winds control wind erosion more effectively than rows parallel with the wind. Standing residues are more than twice as effective as flattened residues. Windbreaks, grass barriers, stripcropping, and/or clod-producing tillage can supplement or be used in lieu of vegetative cover.

Cropping systems that preserve surface residues year-round are a practical approach to reduce soil erosion potential by wind as well as water.

Maintaining soil surface area cover of 30% reduces soil erosion to tolerable levels, with only marginal improvements with higher cover percentages (Fryrear 1985). Excessive tillage that buries crop residues is a major cause of inadequate vegetative cover on cropland. Practices such as mulch tillage, no-till, ridge-till or strip-till involve managing the amount and distribution of plant residues on the soil surface. For the most erodible surfaces, maintenance of permanent vegetative cover may be needed to keep soil loss under control.

When vegetation is sparse because of drought or cropping practices/types, then ridges and large soil clods (or aggregates) are frequently the only means of controlling wind erosion on large areas. While cloddy soil surfaces absorb more wind energy than flat, smooth surfaces, both ridged and cloddy surfaces absorb even more. Because of the directional nature of winds, ridges formed by tilling and planting across the prevailing wind erosion direction are more effective than parallel to the winds. However, roughness degrades over time with the weathering effects of rainfall, freezing and thawing, and the wind itself (Skidmore et al. 1986). Roughening the land surface with ridges and clods reduces the wind velocity and traps drifting soils.

Cross wind strip-cropping grows crops in strips perpendicular to the prevailing wind direction. Strips susceptible to wind erosion should alternate with strips having a wind-resistant cover. This practice reduces the downwind effect by limiting the distance particles can travel before being trapped.

In contrast to these previous methods, barriers alter the wind force on the soil surface (Hagen et al. 1981). Wind barriers, either vegetative (e.g., windbreaks) or non-living (e.g., snow fencing), help reduce wind speed on the downwind side of the barrier and trap moving soil (Figure 5.7). Barriers significantly reduce wind speed for a distance about 10 times the height of the barrier, in effect, reducing the field length along the erosive wind direction. However, the fully protected zone varies with barrier porosity, wind velocity, and wind direction deviation from perpendicular to the barrier. Barriers also protect crops from wind and abrasive erosion, shelter livestock, promote snow capture, and provide wildlife habitat. One advantage of barriers over most other types of wind erosion control is they are relatively permanent, and during drought years, may be the only effective and persistent control measure on cropland.

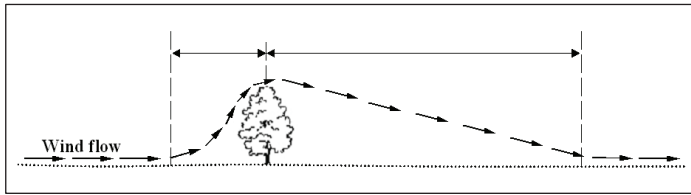


Figure 5.7 Wind barriers, such as a row of trees, reduces exposure to wind and provides a protected zone (Figure credit: USDA-ARS).

Cross wind trap strips consist of herbaceous vegetation resistant to wind erosion, established in one or more row strips perpendicular to the prevailing wind direction. Since saltating particles can travel up to 4.5 m, cross wind trap strips should be at least that wide and up to 8 m for shorter vegetation. The purpose is to trap saltating particles and provide protection from wind erosion effects.

Control practices for water erosion fall into two main categories: 1) land management, and 2) engineering solutions. Land management includes the use of alternative crops, crop rotations, contour farming, contour strip cropping, conservation tillage, cover crops, and buffer strips. Engineering solutions can include terracing, grassed waterways, impoundments, and grade-control structures. Often, combinations of land management and engineering solutions can provide a comprehensive systems approach to minimising or eliminating detrimental soil erosion by water.

Land management practices are the easiest and least costly to implement. Some multi-year perennial crops, such as deep-rooted legumes (e.g., alfalfa), can provide substantial surface and canopy cover, with extensive root systems that hold the soil in place and can reduce soil loss to very low levels (Toy et al. 2002). Growing these types of crops, even in a rotation that includes row crops (e.g., corn), can reduce overall long-term soil loss rates.

Farming along the contours (perpendicular to the downslope direction) can also reduce soil erosion by water considerably. If the contour ridges are high enough so that any runoff water is held between them or slowly moves down these low slope gradient rows, then soil detachment by the flowing water will be greatly reduced. Additionally, an even more effective practice is to use alternating strips of different crops on a contoured hillslope, so that a region of row crops will be next to region of perennial or small grain crops that catch any sediment that might leave the more intensively tilled areas (Figure 5.8). A specialised type of contour strip is a grass or riparian buffer strip (Toy et al. 2002). These are regions of permanent vegetation (e.g.,

grass) at the bottom edges of fields and/or along adjacent stream channels, ditches, or ponds that filter out sediment or pollutants and enhance infiltration. Buffer strips are commonly ≥ 10 m wide.



Figure 5.8 Contour strip cropping being used as a soil conservation practice in central Wisconsin, US (Photo credit: USDA-NRCS).

Conservation tillage includes no-till and reduced-till systems that leave substantial ($>30\%$) crop residue surface cover after planting (Toy et al. 2002). No-till systems minimise disturbance of the soil and surface residues and usually only consist of a single planting operation with a no-till planter. By maintaining almost all the previous crop's residue, the soil is protected from raindrop detachment and from surface sealing, so greater infiltration rates can be sustained, reducing potential runoff and flow detachment. Minimally disturbed soil also remains consolidated and has strong cohesive forces that resist erosion by water. Reduced-till systems incorporate some tillage, which might include a straight shank chisel-plough, but minimise secondary tillage passes and maintain more protective residue cover.

Cover crops are increasingly being used to provide a living plant canopy cover to protect the soil from erosion while also scavenging excess nutrients (N and P) to minimise their off-site losses (SARE 2007). The four types of cover crops are grasses (e.g., ryegrass), legumes (e.g., clover), brassicas (e.g., radishes), and non-legume broadleaves (e.g., flax). They are either interseeded into a growing cash crop (e.g., ryegrass broadcast by airplane

into a soybean field) or planted immediately after cash crop harvest. The goal is to get the cover crop established and providing protective canopy cover over the soil surface during the period prior to planting the subsequent cash crop. The cover crop is then killed by herbicides or tillage, and the next cash crop planted. Most recently, mixes of multiple classes of cover crops are being planted together to provide the range of benefits from each type of plant.

The other main class of water erosion control is the engineering solution. Engineering solutions are needed when factors such as topography, climate and soils combine in ways that prevent effective control through land management approaches alone. Terracing involves reducing slope lengths and slope gradients with berms or walls and has been used for thousands of years by various civilisations, particularly those inhabiting steep landscapes (Figure 5.9).



Figure 5.9 Terraces in Nepal, constructed and farmed largely with human and animal labour (Photo credit: USDA-NRCS).

Grassed waterways are channels designed to safely convey runoff water downslope from agricultural fields and then into a main channel, ditch, or subsurface drainage inlet (Fiener and Auerwald 2003). The shape, width, and slope gradient of these waterways, as well as the grass characteristics, are accounted for in the design to prevent flow velocities, flow depths, and flow shear forces that would erode the channel soil. Additionally, subsurface tile drainage is commonly installed beneath the soil surface in these waterways to enhance infiltration and minimise soil seepage potential.

Often grassed waterways are part of a comprehensive erosion control plan for a farm or small watershed. For example, conservation tillage on the adjacent cropped fields is also often needed to minimise excessive sediment

entering the channel and depositing in the grass, reducing its capacity and effectiveness. Additionally, grade-control structures are often included to safely convey flow to lower elevations and prevent gully formation. Figure 5.10 shows a field conservation system with a long, grassed waterway above a drop structure.



Figure 5.10 A drop structure made of timbers and concrete blocks stabilises a grassed waterway channel in Iowa, U.S. The waterway extends hundreds of metres up through the surrounding crop fields, safely conveying runoff water away, while the structure prevents a gully from forming in the channel (Photo credit: USDA-NRCS).

Grade-control structures are used to safely move erosive water flows from higher to lower elevations (or grades), typically in an agricultural field channel, grassed waterway, or another stream channel. Common features of all grade-control structures are a control section that accomplishes the change in elevation, an energy dissipation section, an approach section, and a downstream exit section. A variety of construction materials can be used, including concrete, stone riprap, treated lumber (Figure 5.10), etc. They are built to accommodate some design storm criteria (e.g., flows generated by a 10-year return-period, 24 h rainfall event). Two requirements for grade-control structures are that they must be large enough to handle the design flow rate, and the flow energy must be dissipated so that the structure and the downstream channel are protected from scouring and erosion. An excellent resource detailing numerous sediment control practices and

designs is available from Clean Water Services (2020).

Stabilisation and remediation of eroded lands

Many areas in the world have soils that have undergone erosion to some extent. Soils only minimally affected by loss of their surface topsoil may not require any stabilisation or remediation. If subsequent agricultural cropping practices minimise the risk of future erosion and include practices to increase organic matter, crop productivity can be maintained or improved. Other soils that have been extremely damaged by erosion and that have lost much of their surface topsoil, can be more difficult to manage and restore. On soils that can still support crop production, the use of no-till practices plus cover crops, long rotations with deep-rooted plants, and other systems that increase soil organic matter and structure can be implemented. In other situations where soil quality is so poor that crops cannot be successfully grown, then more intensive practices such as large-scale additions of mulches, manure, or other organic residues may be necessary. At some locations, land may simply be abandoned and allowed to return to native vegetation for long periods of time. With sufficient time, the soil may stabilise. However, it may never return to its former level of agricultural productivity, or at least not for many centuries.

Often, wind erosion starts in a small field area where surface conditions are vulnerable to erosive wind forces. Highly erodible areas include sandy regions and elevated locations. If these are left to erode, saltating material can cause other field areas to detach until, eventually, the entire surface is blowing. Thus, it is better to stabilise these areas before erosion begins, rather than after. Planting permanent vegetation can stabilise areas that cannot be stabilised by typical management practices. Adding surface cover is cost effective in stabilising continually eroding soils (Armbrust 1977). Spreading crop residues or other materials reduces wind velocity and traps moving soil particles. Residues for surface stabilisation must be anchored into the soil and are more effective if vertical as opposed to lying flat. Livestock manure, creating large clods, or spraying chemical suppressants can also stabilise actively eroding surfaces (Armbrust 1999).

Emergency tillage is a practice designed to stabilise actively blowing fields by tilling strips across the eroding area perpendicular to the expected wind direction to provide a rough, ridged, cloddy surface (Lyles and Tatarko 1982). It should only be used as a last resort when other controls have failed. Emergency tillage is only a temporary measure because clods can disintegrate rapidly under saltating conditions and wind direction can change.

During periods of extended drought, wind erosion may damage the soil to such an extent that the land is permanently impaired. This occurred after prolonged drought during the U.S. Dust Bowl period of the 1930s, which resulted in crop failures, overgrazing, and loss of soil structure (Hurt 1985). A comparable period occurred in Australia during 1895-1945. Such devastation led to the establishment of conservation programs in the U.S., Canada, and Australia. In the U.S., 235,000 ha of severely eroded lands in the Dust Bowl region were purchased and vegetation established as National Grasslands (Lee and Gill 2015). In addition, the U.S. attempted to reduce wind speed at ground level by planting large areas with rows of trees. However, these plantings were not successful in drier areas where they were most needed, and landowners removed many, starting in the 1950s. More recently, large-scale tree planting began in China with the Three-North Shelterbelt Programme, a national effort to eventually plant 4.06 million km² of trees along the border of northern China's encroaching desert by 2050. Similar projects have been attempted across the Sahara Desert in northern Africa in Burkina Faso, Nigeria, and Ethiopia with only limited success (Goffner et al. 2019).

Many of the stabilisation/remediation methods for lands impacted by wind erosion are also applicable to soil degraded by water erosion. Establishment of permanent vegetation cover with perennial grasses and/or legumes can stabilise soils affected by water erosion, and with enough time, add organic matter and more structure to these lands. However, as mentioned previously, it is unlikely that degraded soils can be fully remediated to their former (un-eroded) condition and levels of agricultural productivity.

Other practices may include the use of soil amendments such as gypsum, anionic polyacrylamide (PAM) and/or biochar to stabilise soils, enhance infiltration, and increase resistance to detachment by water flows (Peterson et al. 2002; Flanagan et al. 2003; Rodriguez-Franco 2021). Gypsum can have substantial beneficial effects for soils, including reducing particle dispersion, enhancing infiltration, and reducing runoff and erosion (Peterson et al. 2002). PAM applied to soils alone, or in combination with gypsum, can greatly reduce runoff and erosion, and can temporarily stabilise an eroding soil surface, giving time for permanent vegetation establishment and erosion control (Flanagan et al. 2003). Biochar is a form of charcoal produced by exposing organic waste matter (e.g., wood chips) to heat in a low-oxygen environment and can be used as a high carbon soil amendment to restore highly disturbed and eroded lands (Rodriguez-Franco 2021).

Erosion assessment tools

Erosion prediction tools have been developed for widespread applications by conservation agencies and land managers to identify and understand potential wind and water erosion problems. Erosion simulation models can be useful not only for assessing soil losses, but also the design of erosion control systems. By observing how soil loss is affected by changes in simulated field and crop management practices, alternatives can be developed to control soil loss (e.g., Tatarko et al. 2019). A recent article by Borelli et al. (2021) provides a comprehensive listing and database of wind and water erosion modelling studies from 1994 to 2017, reviewing over 8400 papers.

Different soil erosion models have been developed over the past century with varying degrees of complexity, capability, and ranges of spatial and temporal scales of application. Each is unique in its structure, processes, limitations, applications, and the databases required, as well as inputs and outputs. Models range from empirical to physical process-based, or some combination of both. Empirical models attempt to describe the statistical relationships between soil loss and the soil surface under the given fluid force, without fully describing system behaviour and interactions. Process-based approaches simulate the detailed physical and/or biological processes that explicitly describe the natural system and its fluid force responses.

Jarrah et al. (2020) provides a comprehensive review of the most established wind erosion models available. Among them, the Wind Erosion Prediction System (WEPS) model is the most widely used in the U.S. and has been applied throughout the world (Tatarko et al. 2019). WEPS was developed by the United States Department of Agriculture (USDA) - Agricultural Research Service (ARS) as a replacement for the empirical Wind Erosion Equation (Woodruff and Siddoway 1965) for improved accuracy and applicability of wind erosion predictions. WEPS is a physical process-based model that simulates the effects of weather and management on field surface conditions to calculate wind erosion potential, daily (USDA-ARS 2020). Figure 5.11 shows that the WEPS structure consists of a user interface (programmed in Java), a science model (programmed in Fortran) with a main controlling routine and six science sub-models (hydrology, management, soil, plant growth, residue decomposition, and erosion), and five databases (soil, plant growth and residue decomposition, operations, wind barriers, and climate). The WEPS Technical Documentation provides detailed descriptions of science in each of the sub-models (USDA-ARS 2020).

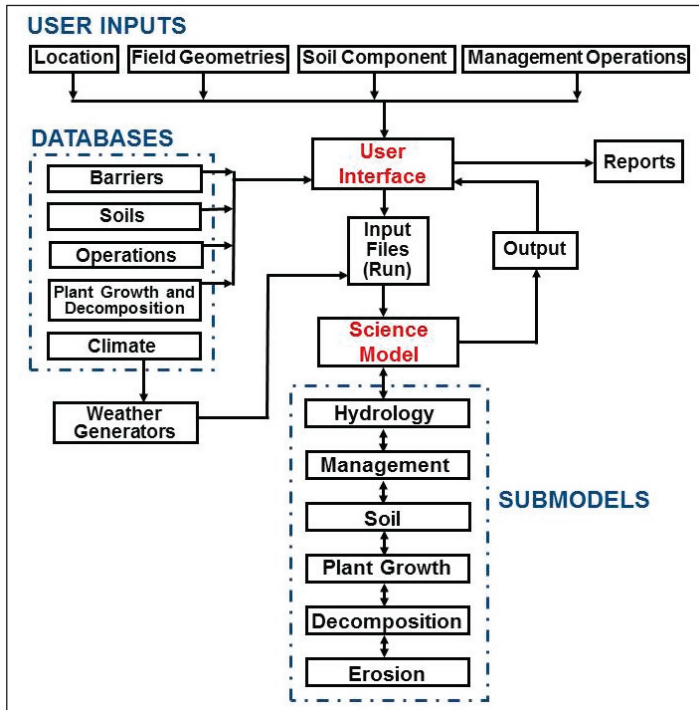


Figure 5.11 Structure of the WEPS model (Figure credit: USDA-ARS).

Water erosion prediction equations began to be developed in the early 20th century. The empirical Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978), first available in the early 1960s, was the first comprehensive tool for estimating sheet and rill erosion from cropland in the U.S. and was subsequently applied in over 100 countries worldwide. USLE was developed from over 10,000 plot years of natural rainfall/runoff erosion plot data from 35 Soil Conservation Experiment Stations in the U.S. (Gilley and Flanagan 2007). Revisions include RUSLE (Renard et al. 1997) and RUSLE2 (USDA-ARS 2013), provided as computer simulation programs with user-friendly interfaces and databases for widespread application. The USDA Natural Resources Conservation Service (NRCS) uses RUSLE2 for agricultural field conservation planning.

However, as described earlier in this chapter, sheet and rill erosion are only part of the processes contributing to total soil loss, and substantial erosion can occur in ephemeral gullies and other channels. Water erosion

control practices can include grassed waterways, grade control structures, and impoundments that cannot be simulated by USLE-type technologies. Most sediment leaving fields and farms is transported in larger concentrated flows. Thus, to comprehensively evaluate field-scale water and sediment losses, other types of water erosion assessment tools are needed.

In the U.S., USDA scientists developed the process-based Water Erosion Prediction Project (WEPP) model for application to hillslope profiles and small agricultural watersheds composed of multiple hillslopes, channels, and impoundments (Flanagan and Nearing 1995; Flanagan et al. 2007). WEPP is a physical process-based, distributed parameter, continuous simulation erosion model, implemented in multiple interfaces (Windows standalone, GeoWEPP ArcGIS, web-based) with large nationwide databases for climate, soils, land management, and cropping (Flanagan et al. 2007). Most recent WEPP development activity is concentrated on the creation of web-based interfaces (Figure 5.12) that allow for user selection of specific crop fields, automatic climate and soils identification by geospatial location, access to large NRCS crop and land management operations databases, and extensive tabular and graphical outputs.

Globally there are numerous other process-based water erosion prediction models, including EUROSEM (European Soil Erosion Model) (Morgan et al. 1998), LISEM (Limburg Soil Erosion Model) (De Roo et al. 1996), KINEROS2 (Kinematic Wave Overland Flow, Channel, Routing and Erosion Model) (Smith et al. 1995), and GUEST (Griffith University Erosion System Template) (Misra and Rose 1996). These all include the simulation of detachment, transport, and deposition erosion processes.



Figure 5.12 Recent WEPP model development includes web-based interfaces utilising geospatial information and linkages to extensive USDA climate, soils, land management, and crop databases (Figure credit: USDA-ARS).

Erosion research needs

While many of the water and wind erosion processes have become better understood during the past century, there is still much to be learned. The early water erosion plot research measured total gross runoff and sediment loss from a storm event or events, resulting in a large database of over 10,000 plot-years of data. However, while adequate for statistical analyses and development of empirical tools like the USLE, these data are largely missing runoff and sediment discharge rates within events. Without these, they have limited value in understanding hydrologic and erosion processes.

Current process-based water erosion models determine soil detachment spatially through solution of differential equations down a slope. However, there are extremely few data where detachment rates have been measured spatially down an erosion plot or rill, making model validation difficult or impossible. Almost all plot and small watershed research up to this point has measured flow and sediment losses that leave the end of a plot or exit a watershed. New experiments utilising new technologies such as drones or photogrammetry to provide detailed surface elevation changes between erosion events could help to measure temporal and spatial soil loss rates for interrill areas, rills, and gullies. Other new technologies including sediment tracing/fingerprinting using rare earth elements or isotopes may help with identifying points on landscapes where sediment reaching a watershed outlet originated.

Erosion is a dynamic process in which the surface and soil loss change spatially and temporally, not only from day to day and field to field, but also during an erosion event. Research is needed to support wind and water erosion models that better simulate both spatial and temporal variability in detachment and deposition as well as controlling processes for surface erodibility in terms of soil aggregation and crusts, water balance, plant growth, and residue decomposition. Many supporting modules for erosion calculations were simplified because of computational constraints at the time of their development, yet they need to be accounted for in new models. For example, wind and water erosion selectively transport the finer, lighter particles (i.e., clays, organic matter), which are the most fertile part of the soil. However, little is known about the interactions of wind and water erosion with nutrient cycling. Better understanding of these selective processes is needed to better predict erosion effects on crop productivity.

Researchers have also found that wind erosion models underestimate the occurrence of small storms, which could lead to subsequent larger storms (Hagen 2004). This was attributed to the spatial variability of the test sites and these having small inclusions of surfaces that were more erodible than

the average surface. While some process-based models attempt to represent dynamic changes of soil surfaces, such processes are still poorly understood. Like processes at finer spatial scales, research into large spatial scales of dust transport is necessary to understand continental and global dust budgets and their effects on climate change. Erosion and deposition occur across field boundaries with differing land use or cover. Thus, estimations of sediment flux changes across changing surfaces, as well as regional dust transport models, are also needed. In addition, measurements of threshold velocities are needed through integrated field experiments that, in part, use remote sensing to collect and deliver data over a range of temporal and spatial scales. The U.S. has established a National Wind Erosion Research Network (Webb et al. 2016) that will provide valuable data for spatially distributed models.

The climate data and drivers used in wind and water erosion prediction technologies are not sufficient to represent climate parameters for future models (Sharratt et al. 2015). An improved climate engine would rapidly incorporate traditional instrumented climate networks and various gridded products to produce consistent climate forcing at the required spatial and temporal scales. This engine could be used to forecast and assess past, present, and future climate scenarios and corresponding uncertainties for both climate variability and climate change. For example, developing models that predict the likelihood of events will allow land managers and government agencies to apply pre-emptive controls for such events.

Natural and anthropogenic factors also affect erosion at a wide range of spatial and temporal scales. For example, tillage-induced microtopography or surface roughness at the clod scale can change within a single rain- or windstorm, but the oriented roughness from tillage implements (e.g., wheel tracks) may last throughout the entire growing season or even decades. These random and oriented roughness parameters affect the erosive power of the wind and rain on soil. Erosion from cold-season hydrologic processes on croplands remains understudied. For snow-dominated areas, the greatest erosion rates commonly result from rain-on-snow, rain on-frozen soil, and snowmelt runoff events. Repeated freezing and thawing break down aggregates, resulting in friable surfaces prone to wind and water erosion.

Both wind and water erosion transport and deposit sediment on the landscape, resulting in source material for both types of soil loss. For example, wind erosion may deposit loose sediments in a field or stream that subsequently are eroded and transported by rainfall/runoff water. Similarly, unvegetated alluvium acts as a source area for aeolian entrainment. The interactions between these processes, and how to account for changes in surface soil composition in deposition areas where sediment may become a

source, have not been quantified. Combining wind and water erosion models could accomplish this. The new modelling system would have common databases with identical support modules for climate, hydrology, water balance, plant growth, residue decomposition, soil, and management effects, and account for interactions between wind and water erosion.

Scientists from USDA-ARS, USDA-Forest Service, USDA-NRCS, and other agencies and universities recently met to discuss future needs in erosion research and modelling. Many topics covered in this section are discussed in more detail in a recently published strategic plan (Weltz et al. 2020).

Summary

Soil erosion on agricultural lands can have serious on-site and off-site impacts, as it can threaten crop productivity as well as air and water quality, human health and safety, flood control, and navigability. When left unchecked, soil loss due to wind and/or water erosion can substantially damage soil resources and the land's ability to support sustainable food production. Fortunately, there are numerous soil conservation practices that can be applied to minimise erosion, as well as sophisticated computer simulation models that allow for rapid and accurate estimation of soil losses and the effectiveness of alternative land management practices. Future erosion research and modelling efforts should further improve our knowledge and ability to effectively mitigate the risk of soil losses.

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PART C:

CHEMICAL CONSTRAINTS

CHAPTER 6

SOIL ALKALINITY/SODICITY: DEGRADATION PROCESSES, CONSTRAINTS FOR CROP PRODUCTION, AND THEIR MANAGEMENT

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Abstract

The presence of undesirable amounts of sodium (Na), toxic concentrations of bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}), and alkaline hydrolysis of Na in the soil solution make sodic soils unproductive and economically unviable. Sodic soils are characterised by poor aggregation, low soil permeability, high exchangeable Na per cent and sodium adsorption ratio in the soil water solution, mobility of organic matter, losses of available mineral nutrients, and impaired soil productivity. Reclamation and management of sodic soils require an appreciable reduction of soluble and exchangeable Na and its replacement by the more favourable calcium ions and neutralisation of HCO_3^- and CO_3^{2-} in the root zone. The reclamation of these soils depends on the relative effectiveness of amendments, the target crop to be cultivated, the availability of quality irrigation water, and the relative costs involved.

Keywords: Sodic soils, soil taxonomy, secondary sodification, reclamation, amendments, management.

Introduction

Soluble salts are a natural feature of the landscape, but their excessive accumulation can be problematic. The process of soil salt accumulation is called salinisation. This is an *in-situ* form of soil degradation that arises due to the build-up of soluble salts to deleterious levels at or near the soil surface (Schofield, Thomas, and Kirkby 2001), and it inflicts a negative impact on plant growth. Salt-affected soils have been found to occur in more than 120 countries, under different environmental conditions, and they exhibit diverse morphological, physical, chemical, and biological properties (Szabolcs 1989).

Geology, climate, topography, and human activities are some of the important factors responsible for the accumulation of salt in soils. The transport of salt from salt rich geological sediments by surface or ground water increase salinisation in closed basins or lowland areas due to poor drainage. In dryland areas, low rainfall and high rates of evapotranspiration reduce profile leaching and lead to the accumulation of salt added to the soil in rainfall over time. Over-irrigation with water of high salt concentration and human-induced factors, such as inappropriate land use, are responsible for aggravating soil salinity. The accumulated ions causing salinity or alkalinity include sodium (Na), potassium (K), magnesium (Mg), calcium (Ca), chlorides, carbonates, and bicarbonates. In many classification systems the term “salt-affected soils” is limited to those saline and alkali soils where neutral or alkaline hydrolyzing sodium salts dominate. In the present chapter the discussion is restricted to the soil alkalinity/sodicity inducing degradation processes, their classification, distribution, characteristics, and management options.

Classification, taxonomy, and extent of salt-affected soils

Salt-affected soils affect over 955 Mha, accounting about 10% of the total surface of dry land (Szabolcs 1989), and occur in 120 countries across all the continents. However, as per the FAO/ UNESCO Soil Map of the World (FAO 2008), out of the total 835 Mha of salt-affected area, sodic and saline soils extend over 438 and 397 Mha, respectively.

One of the defining features of many salt-affected soils is the high proportion of monovalent versus divalent ions, particularly Na, on the soil exchange. Soil aggregate stability is determined by the attractive and repulsive forces at the surface of soil particles, which are governed by a range of factors, but particularly the nature of the cations on the soil exchange and soil solution electrolyte concentration (Rengasamy and

Olsson 1991). When a high proportion of monovalent Na ions exist, this can reduce aggregate stability and cause aggregates to slake, swell, and disperse when wet, leading to poor soil physical structure, problems of chemical infertility and toxicity, and low soil biological activity. Due to the tendency of high soil solution salt concentrations to promote clay flocculation, dispersion is particularly prevalent in soils with lower soil solution electrolyte concentrations (Rengasamy and Olsson 1991).

In the US, salt-affected soils have been grouped into 3 categories i.e., saline, saline-alkali, and alkali soils. This classification is based on a soil's total soluble salts (electrical conductivity of the soil-saturation paste extract (EC_e)), exchangeable Na percentage (ESP) or Na adsorption ratio (SAR), and pH (saturation paste (pH_s)) (SSSA 1987). Saline soils, owing to the presence of excess neutral soluble salts ($EC_e > 4 \text{ dS m}^{-1}$) and an absence of significant amounts of exchangeable Na (ESP < 15), are generally flocculated. Saline-alkali soils ($EC_e > 4 \text{ dS m}^{-1}$; ESP > 15) behave similarly to saline soils as long as sufficient salts are present to promote clay flocculation; whereas upon leaching, these soils disperse and may become alkaline ($pH_s > 8.2$), and their permeability reduces to levels that affect crop growth. The term "alkali" was later replaced with "sodic". Sodic soils contain sufficient exchangeable Na (ESP > 15), and low enough salt content ($< 4 \text{ dS m}^{-1}$) to affect the physical behaviour of soils and interfere with growth of most crops.

Various modifications to the US classification system have been made worldwide to better represent local soils. While an ESP of 15 is commonly used to identify the presence of sodicity in these classifications, in some instances, a different threshold based on the point at which dispersion more commonly occurs in local soils is used. For example, a critical ESP of 6 is used in Australia (Rengasamy and Olsson 1991) while a value of 8 is considered more appropriate for Indian Vertisols (Minhas and Sharma 2003). This variation in threshold ESP occurs due to the variety of factors other than just Na that influence soil aggregate stability and dispersion (e.g., organic matter content, soil mineralogy, EC_e) (Rengasamy and Olsson 1991).

Characteristics of sodic/alkali soils

Sodic soils can have a neutral, acidic, or alkaline pH, however, worldwide they are predominantly alkaline and alkaline sodic soils, or "alkali soils", will form the focus of discussion in the current chapter. Alkaline sodic soils have an ESP ≥ 15 , $pH_s \geq 8.2$ and variable EC_e . These soils contain disproportionately high levels of Na relative to Ca and Mg on

the soil exchange, and in solution phases, are dominated by carbonates and bicarbonates of Na. For example, in the sodic soils of Uttar Pradesh, soil pH can be very high (10.8), ESP as high as 90 and ECe in upper soil layer (up to 70 cm) ranges from 4 to 24 dS m⁻¹ (Tiwari, Sharma, and Tripathi 1989). The surface horizon usually has a platy structure with very low water transmission properties and is highly dispersed. Usually, there is a zone of clay accumulation with high bulk density in the subsurface horizons, along with the presence of argillans. A natric horizon is also usually present and a zone of CaCO₃ accumulation typically lies within the zone of a fluctuating water table. This water table may appear just below the surface during high rainfall periods (e.g., monsoon) but recede to about 5 m depth in subsequent periods. The groundwater in these areas usually has low salt content, comprising mainly of Na⁺ and HCO₃⁻ ions.

There are considerable differences in the properties of sodic soils with and without structural B horizons. When the concentration of Na carbonate remains high in the upper horizon of the soil profile, soils without structural B horizons develop. These alkali soils resemble saline soils, having a saline (salic) horizon found within 125 cm of the surface, or ECe >4 dS m⁻¹ at the surface, but pH >8.5 somewhere in upper 25 cm layer. These are found in various climatic zones in western Siberia, central and south-eastern Europe, Argentina, and China, as well as in many subtropical and tropical areas. Alkaline sodic soils with a structural B horizon form under the influence of comparatively low concentrations of Na compounds capable of alkaline hydrolysis. The definition of these soils is based on the prismatic or columnar structure of the B horizon, which is accompanied by a high proportion of exchangeable Na ions (ESP>15). The structural B horizon always markedly differs from the A horizon, not only in morphology, colour, and structure, but also in its physical, chemical, and biological properties. The B horizon is much more compact than the A. In the B horizon, the accumulation of clay particles and sesquioxides may be observed, maximum values of water-soluble organic matter and ESP occur, while the ratio of SiO₂:R₂O₃ is lowest, signifying that the silicon compounds are comparatively minimal. These often occur in areas where climatic conditions are more humid, e.g., in tropical Africa, some states in the US, Argentina, Canada, Victoria (Australia), and the Hungarian Plain.

Regardless of the profile type, high ESP has an adverse effect on soil physical properties and plant growth. When the ESP is high and the soil EC below the threshold required to flocculate clay particles, slaking, swelling, and the dispersion of clay aggregates will occur. This decline in aggregate stability causes reduced pore space and increases soil strength and promotes crust formation at the surface. This limits the entry, movement, and storage

of air and water within the profile, and can lead to waterlogging and the creation of perched watertables during periods of high rainfall (when sodicity occurs in the B horizon). These physical changes can restrict germination, impair root growth, and limit plant available water. Further,



Figure 6.1 Typical alkali soils from the Indo-Gangetic plains during the dry season (top), and waterlogged sodic soils near a canal during the rainy season (bottom) (CSSRI, Karnal).

toxic concentrations of Na , HCO_3^- and CO_3^- can restrict plant growth and the precipitation out of Ca as less soluble CaCO_3 can cause nutritional deficiencies and exacerbate Na induced toxicity. Under alkaline conditions, the solubility of zinc (Zn), copper (Cu), iron (Fe), and manganese (Mn) also decreases and can lead to nutrient deficiencies. Where sodic soils are also saline, the abundance of electrolytes causes

exosmosis or dehydration, while the presence of high Cl concentrations can be toxic to plants (Rengasamy 2016). Concentrations of organic matter, biological activity and nutrient cycling are also typically low due to inhospitable growing conditions and the high solubility of organic carbon under sodic conditions. The degree of alkali to alkaline hydrolysis is $\text{Na}^+ > \text{Mg}^{2+} > \text{Ca}^{2+} > \text{K}^+$ and $\text{CO}_3^{2-} > \text{HCO}_3^-$ and Cl^- and SO_4^{2-} are near neutral to acidic. Therefore, the combination of ions present produces the degree of alkalinity. An example of a typical alkali soil from the Indo-Gangetic plains during the dry season and a waterlogged sodic soil near a canal during the rainy season are depicted in Figure 6.1.

Reclamation/management of sodic soils

The reclamation and rehabilitation of sodic soils requires that excess Na ions on the soil-exchange complex be replaced with Ca, and Na leached out of the root zone. A brief description of appropriate management practices evolved over the past couple of decades, mainly in India, is described below. These strategies focus on situations where irrigation water is available. Similar strategies can be used in rainfed systems, although remediation is much more difficult and takes longer due to the slower rate of profile leaching - particularly when sodicity occurs in the subsoil. For further detailed discussion of remediation in rainfed systems, the reader is referred to Page et al. (2021).

Irrigation scheduling and management

Appropriate irrigation management can both help ameliorate sodic soils and improve production and water use efficiency. Particularly when combined with the application of amendments or plant-based strategies (discussed below), irrigation can assist with the movement of Na and high salt concentrations of the root zone.

Sodic lands should be properly levelled before irrigation to create a uniform soil surface. Unevelled fields require greater water application, leading to low water application efficiencies and water inundation due to poor infiltration. In sodic soils (pH 9.2, ESP 38), irrigation at critical crop growth stages (especially crown root initiation, tillering, and milk stage) has been observed to lead to significantly higher wheat grain yield (Sharma, Kumar, and Singh 1990). Sprinkler and low energy water application methods using sodic water have also been shown to give 10-20% higher grain yield than farmers' practice (surface irrigation) and use 30-40% less irrigation water, indicating their potential to decrease the water and energy

required to remediate sodic soil (Singh et al. 2016).

The use of gypsum-beds can also reduce sodicity in areas where irrigation with sodic groundwater is common. In this instance, irrigation water is passed through gypsum-beds, and the water dissolves gypsum to increase its Ca ($3\text{--}5 \text{ meq L}^{-1}$) content, making it considerably less harmful than the original Na-saturated water. Long-term studies have shown that the decline in ESP is similar under both bed and soil application of gypsum. However, bed application can lower the costs incurred in grinding and applying gypsum to the soil and in the ponding of water for Na leaching (Bajwa and Choudhary 2014).

Drainage

In a closed basin, provision of subsurface drainage and the use of amendments (e.g., gypsum) is successful in lowering both the water table and the soil pH. Vertical drainage, involving the making of a borehole through which underlying water is extracted by pumping, is a particularly effective measure for controlling the groundwater table. In monsoonal climates, where rainfall averages 600–700 mm during the rainy season, rainwater can be successfully managed by adopting a 3-tier model that (Abrol and Gupta 1990): (i) conserves rainfall in the cropped (paddy) land while it will not be harmful to the standing crop; (ii) directs excess water (after storage) into a dug-out storage pond to be utilised for irrigation in adjacent fields during dry spells; and (iii) discharges excess water into shallow surface drains or nearby canals.

Land shaping models

Sodic areas are prone to water stagnation during high rainfall periods, which may adversely affect crop growth. Repeated instances of waterlogging can also cause secondary sodification; especially where poorly drained lands are underlain with sodic groundwater. Sodic soils suffering from waterlogging with a watertable close to the surface ($\leq 2 \text{ m}$) do not respond to gypsum and other amendments and require alternative approaches to revive productivity. Although subsurface horizontal drainage may lower the watertable, the lack of (natural) gravity outlets can render this technology unworkable, especially where pumped outlets are cost prohibitive for resource poor farmers. Such waterlogged sodic lands can be ameliorated using land shaping methods, whereby fishponds are dug and the less sodic soil deeper in the profile is brought to the surface to create raised and sunken beds. In the raised beds, the watertable falls below a critical depth and

internal drainage is improved. In sunken beds, the watertable lies at ≈ 1 m depth, resulting in reduced upward salt translocation. Cultivation of rice, water chestnut, and integrated rice-fish cultures are economically viable options for the sunken beds, while vegetable crops can be grown on the raised beds for higher returns (Verma et al. 2015). Land modification of waterlogged sodic soil can immediately restore production without the need to apply amendments or excavate huge quantities of soil, which may require large investments. Fishpond-based integrated farming systems have shown potential to produce fish immediately after digging the pond.

Application of amendments for reclamation of alkali soils

Chemical ameliorants and industrial bioproducts

Gypsum. Chemical amendments can be used to replace the excess exchangeable Na in sodic soils with Ca, which promotes flocculation. Gypsum is the most commonly used amendment and is often economically attractive due to its ready availability, price, and easy handling – although due to its widespread use, gypsum supplies are diminishing in some regions. Gypsum adds Ca to the soil and increases electrolytes concentration, thereby increasing soil permeability and preventing the dispersion and swelling of soil clays (Keren 2005). The quantity of gypsum required to neutralise soluble carbonates and replace exchangeable Na [the gypsum requirement (GR)] depends on the quantity of exchangeable Na that must be replaced and the depth to which soil improvement is desired. Abrol Dahiya, and Bhumbla (1975) identified a direct relationship between soil pH and the GR of sodic soils, since the pH is very well correlated with soil ESP (Gupta, Chhabra, and Abrol 1981). Therefore, GR can be calculated on the basis of soil pH. If more tolerant crops are grown, it may also be sufficient to add gypsum to improve only the surface 15-cm soil, thereby reducing the reclamation cost. For highly deteriorated illitic soils of the Indo-Gangetic plains, about 15 Mg ha⁻¹ gypsum incorporated in the top 15 cm was found to be sufficient to allow rice-wheat cropping. Rice, owing to its high tolerance to sodicity and waterlogging, is preferred as first crop during the reclamation process.

Ahmad et al. (2016) evaluated the effects of gypsum at four rates (no gypsum, GR25%, GR50%, and GR75%) in variably textured saline-sodic soils [$EC_e \sim 8$ dS m⁻¹, SAR ~ 44 (mmol L⁻¹)^{1/2} and ESP $\sim 41\%$]. When leached with low-carbonate water [EC 0.39 dS m⁻¹, SAR 0.56 (mmol L⁻¹)^{1/2} and RSC 0.15 mmolc L⁻¹], marked improvements in hydraulic conductivity and decreases in EC_e and SAR were seen with GR25%, GR50% and GR75% in loamy sand, sandy loam, and clayey loam soils, respectively.

Singh, Singh, and Sharma (2009) found that using salt tolerant varieties of rice (CSR 13) and wheat (KRL 19) in place of normal varieties further reduced the GR to 25%. The grain yields obtained in the 25 and 50% GR treatments did not differ significantly when using salt-tolerant varieties, therefore, a 25% GR was recommended with the use of these varieties. Salt tolerant varieties receiving 25% GR not only outperformed their locally popular counterparts but could reduce the cost of gypsum application by ~43% (50% GR).

Synthetic gypsum products. These can include flue gas desulfurisation gypsum (FGDG), phosphogypsum (PG), titanogypsum (TG), fluorogypsum (FG) and citrogypsum (CG). These are by-products of coal-fired power plants (FGDG), and the manufacturing of phosphoric acid (PG), titanium dioxide (TG), hydrofluoric acid (FG) and citric acid (CG) (Sharma et al. 2016; Sharma and Singh 2019; Zhao et al. 2018). These can all add useful amounts of Ca to soil and reduce the impact of sodicity. For example, FGDG has been observed to lead to marked reductions in soil pH, ESP, and EC in the top 0-20 cm soil, with their values decreasing by ~92%, 58%, and 95%, respectively, after 4 years of application (Sharma et al. 2016). FGDG treated soils also showed higher concentrations of Ca^{2+} and SO_4^{2-} and low concentrations of HCO_3^- , and CO_3^{2-} ions. Similarly, the application of powdered PG (5 Mg ha⁻¹) significantly increased the infiltration rate in sodic soil (ESP= 21) from 0.9 mm h⁻¹ to 8.3 mm h⁻¹ by preventing crust formation (Agassi, Morin, and Shainberg 1985). When applied to a saline-sodic soil (pH= 8.5, EC= 4.8 dS m⁻¹) leached with canal water (SAR= 4 and EC= 2.16 dS m⁻¹), PG efficiently reduced the salt content and removed 79% and 60% of the total Na and soluble salts, respectively. PG also releases an appreciable amount of S and P for growing plants (Nayak et al. 2013; Zoca and Penn 2017). However, PG can quickly lose its initial efficiency because of the crystal coating by carbonate films.

Acid-forming, or acidic amendments. These can include sulfuric acid (H_2SO_4), elemental sulfur (S), distillery spent wash (a product of the alcohol industry), and pyrite.

- Sulfuric acid reacts immediately with CaCO_3 in the soil to release Ca for exchange with Na. The H_2SO_4 also reacts with NaHCO_3 to produce less harmful and leachable sodium sulphate (Na_2SO_4) and decreases soil pH. The use of H_2SO_4 has given comparable reductions in SAR to the application of a 25% GR in a rice-wheat-*Sesbania* cropping system (Zia et al. 2007). It has also out-performed

lacto-gypsum, coal-gypsum, and mined-gypsum in improving infiltration rates (21 mm h^{-1} compared with $9\text{--}17 \text{ mm h}^{-1}$ for the other amendments) (Amezketta, Aragüés, and Gazol 2005). However, although it is an efficient amendment, its use has been hampered by difficulty with safe handling under field conditions.

- Elemental S and pyrite can also generate acidity, but these must be oxidised by soil bacteria and react with water to form H_2SO_4 . The formation of sufficient amounts of H_2SO_4 from these materials may take several months, particularly if soil conditions, or a lack of biological activity, hamper oxidation. Elemental S is not soluble in water and its oxidation starts when exposed to air. To minimise the loss of S as sulfur oxide (SO_2 , or other intermediate valence states), S should be applied to moist soil to ensure direct conversion into H_2SO_4 (Chhabra 1996). The oxidation of S is mediated by the bacteria *Thiobacillus thiooxidans*, which requires a warm, well aerated, and moist soil with low pH.
- Distillery spent wash can also be used to produce nutrient rich post-methanation effluent (PME). PME can reduce soil pH and increase soil organic carbon (SOC), nutrient availability and soil enzyme activity, reducing the dependence on chemical fertilisers (Kamble, Dasar, and Gundlur 2017) as well as improving sodic soil conditions (Kaushik et al. 2005).

Nano-enhanced materials or engineered nanoparticles (ENPs). Such compounds can be categorised into metal ENPs (e.g., elemental Ag and Fe), fullerenes (Buckminster fullerenes and nanocones), metal oxides (TiO_2 , CuO, and FeO_2), complex compounds (Co-Zn-Fe oxide), and polymer coated quantum dots (cadmium-selenide and polystyrene) (Dinesh et al. 2012). ENPs, by improving the infiltration rate and soil aggregate stability, decrease surface run-off and structural instability in sodic soils. For example, polymeric aluminium ferric sulphate (PAFS) application decreased the soil pH, EC, bulk density, and CaCO_3 , while increasing the saturated hydraulic conductivity from 0.05 mm d^{-1} to 40.01 mm d^{-1} in a highly dispersed saline-sodic soil. Rice yield in PAFS treated plots was 4.66 Mg ha^{-1} compared to only 0.83 Mg ha^{-1} in the control (Luo et al. 2015). Application of urban solid waste compost with coated sulfur (15 Mg ha^{-1}) and nano iron oxide powder (20 mg kg^{-1}) decreased the pH and SAR of a saline-sodic soil, resulting in increased availability of nutrients to sunflower plants and thus, increased yield (Sharma et al. 2016). Nano-gypsum (100% GR) was promising in reducing the soil pH and ESP compared to mined gypsum applied at GR

25, 50, 75 and 100% GR (Kumar and Thiyageshwari 2018). Among 17 different amendments, hydrolyzed polymaleic anhydride (HPMA) was found to be the most effective in promoting plant growth in a saline-sodic soil by decreasing soil pH, EC, and bulk density, and increasing clay flocculation and water infiltration rate (Wang et al. 2011).

Organic and biological reclamation

Crop residues hold considerable potential for improving soil productivity when used properly. These improve soil aggregation, SOC, and water retention. Application of organic manures and composts also improves the productivity of salt-affected soils by improving soil cation exchange capacity and plant water and nutrient availability. The acidity generated during the decomposition of organic materials can also accelerate the dissolution of precipitated CaCO_3 , providing a source of Ca to replace Na on the exchange complex (Avnimelech et al. 1992). Municipal solid waste compost (MSWC) is increasingly being tapped as a potential source of organic matter to improve crop productivity on sodic soils, while also providing a solution to the problem of urban waste disposal. For example, MSWC improved the activity of many soil enzymes (dehydrogenase, alkaline phosphatase, and urease), microbial biomass carbon, and nutrient availability in a saline-sodic soil (EC_e 7.2 dS m^{-1} , pH 8.4), leading to significant increases in the grain and straw yields of mustard and pearl millet crops (Meena et al. 2016).

Growing leguminous green manure plants, such as *Sesbania aculeata/cannabina* and *Crotalaria juncea*, can also help ameliorate sodic soils. These not only improve soil physical properties, but also fix nitrogen for subsequent crops. A combination of plant growth and amendments can also be effective. For example, application of PG (10 Mg ha^{-1}) along with the cultivation of Kallar grass (*Leptochloa fusca*) resulted in greater reductions in soil pH and ESP compared to PG applied alone (Nayak et al. 2013). In these situations, the release of H^+ by plant roots can acidify the rhizosphere and increase the release of Ca from CaCO_3 . The action of deep-rooted plant species can also help loosen compacted soil layers and improve conditions for water movement. These changes, particularly when combined with irrigation, can improve profile leaching and reduce ESP and EC in the root zone.

A range of microorganisms have been tested for overcoming salt-stress in a cost-effective and environmentally friendly way. For example, blue green algae lessen sodicity stress by decreasing the soil pH, EC_e , and ESP with concurrent improvements in water permeability, soil aggregation, and nutrient availability; especially in moderately affected soils (Rao 1998). A

low-cost microbial bioformulation “CSRBIO”, based on a consortium of *Bacillus pumilus*, *Bacillus thuringensis*, and *Trichoderma harzianum* on dynamic media, has also been developed. This acts as a soil conditioner that mobilises nutrients and significantly increases the productivity of rice, banana, vegetables, and gladiolus in sodic soils and also facilitates soil amelioration (Damodaran et al. 2013).

Increasing emphasis has also been placed on the combined use of bio-augmented materials because use of two or more organic sources may give better results than the application of a single amendment. One such bio-augmented formulation (SF_{OA}) consisted of vermicompost (inoculated with plant growth promoting fungi), pressmud (a by-product of sugar production), and neem (*Azadirachta indica*) seed cake, applied along with a recommended dose of fertiliser. This combination significantly increased wheat grain yield and quality in a silty loam sodic soil (pH 9.2 and ESP 60) compared to the control treatment and improved SOC and soil enzyme activities (Srivastava et al. 2016). Similarly, favourable effects of combined SF_{OA} and gypsum use were ascribed to decreased soil bulk density and pH, coupled with increased water-holding capacity, SOC, and soil enzyme activities (Gupta et al. 2016). Likewise, the effects of FGDG (33.6 Mg ha⁻¹ and 66.2 Mg ha⁻¹) and sugar beet (*Beta vulgaris*) biochar (16.8 Mg ha⁻¹), either alone or in combination, were studied in a saline-sodic soil. Biochar application increased SOC and soil respiration rate, and when used in conjunction with FGDG, EC_e increased from 3.5 to 7.7 dS m⁻¹ while SAR_e decreased from 16 to 9 (Schultz et al. 2017). The individual and synergic effects of biochar (B), humic substances (HS) and gypsum (G) were studied on soil properties and quinoa (*Chenopodium quinoa*) plant growth in a saline-sodic soil. Soil EC_e, SAR, and ESP decreased significantly in all the treatments. In comparison to the control, ESP decreased by 11-fold with gypsum and by 9-13-fold in rest of the treatments. Similarly, soil microbial biomass increased by 112 and 322% in the B+HS+G treatment in quinoa genotypes AZ-51 and AZ-103, respectively, implying that the combined application of biochar, humic substances, and gypsum was an effective means for reclaiming degraded saline-sodic soil (Alcívar et al. 2018).

Alternate management strategies for improving soil health and crop production

A variety of strategies are required for amelioration of sodic lands to allow their productive utilisation. These can include both reclamation by applying amendments, as discussed above, but also the utilisation of salt-

tolerant forest and fruit trees, grasses, medicinal plants, and arable crops. Most of the options discussed above for the management of sodic soils are essentially “reactive” in nature and based on the application of amendments, which are costly and may not always be available. Contrarily, the “adaptive” approach to improving the productivity of salt-affected soils involves growing salt-tolerant crops and cultivars, either in isolation or in combination with agroforestry. The integrated use of amendments and salt-tolerant species/crops/cultivars can save up to 75% of reclamation costs in sodic areas (Sharma et al. 2016). Using this approach, several plant species gradually remediate the calcareous sodic soils by improving the dissolution of native calcite to release sufficient Ca for removing excess Na. Some of these interventions are discussed here in brief.

Agroforestry and plant-based interventions

Many investigations have shown that developing agroforestry systems on sodic soils could be a feasible way to generate additional returns while simultaneously improving the quality of degraded and barren soil (Figure 6.2). The choice of species for alkali lands is determined by the ability of tree species that can survive and withstand the adverse conditions associated with excess sodicity. There are very few tree species that can grow on highly sodic soils (pH >10), although in India, *Prosopis juliflora*, *Acacia nilotica*, *Casuarina equisetifolia*, *Tamarix articulata*, *Eucalyptus tereticornis*, and *Parkinsonia aculeata* have demonstrated a higher tolerance (Dagar, Singh, and Singh 2001) and can produce between 19.2-56.5 Mg ha⁻¹ air-dry biomass (Singh et al. 2008).

Trees not only provide timber, fodder, and fruits and contribute towards environmental services such as carbon sequestration, improvement in the local environment, biodiversity, and soil health, but also improve the physicochemical properties of soils including bulk density, soil porosity, infiltration rate, pH, organic carbon, ESP, and available P and K (Khan and Shukla 2003; Singh and Gill 1992; Singh, Singh, and Tomar 1993; Singh, Singh, and Sharma 2011). Tree-based reclamation systems can also have a major impact on soil biological properties. For example, microbial biomass carbon, nitrogen, and phosphorus all increased by approximately three-fold in a sodic soil treated with afforestation (Singh and Goel 2012).

Tolerant grass species can also be used for fodder production and to improve conditions in highly sodic soils. Based on series of long-term experiments (Kumar 1998; Singh and Dagar 2005) it was found that Kallar grass (*Leptochloa fusca*) was tolerant to highly sodic and waterlogged soil, producing 15-20 Mg ha⁻¹ dry forage without applying any amendment. Other tolerant grasses include Rhode grass (*Chloris gayana*), Gutton panic

(*Panicum maximum*), Para grass (*Brachiaria mutica*), *Panicum antidotale*, *P. laevifolium*, *P. purpurium*, and *Setaria anceps*, which are suitable for soils up to pH 9.6. These grasses can also be grown successfully in conjunction with tree species. On sodic vertisols with ESP 40, grasses *Leptochloa fusca*, *Brachiaria mutica*, and *Vetiveria zizanioides* could remove 144.8, 200.0 and 63.5 kg ha⁻¹ Na from the soil, respectively within three years of growth (AICRP 2002-04).



Figure 6.2 Afforestation of highly alkali soil (pH >10). Original soil (in front) afforested by trees in background [Source: CSSRI Karnal; earlier used by Singh and Dagar (2005) and Dagar (2014)].

Many fruit trees also can be grown with success on high pH (10) sodic soils using wider spaces (row to row 5-6 m, and plant to plant 4-5 m), with arable crops cultivated in the interspaces (Dagar 2014). In one trial, Egyptian clover (*Trifolium alexandrinum*), wheat, rice, onion, and garlic were grown successfully in the interspaces of Karonda (*Carissa carandas*), pomegranate (*Punica granatum*), goose berry (*Emblica officinalis*), guava (*Psidium guajava*), Jamun (*Syzygium cuminii*), and Ber (*Ziziphus mauritiana*). The yield was 10.6 to 16.7 Mg ha⁻¹ forage from *Leptochloa fusca* grass, 1.6 to 3.0 Mg ha⁻¹ grains from wheat, 1.8 to 3.4 Mg ha⁻¹ onion bulb, and 2.3 to 4.1 Mg ha⁻¹ garlic when harvested from different tree systems (Tomar, Dagar, and Singh 2004).

Some medicinal and flower-yielding plants are also salt-tolerant and can be cultivated on alkaline sodic sites, e.g., *Matricaria chamomilla*, *Catharanthus roseus*, and *Chrysanthemum indicum* (Dagar et al. 2009). Some other medicinal and commercial plants such as lemon grass (*Cymbopogon flexuosus*), rosa grass (*C. martini*), vetiver (*Vetivaria zizanioides*), psyllium (*Plantago ovata*), periwinkle (*Catharanthus roseus*), and *Ocimum sanctum* also can be successfully cultivated on moderately alkali soils with pH up to 9.6 (Dagar et al. 2004; Dagar, Yogender-Kumar, and Tomar 2006). Dagar et al. (2015) have reported liquorice (*Glycyrrhiza glabra*) as a profitable option on different pH soils, with this species also ameliorating soil sodicity to a greater extent.

Genotypic variability in different crops is also now exploited to develop crop cultivars with greater tolerance to sodic soil conditions, and salt-tolerant cultivars of rice, wheat, mustard, and chickpea have been developed (Table 6.1).

Table 6.1 Sodicity-tolerant varieties of some crops

Crops	Tolerant varieties	Sodic tolerance limit (pH)
Rice	CSR 10	9.8-10.2
	CSR 13, CSR 23, CSR 46, CSR 60	9.4-9.8
	CSR 27	9.9
	Basmati CSR 30	9.5
	CSR 36	9.9
	CSR 43	10.0
	KRL1-4, KRL 19	9.2-9.3
Wheat	KRL 210, KRL 213, KRL 283	9.3
	WH 157, Raj 3077	9.2-9.3
Barley	CSB 1, CSB 2, CSB 3, Ratna	9.3
Indian mustard	CS 52, CS 54, CS 56	9.0 - 9.3
	CS58, CSR 60*	9.5
	Pusa bold, Varuna, Kranti	8.8-9.0
Gram	Karnal chana 1	8.8-9.0
Sugar beet	Ramonskaaya 06, Maribo Resistapoly,	9.5-10.0
	Polyrava E	
Sugarcane	Co 453, Co 1341, CO 6801, CO 62329, CO 1111	9.0

Source: <https://cssri.res.in/>, Dagar (2014).

Nutrient management

Sodic soils may be nutrient deficient (e.g., N, Ca, K, and P), and have low organic matter content, so appropriate nutrient supply can improve crop growth and yield in some instances. The efficiency of N fertiliser is generally low in alkaline sodic soils, partly due to high rates of loss via volatilisation, and partly due to sub-optimal plant growth leading to low N uptake. In some instances, additional N supply to compensate for the losses of N can increase yield (Tiwari, Sharma, and Tripathi 1989). For example, in loamy alkali soils, a three-year study revealed that increased N doses of up to 200 kg ha⁻¹ resulted in significant increase in wheat and paddy yields (Sharma, Sharma, and Minhas 1998). However, in black alkali soils, wheat, paddy, cotton, and barley responded only up to 120 kg N ha⁻¹. Among different sources, ammonium sulphate proved to be a better source of N than other common N fertilisers (Khaddar, Ray, and Singh 1987).

The availability of phosphates in sodic soils may also be a limiting factor for plant growth. In sodic soils amended with gypsum, octa-Ca phosphate is reported to be a dominant form of P. Thus, the available P contents of alkaline sodic soils after gypsum applications show a decreasing trend. The type of amendment used for reclamation of alkali soils thus plays an important role in determining the responses to applied P (Sharma, Sharma, and Minhas 1998). Experiments with rice-wheat on alkali soils amended with gypsum and pyrites showed better responses to applied P, especially in wheat, when soil was amended with pyrites. The magnitude of the decrease in available P varied with the nature of amendment used. An increase in the rate of applied gypsum decreased the water-soluble P greatly but had little effect on the strongly adsorbed P fraction (NaOH-P) or Ca-bound P (HCl-P) (Swarup 1998).

Due to their mineralogy, many sodic soils may have adequate reserves of available K. However, some alluvial alkali soils have very low levels of available soil K₂O (72.5 mg kg⁻¹) (Sharma, Sharma, and Minhas 1998) and applications of K will increase rice and wheat yields. Even when the soluble K fraction is high to very high, and no responses to K applications are observed, it is necessary to maintain a high K supply to regulate the Na: K balance in plants.

Soluble and exchangeable form of Ca becomes deficient in both high (sodic soils) and low pH (acid) soils and require supplementation through amendments. Absolute Ca concentration and the Na:Ca ratio are both important indices of plant response to sodic conditions and ultimately yield. Amendments like gypsum are a direct source of Ca, but acid (H₂SO₄) and acid forming (S, pyrites) amendments also mobilise Ca from precipitated CaCO₃, thereby ensuring adequate supply of available Ca in calcareous

soils.

Zinc precipitates as hydroxide and carbonate in alkali soils. Due to high pH, the presence of CaCO_3 , high soluble P, and low organic matter content, sodic soils often contain less than 0.6 mg kg^{-1} DTPA extractable Zn (Katyal, Randhawa, and Sharma 1980). The addition of amendments can also decrease the availability of added Zn. However, Zn supplementation can improve plant growth in sodic soils by enhancing absorption of Ca and K, and thereby widening the Ca: Na and K:Na ratios in plants. Rice, though fairly tolerant to sodicity, is sensitive to Zn deficiency, which may cause stunted growth and severe yield reductions. In soils amended with $10\text{-}15 \text{ Mg ha}^{-1}$ gypsum, addition of $10\text{-}20 \text{ kg ha}^{-1}$ of Zn sulphate was enough to meet plant Zn requirements (Singh, Chhabra, and Abrol 1987) and accordingly, application of $20\text{-}40 \text{ kg Zn ha}^{-1}$ was recommended during reclamation. In black alkali soils, applications of 50 kg ha^{-1} of ZnSO_4 had a positive residual effect on the succeeding wheat crop (Sharma, Sharma, and Minhas 1998).

Research gaps and conclusions

Sodicity causes land degradation and impairs soil health, which limits crop yield and has adverse impacts on farmer livelihood and security. Alkali/sodic soils can be ameliorated using gypsum. However, gypsum is becoming increasingly scarce worldwide and affordable alternatives are required. Some alternative amendments, such as sulfuric acid, pyrite, PG, FGDG, composts, biochar, and nano-enhanced materials have been identified and used successfully to ameliorate sodic soils in some instances. However, many of these have inherent problems due to difficulties around cost, availability, transportation, and handling, and some can contain toxic elements and compounds. Consequently, greater research is required.

The potential of phytoremediation and bioremediation to remediate sodic areas also needs greater assessment. In particular, the use of agroforestry to improve areas of highly underproductive or barren land is a promising approach that has great potential to help increase production and food availability (and thus meet the United Nations objectives of Sustainable Development). Further development of sodicity, salinity, and waterlogging tolerant clones of the important fast-growing trees, the domestication of high value salt-tolerant native halophytes, and the development of more crop cultivars and fruit trees tolerant to sodicity is likely to significantly improve production on sodic soils. The identification of suitable market and policy support mechanisms for emerging crops to allow farmers to sell their product profitably is also required.

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CHAPTER 7

SOIL SALINITY

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Abstract

Soil salinity consists of the major dissolved inorganic solutes in the soil solution, including major cations (i.e., Na^+ , K^+ , Mg^{+2} , Ca^{+2}) and major anions (i.e., Cl^- , HCO_3^- , NO_3^- , SO_4^{-2} and CO_3^{-2}). Salinity has a major impact on global agricultural productivity, with an estimated global economic loss of \$27.3 billion USD in 2013. Soil salinity can result in reduced plant growth, reduced yields, and in severe cases, crop failure. Salinity reduces the osmotic potential making it more difficult for the plant to extract water, increases specific-ion toxicity, upsets the nutritional balance of plants, and/or influences soil tilth and permeability. Measurement and management of soil salinity are the major edaphic and anthropogenic factors, respectively, for minimising the impact of salinity on crop productivity. Corwin and colleagues at the USDA-ARS U.S. Salinity Laboratory have developed protocols and guidelines for measuring soil salinity across multiple scales. The primary means of soil salinity management is the leaching of salts through the root zone to salinity levels that will not cause a yield decrement in the crop or will not cause a significant yield decrement, usually set at 10% or less. This chapter provides a discussion of the measurement and irrigation management of soil salinity. The topics include (1) basic soil salinity, (2) apparent soil electrical conductivity directed soil sampling to measure soil salinity at the field scale, (3) plant salt tolerance, (4) water and salt balance, (5) leaching requirement (LR), (6) steady-state vs transient LR models, and (7) ramifications of the use of transient LR models over traditional steady-state LR models. This information can be effectively used by Natural Resources Conservation Service (NRCS) staff, agricultural and irrigation consultants, natural resource specialists, and soil

scientists in the university, government, and private sectors.

Keywords: Soil salinity, agricultural productivity, leaching, apparent soil electrical conductivity, irrigation, proximal sensors, electromagnetic induction, electrical conductivity, electrical resistivity.

Introduction

Soil salinity consists of the major dissolved inorganic solutes in the soil solution, including soluble and readily dissolvable salts. The soluble and readily dissolvable salts include charged species (e.g., Na^+ , K^+ , Mg^{+2} , Ca^{+2} , Cl^- , HCO_3^- , NO_3^- , SO_4^{-2} and CO_3^{-2}), non-ionic solutes, and ions that combine to form ion pairs. The primary source of salts in soil and water is the geochemical weathering of rocks from the earth's upper strata. Atmospheric deposition and anthropogenic activities serve as secondary sources.

The accumulation of soil salinity is a consequence of a variety of processes, some of which are illustrated in Figure 7.1. Salt accumulation in soil is most common in arid and semi-arid climates where evaporation exceeds precipitation. In arid and semi-arid areas, salts can accumulate at the soil surface when the depth to the water table is less than 1-1.5 m, depending on the soil texture. The accumulation of salts at the soil surface is the consequence of the upward flow of water and subsequent transport of salts due to capillary rise driven by evaporative process. However, the most common cause for the accumulation of salts is evapotranspiration (ET) by plants, which results in an increase in salt concentration with depth through the root zone (see graph in Figure 7.1) and the accumulation of salts below the root zone. The level of salt accumulation within and below the root zone due to ET depends upon the fraction of irrigation or precipitation that flows beyond the root zone, referred to as the leaching fraction (LF). As the LF increases the total salts within the root zone decrease due to their removal from the root zone by leaching. A third process, which is common in the northern Great Plains of the USA, is the formation of saline seeps. In general, saline seeps form downslope of recharge areas in locations where discharge is occurring because of the presence of a low conductivity layer and shallow water table (Figure 7.1). Salts are leached from the upslope recharge area, which tends to be an area of higher conductivity than the downslope discharge area. Once the water and salts from upslope reach the downslope low conductivity layer, they accumulate and are forced to the surface by evaporation.

The accumulation of soil salinity can result in reduced plant growth, reduced yields, and in severe cases, crop failure. Salinity limits water uptake

by plants by reducing the osmotic potential. Salinity may also cause specific-ion toxicity (e.g., Na^+ ion toxicity) or upset the nutritional balance of plants. Furthermore, the salt composition of soil water influences the composition of cations on the exchange complex of soil particles, which influences soil permeability and tilth (e.g., when the accumulation of excess Na^+ ions create sodic soils and leads to dispersion).

What Causes Salt Accumulation?

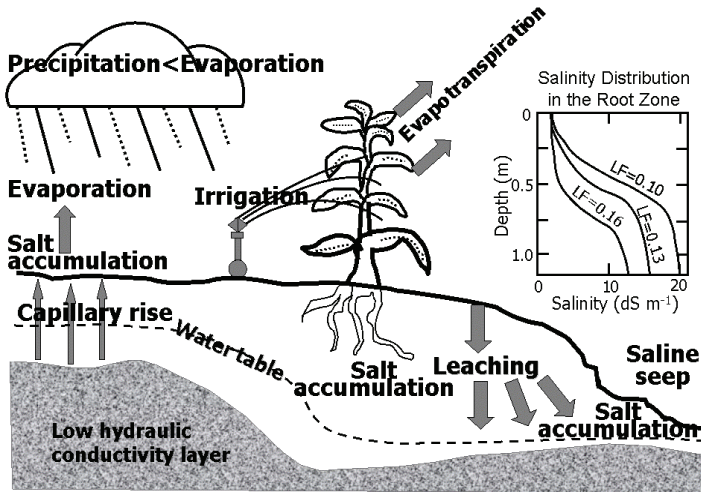


Figure 7.1 Various examples of how salts accumulate in soil.

Currently, there is no worldwide inventory of salt-affect soils based on quantitative measurements of salinity and sodicity because of the tremendous scale of such an undertaking and the prohibitive cost, resulting in differences of past global assessments. Only estimates of soil salinity at a global scale are available. Squires and Glenn (2009) estimated the global extent of saline soils to be 412 Mha. This agrees with the estimate by FAO (<http://www.fao.org/soils-portal/en/>) of 397 Mha. According to Rengasamy (2006) and Squires and Glenn (2009), approximately one-third of all agricultural lands are becoming saline, with over 100 countries experiencing problems from salt-affected soils. It is estimated that over the past 20 years the world has lost an average of 2 000 ha of farmland daily to salt damage (Qadir et al. 2014). In the conterminous USA, approximately 30% of the soil is regarded as having moderate to severe potential for salinity issues (Tanji 1996). In California alone, an estimated 1.72 Mha (29%) of all non-

federal land is either saline or sodic (Tanji 1996).

In the 1990s, a conservative estimate of the cost of salinity to agriculture was approximately \$12 billion USD per year (Ghassemi et al. 1995). More recently, an inflation-adjusted cost of salt-induced land degradation was estimated at \$441 ha⁻¹, resulting in an estimated global economic loss of \$27.3 billion USD for 2013 (Qadir et al. 2014). Welle and Mauter (2017) estimated an income loss due to salinity within California alone of \$3.7 billion USD in 2014.

The two greatest concerns for minimising the impact of soil salinity on crop production are measurement and management. It is the objective of this chapter to provide a discussion of the measurement of soil salinity at field scale and of field-scale leaching as a management tool to control soil salinity in the root zone. The topics covered include (1) a basic introduction to soil salinity, (2) an introductory discussion of apparent soil electrical conductivity (EC_a) directed soil sampling to measure soil salinity at field scale, (3) plant salt tolerance, (4) water and salt balance, (5) leaching requirement (LR), (6) steady-state vs transient LR models, and (7) ramifications of the use of transient LR models over traditional steady-state LR models.

Root zone soil salinity measurement

The most common technique for the measurement of soil salinity is laboratory analysis of the aqueous extracts of soil samples. Soil salinity is quantified in terms of the concentration of total salts in soil extracts, measured either directly through chemical analysis, or indirectly through the measurement of electrical conductivity (EC). The chemical species of primary interest in salt-affected soils include major cations (i.e., Na⁺, K⁺, Mg⁺², and Ca⁺²) and anions (i.e., Cl⁻, HCO³⁻, SO⁴⁻², and CO³⁻²) in the soil solution, exchangeable cations (i.e., Na⁺, K⁺, Mg⁺², and Ca⁺²), and precipitates of calcium carbonate (lime) and calcium sulfate (gypsum). Other soil properties of concern include pH, water content of the saturation paste, sodium adsorption ratio (SAR), and exchangeable sodium percentage (ESP). Detailed analytical techniques for measuring all these salinity-related properties can be found in *Methods of Soil Analysis* (Part 3, Sparks 1996; Part 4, Dane and Topp 2002; Corwin and Yemoto 2020). However, a chemical analysis of the salinity-related properties of primary concern is often too labour and cost intensive to be practical, particularly at field scale; consequently, salinity is often measured by EC.

It is well-known that the EC of water is a function of its chemical salt composition and total salt concentration (U.S. Salinity Laboratory Staff

1954). In the laboratory, soil salinity is commonly determined from the measurement of the EC of soil solution extracts, where the current-carrying capacity of the soil solution is proportional to the concentration of ions in the solution. The total concentration of soluble salts in soil is measured by EC of the soil solution in dS m^{-1} . Temperature influences EC: consequently, EC must be referenced to a specific temperature to permit comparison. Electrolytic conductivity increases at a rate of approximately 1.9% per degree centigrade increase in temperature. Customarily, EC is expressed at a reference temperature of 25 °C. The EC measured at a particular temperature t (in °C), EC_t , can be adjusted to a reference EC at 25°C, $EC_{25^\circ\text{C}}$, using Eq. [1] from Handbook 60 (U.S. Salinity Laboratory Staff 1954):

$$EC_{25^\circ\text{C}} = f_t \cdot EC_t \quad [1]$$

where, $f_t = 0.4470 + 1.4034 \exp(-t/26.815)$ from Sheets and Hendrickx (1995).

Historically, four principal methods have been used for measuring soil salinity: (1) the EC of soil solution at or near field capacity, or from extracts at higher than normal water contents (i.e., including saturation and at soil to water ratios of 1:1, 1:2, and 1:5), or of a saturation paste; (2) in situ measurement of electrical resistivity (ER); (3) non-invasive measurement of EC with electromagnetic induction (EMI); and most recently, (4) in situ measurement of EC with time domain reflectometry (TDR).

Electrical conductivity

To determine the EC of a soil solution extract, the solution is placed in a cell containing two electrodes of constant geometry and distance of separation. An electrical potential is imposed across the electrodes, and the resistance of the solution between the electrodes is measured. The measured conductance is a consequence of the solution's salt concentration and the electrode geometry, whose effects are embodied in a cell constant. At constant potential, the current is inversely proportional to the solution's resistance as shown in Eq. [2]:

$$EC_t = k/R_t \quad [2]$$

where, EC_t is the electrical conductivity of the soil solution in dS m^{-1} at temperature t (EC), k is the cell constant, and R_t is the measured resistance in ohms at temperature t . One dS m^{-1} is equivalent to one mS cm^{-1} and one

mmho cm^{-1} , where mmho cm^{-1} are the obsolete units of EC.

Except for the measurement of EC of a saturated soil paste (EC_p), the determination of soluble salts in disturbed soil samples consists of two basic steps: (1) preparation of a soil-water extract, and (2) the measurement of the salt concentration of the extract using EC. Customarily, soil salinity has been defined in terms of laboratory measurements of the EC of the extract of a saturated soil paste (EC_e). This is because it is impractical for routine purposes to extract soil water from samples at typical field water contents; consequently, soil solution extracts must be made at saturation or higher water contents. The saturation paste extract is the lowest soil-to-water ratio that can be easily extracted with vacuum, pressure, or centrifugation, while providing a sample of sufficient size to analyse. The water content of a saturation paste is roughly twice the field capacity for most soils. Furthermore, EC_e has been the standard measure of salinity used in salt-tolerance plant studies. Most data on the salt tolerance of crops have been expressed in terms of the electrical conductivity of the saturation paste extract (Bresler et al. 1982; Maas 1986).

Unfortunately, the partitioning of solutes over the three soil phases (i.e., gas, liquid, solid) is influenced by the soil-to-water ratio at which the extract is made so that the ratio needs to be standardised to obtain results that can be applied and interpreted universally. Commonly used extract ratios, other than a saturated soil paste, are 1:1, 1:2, and 1:5 soil-to-water mixtures. These extracts are easier to prepare than saturation paste extracts. With the exception of sandy soils, soils containing gypsum, and organic soil, the concentrations of salt and individual ions are approximately diluted by about the same ratio between field conditions and the extract for all samples, which allows conversions between water contents using dilution factors. The conversion of EC from one extract to another is commonly done using a simple dilution factor. For example, if the gravimetric saturation percentage (SP) is 100%, then $\text{EC}_e = \text{EC}_{1:1} = 5 \cdot \text{EC}_{1:5}$ or if $\text{SP}=50\%$, then $\text{EC}_e = 2 \cdot \text{EC}_{1:1} = 10 \cdot \text{EC}_{1:5}$. However, this is not recommended because of potential dissolution-precipitation reactions that may occur. At best, the use of a dilution factor to convert from one extract to another is an approximation.

Any dilution above field water contents introduces errors in the interpretation of data. The greater the dilution is, the greater the deviation between ionic ratios in the sample and the soil solution under field conditions. These errors are associated with mineral dissolution, ion hydrolysis, and changes in exchangeable cation ratios. Soil samples containing gypsum deviate the most because the calcium and sulfate concentration remain nearly constant with sample dilution, while the concentrations of other ions decrease with dilution. The standardised

relationship between the extract and the conditions of the soil solution in the field for different soils is lost with the use of soil-to-water above saturation. However, the recent development of software by Suarez and Taber (2007) allows for the accurate conversion from one extract ratio to another, provided sufficient chemical information is known (i.e., knowledge of the major cations and anions and presence/absence of gypsum). The disadvantage of determining soil salinity using a soil sample is the time and labour required, which translates into high cost. However, there is no more accurate way of measuring soil salinity than with extracts from soil samples.

Soil salinity can also be determined from the measurement of the EC of a soil solution (EC_w), where the water content of the soil is less than saturation, usually at field capacity. Ideally, EC_w is the best index of soil salinity because this is the salinity actually experienced by the plant root. Nevertheless, EC_w has not been widely used to express soil salinity for various reasons: (1) it varies over the irrigation cycle as the soil water content changes; and (2) the methods for obtaining the soil solution are too labour, time, and cost intensive to be practical (Rhoades et al. 1999). For disturbed soil samples, soil solution can be obtained in the laboratory by displacement, compaction, centrifugation, molecular adsorption, and vacuum- or pressure-extraction methods. For undisturbed soil samples, EC_w can be determined with a soil-solution extractor, often referred to as a porous cup extractor, or using an in situ, imbibing-type porous-matrix salinity sensor.

Obtaining the EC of a soil solution when the water content is at or less than field capacity, which is common in the field, is considerably more difficult than extracts for water contents at or above saturation because of the pressure or suction required to remove the soil solution at field capacity and lower water contents. The EC of the saturated paste is the easiest to obtain followed by the EC of extracts greater than SP, followed by the EC of extracts less than SP. However, it is EC_e above all others that is most preferred; consequently, either measuring EC_e or being able to relate the EC measurement to EC_e is critical.

Electrical resistivity

Because of the time and cost of obtaining soil solution extracts and the lag time associated with porous ceramic cups, developments in the measurement of soil EC shifted in the 1970s to the measurement of the soil electrical conductivity of the bulk soil, referred to as apparent soil electrical conductivity (EC_a). The EC_a provides an immediate, easy-to-take measurement of conductance with no lag time and no need to obtain a soil

extract. However, EC_a is a complex measurement that has been misinterpreted and misunderstood as it is a measure of the electrical conductance of the bulk soil and not just a measure of the conductance of the soil solution, which is the desired measurement since the soil solution is the soil phase that contains the salts affecting plant roots. The techniques of ER, EMI, and TDR measure EC_a . The most recent reviews of research related to EC_a measurement of soil salinity can be found in Rhoades et al. (1999), Corwin (2005), Corwin and Lesch (2005a), and Corwin and Scudiero (2019).

Electrical resistivity methods involve the measurement of the resistance to current flow across four electrodes inserted in a straight line on the soil surface at a specified distance between the electrodes (Corwin and Hendrickx 2002). The electrodes are connected to a resistance meter that measures the potential gradient between the current and potential electrodes (Figure 7.2).

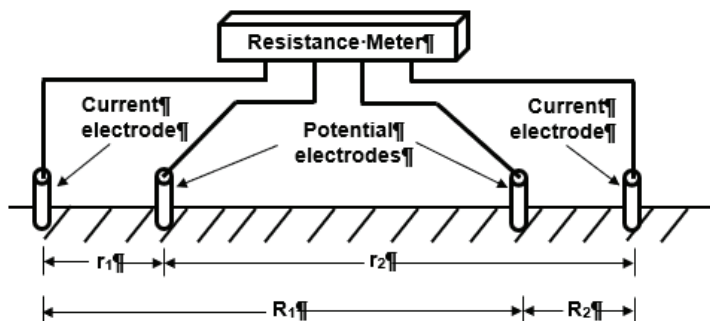


Figure 7.2 Schematic of four-electrode probe electrical resistivity used to measure apparent soil electrical conductivity. Taken from Corwin and Hendrickx (2002).

Electrical resistivity is an invasive technique that requires good contact between the soil and four electrodes inserted into the soil; consequently, it produces less reliable measurements in dry or stony soils than a non-invasive measurement such as EMI. Nevertheless, ER has a flexibility that has proven advantageous for field application, i.e., the depth and volume of measurement can be easily changed by altering the spacing between the electrodes. A distinct advantage of the ER approach is that the volume of measurement is determined by the spacing between the electrodes, which makes a large volume of measurement possible. For example, a 1 m inter-electrode spacing for a Wenner array results in a volume of measurement of

more than 3 m^3 . This large volume of measurement integrates the high level of local-scale variability often associated with EC_a measurements.

Because EC_e is regarded as the standard measure of salinity, a relation between EC_a and EC_e is needed. The relationship between EC_a and EC_e is linear when EC_a is above 2 dS m^{-1} and is dependent on soil texture as shown in Figure 7.3. Rough approximations of EC_e from EC_a in dS m^{-1} when $\text{EC}_a \geq 2 \text{ dS m}^{-1}$ are: $\text{EC}_e \approx 3.5 \text{ EC}_a$ for fine-textured soils, $\text{EC}_e \approx 5.5 \text{ EC}_a$ for medium-textured soils, and $\text{EC}_e \approx 7.5 \text{ EC}_a$ for coarse-textured soils. For $\text{EC}_a < 2 \text{ dS m}^{-1}$ the relation between EC_a and EC_e is more complex as other conductive properties (e.g., water and clay content) and properties influencing conductance (e.g., bulk density) have greater influence than salinity on EC. Below EC_a 2 dS m^{-1} it is recommended that the relation between EC_a and EC_e is established by calibration using soil samples taken at a minimum of 3-4 locations within a study area.

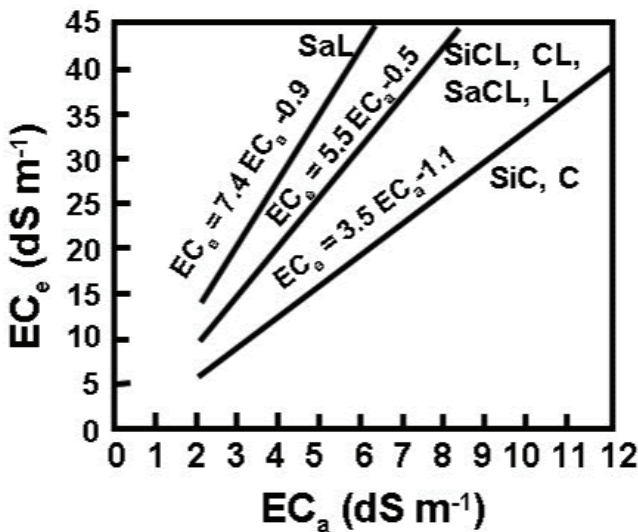


Figure 7.3 Relationships between EC_a and EC_e for representative soil types found in the northern Great Plains, USA. Modified from Rhoades and Halvorson (1977). SaL=sandy loam, SiCL=silty clay loam, CL=clay loam, SaCL=sandy clay loam, L=loam, SiC=silty clay, and C=clay.

Electromagnetic induction

Apparent soil electrical conductivity can be measured non-invasively with EMI. The most used EMI conductivity meters in soil science and in vadose zone hydrology are the Geonics² EM-31 and EM-38 and the DUALEM-2¹. The EM-38 has had considerably greater application for agricultural purposes because the depth of measurement corresponds roughly to the root zone (i.e., generally 1 to 1.5 m.). When the instrument is placed in the vertical coil configuration (EM_v, i.e., coils are perpendicular to the soil surface) the depth of measurement is about 1.5 m and in the horizontal coil configuration (EM_h, i.e., coils are parallel to the soil surface), the depth of the measurement is 0.75-1.0 m. The EM-31 has an intercoil spacing of 3.66 m, which corresponds to a penetration depth of 3 and 6 m in the horizontal and vertical dipole orientations, respectively, which extends well beyond the root zone of agricultural crops. However, the EM-38 requires calibration, while the DUALEM-2 does not. Further details about the EM-31 and EM-38 equipment and its operation are discussed in Hendrickx and Kachanoski (2002). Documents concerning the DUALEM-2 can be found online at <http://www.dualem.com/documents.html> (verified 6 May 2021).

Both ER and EMI are rapid and reliable technologies for the measurement of EC_a, each with advantages and disadvantages. EMI is noninvasive, so it can be used on dry and stony soils that would not be amenable to invasive ER equipment. While ER has easier instrument calibration. However, EC_a measured with EMI is a depth-weighted value that is nonlinear, whereas ER provides an EC_a measurement that is more linear with depth. Measurements of EC_a by ER and EMI at the same location and over the same volume of measurement are thus not comparable because of the difference of their response functions with depth.

Time domain reflectometry

The TDR technique is based on the time for a voltage pulse to travel down a soil probe and back, which is a function of the dielectric constant (ϵ) of the porous media being measured. Dalton (1992) demonstrated the utility of TDR to also measure EC_a. The measurement of EC_a with TDR is established from the attenuation of the applied signal voltage as it traverses the medium of interest with the relative magnitude of energy loss related to

¹ Geonics Limited, Mississauga, Ontario, Canada and DUALEM Inc., Milton, Ontario, Canada, respectively. Product identification is provided solely for the benefit of the reader and does not imply the endorsement of the USDA.

EC_a (Wraith 2002).

Advantages of TDR for measuring EC_a include (1) an ability to measure both soil water content and EC_a, (2) an ability to detect small changes in EC_a under representative soil conditions, (3) the capability of obtaining continuous unattended measurements, and (4) a lack of a calibration requirement for soil water content measurements in many cases (Wraith 2002). However, TDR has not been widely used to measure salinity, whether in the laboratory or in the field. Although it has been demonstrated to compare closely with other accepted methods of EC_a measurement (Heimovaara et al. 1995; Mallants et al. 1996; Spaans and Baker 1993; Reece 1998), it is still not sufficiently simple, robust, or fast enough for the general needs of field-scale soil salinity assessment (Rhoades et al. 1999).

Edaphic factors influencing the EC_a measurement

The earliest field applications of geophysical measurements of EC_a in soil science involved the determination of salinity through the soil profile of arid zone soils (Halvorson and Rhoades 1976; Rhoades and Halvorson 1977; de Jong et al. 1979; Cameron et al. 1981; Rhoades and Corwin 1981; Corwin and Rhoades 1982, 1984; Williams and Baker 1982). However, it became apparent that the measurement of EC_a in the field to infer soil salinity was more complicated than initially anticipated due to the complexity of current flow pathways arising from the complex interaction of the conductive properties influencing the EC_a measurements and from the spatial heterogeneity of those conductive properties (Corwin and Lesch 2003, 2005a). As shown in Figure 7.4, three parallel pathways of current flow contribute to the EC_a measurement: (1) a liquid phase pathway (Pathway 1) via salts contained in the soil water occupying the large pores; (2) a solid pathway (Pathway 2) via soil particles that are in direct and continuous contact with one another; and (3) a solid-liquid pathway (Pathway 3) primarily via exchangeable cations associated with clay minerals (Rhoades et al. 1999). To measure soil salinity, the electrical conductance of only the soil solution (Pathway 1) is required; consequently, EC_a measures more than soil salinity. In fact, EC_a is a measure of anything conductive within the volume of measurement and is influenced, whether directly or indirectly, by any edaphic properties that affect bulk soil conductance.

The EC_a is influenced by a complex interaction of edaphic properties including salinity, texture (or saturation percentage, SP), water content, bulk density (ρ_b), organic matter (OM), cation exchange capacity (CEC), clay mineralogy, and temperature. Thus, measurements of EC_a *must* be interpreted with these influencing factors in mind (Corwin and Lesch 2003, 2005a). An

understanding and interpretation of geospatial EC_a data can only be obtained from ground-truth measures of soil properties that correlate with EC_a from either a direct influence or indirect association. For this reason, geospatial EC_a measurements are used as a surrogate of soil spatial variability to direct soil sampling when mapping soil salinity at field scales and larger spatial extents and are not generally used as a direct measure of soil salinity, particularly at $EC_a < 2 \text{ dS m}^{-1}$ where the influence of conductive soil properties other than salinity can have an increased influence on EC_a .

Pathways of Electrical Conductance

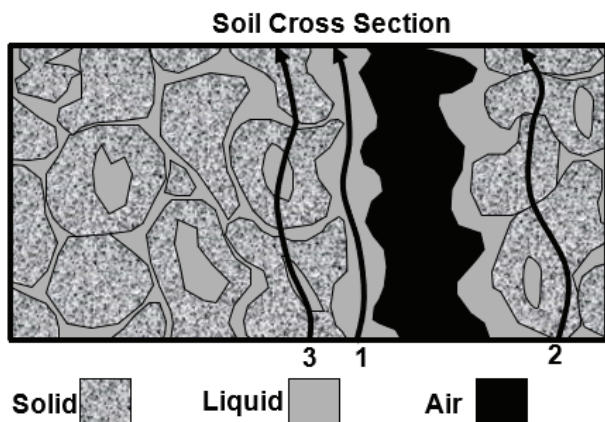


Figure 7.4 Schematic showing the three conductance pathways of apparent soil electrical conductivity. Pathway 1 = liquid phase conductance, Pathway 2 = solid phase conductance, and Pathway 3 = solid-liquid phase conductance. Modified from Rhoades et al. (1989).

Methods of field-scale soil salinity measurement

Soil salinity is a dynamic soil property that is highly spatially and temporally variable, which makes mapping and monitoring difficult. However, mapping and monitoring soil salinity at field scale are essential for management. The use of soil samples to measure salinity (e.g., EC_e , $EC_{1:1}$, $EC_{1:2}$, $EC_{1:5}$, or EC_p) at field scales is impractical because of the large number of samples required. This approach should only be used when sampling is directed to minimise the number of samples that reflect the range and spatial variability of salinity within a study area. This can be

achieved using easily measured spatial information that is correlated to soil salinity (e.g., crop yield observations or EC_a measurements using mobile ER/EMI equipment) as a means of directing where to take the fewest samples.

Visual crop observation

Visual crop observation is a quick method, but it has the disadvantage that salinity is only detected after crop damage has occurred; consequently, crop yield must be sacrificed to locate areas of salinity. Furthermore, decreases in crop yield are not necessarily the consequence of only salt accumulation. Crops respond to a variety of anthropogenic (e.g., irrigation uniformity, farm equipment traffic), edaphic (e.g., salinity, water content, texture, organic matter), biological (e.g., disease, nematodes), meteorological (e.g., precipitation, humidity, temperature), and topographical (e.g., slope, elevation, microrelief) factors, any of which can cause yield reduction.

In addition to visual observation, remote imagery is increasingly becoming a part of agriculture and potentially represents a quantitative approach to measure yield reduction associated with salinity when combined with proximal sensors, such as ER and EMI, to ground truth soil salinity (see Corwin and Scudiero 2019). Remote imagery may offer a potential for early detection of the onset of salinity damage to plants. The expectations for the use of multi- and hyper-spectral imagery to map and monitor soil salinity as well as the spatial variability of other soil properties (e.g., water content, mineralogy, and others) are high and will no doubt prove fruitful as research continues in this area (Corwin and Scudiero 2019).

Geospatial EC_a measurements

Geospatial EC_a measurements can be used to develop a soil sampling plan that identifies sites reflecting the range and variability of soil salinity and/or other soil properties correlated with EC_a (Corwin 2005). This approach has been demonstrated for not only mapping salinity at field scale (Corwin et al. 2003a; Corwin and Lesch 2005c), but for applications in (1) precision agriculture to define site-specific management units (Corwin et al. 2003b; Corwin 2005), (2) monitoring management-induced spatio-temporal changes due to degraded water reuse (Corwin et al. 2006), (3) characterising soil spatial variability (Corwin 2005), and (4) modelling nonpoint source pollutants in the vadose zone (Corwin et al. 1999; Corwin 2005).

Scientists at the U.S. Salinity Laboratory have developed an integrated system for the measurement of field-scale salinity consisting of (1) mobile EC_a measurement equipment (Rhoades 1993), (2) protocols for EC_a -

directed soil sampling (Corwin and Lesch 2005b, 2013), and (3) sample design software (Lesch et al. 2000). The integrated system for mapping soil salinity is schematically illustrated in Figure 7.5. Corwin and Scudiero (2020) provided a review of the integrated system and of the EC_a -directed soil sampling methodology.

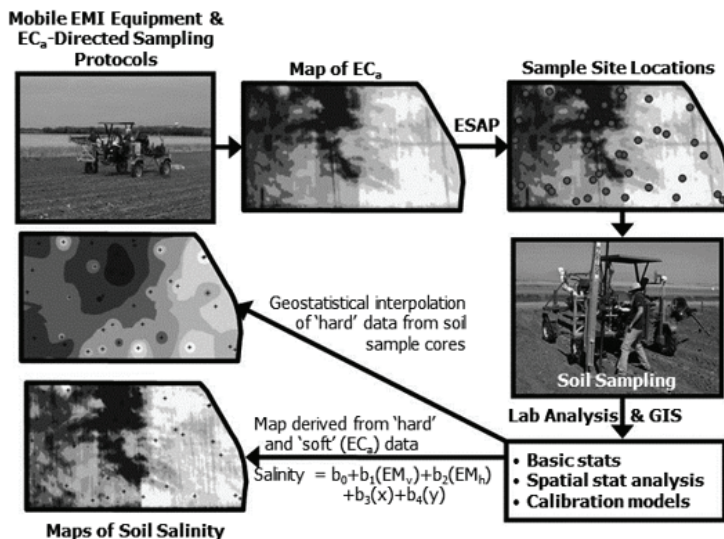


Figure 7.5 Schematic of the integrated system to assess field-scale soil salinity using apparent electrical conductivity (EC_a) directed soil sampling protocols, a mobile electromagnetic induction (EMI) rig, ESAP software, and geographic information system (GIS). EM_v refers to the measurement of EC_a by electromagnetic induction (EMI) in the vertical coil configuration and EM_h refers to measurement of EC_a in the horizontal coil configuration. Taken from Corwin (2015).

In instances where EC_a correlates with a particular soil property, an EC_a -directed soil sampling approach will establish the spatial distribution of that property with an optimum number of site locations to characterise the variability and minimise labour costs (Corwin et al. 2003b). Also, if EC_a is correlated with crop yield, then an EC_a -directed soil sampling approach can be used to identify those soil properties that are causing variability in crop yield (Corwin et al. 2003a). The purpose of an EC_a survey from a soil quality perspective is to establish the within field variation of soil properties that

influence the field's intended use (e.g., agricultural productivity, environmental protection, wastewater recycling, etc.). The purpose of an EC_a survey from a site-specific crop management perspective is to establish the within-field variation of soil properties influencing the variation in crop yield (Corwin et al. 2003a; Corwin and Lesch 2010). Corwin and Lesch (2003, 2005b, 2013) developed protocols and guidelines for characterising soil spatial variability from EC_a -directed soil sampling, with emphasis given to mapping soil salinity.

Plant salt tolerance

Scientists have quantitatively described the relationship between salinity and yield through the piece-wise linear plant salt tolerance equation (Maas and Hoffman 1977):

$$Y_r = 100 - b (EC_e - a) \quad [3]$$

where Y_r represents the relative crop yield, a is the salinity threshold ($dS\ m^{-1}$) and b is the slope expressed % per $dS\ m^{-1}$. Beyond a specific soil salinity threshold, which is characteristic to each plant and variety of plant, crop yield will decrease linearly, where the yield decrement is also plant specific. A compilation of plant salt tolerance work with the salinity thresholds and yield decrement slope for various crops and plants has been presented by Maas and Hoffman (1977), Maas (1990), and Grieve et al. (2012).

In traditional plant salt tolerance studies, the influences on plant yield of all other soil properties aside from salinity (e.g., matric stress, soil permeability, infiltration, pH) are removed by making them optimal for the soil used. Thus, the salinity threshold and yield decrement slopes derived are highly optimised and often do not reflect what actually occurs in the field (Corwin et al. 2003b, 2017), making their relevance questionable. Even so, traditional plant salt tolerance studies are the most widely used to determine the impact of salinity on crop yield and the relative salinity tolerance of different crops/varieties.

Recently, an alternative to traditional plant salt tolerance studies has been presented by Corwin et al. (2017), which uses EC_a -directed soil sampling and boundary line analysis to develop a salt tolerance curve from which a salinity threshold and yield decrement slope are determined. The extent to which these salinity thresholds and yield decrement slopes can be generalised is unknown pending future research, but it is reasonable to assume the parameters are scale dependent and valid within a localised area of similar soil type. The advantage of this approach is that the salt tolerance

parameters are tailored for the site of interest and are thereby more relevant and established under real-world conditions. Furthermore, the time, cost, and labour are less compared to traditional salt tolerance experiments.

Irrigation for soil salinity control

In arid or semi-arid irrigated agricultural areas, sustaining crop production requires irrigation management practices that prevent excessive root-zone salt accumulation. The primary source of salts is a consequence of the process of evapotranspiration. Salts in the irrigation water accumulate in the root zone due to evaporation at the soil surface and removal of pure water by the plant roots, which leaves salts behind. To prevent the accumulation of salts in the root zone, additional irrigation water beyond the consumptive water use by the crop is applied, which leaches the excess salts. In areas where the water table is deep the additional irrigation water simply percolates below the root zone carrying the salts with it, but in areas where the water table is shallow, the ground water is degraded by the salts leached from the root zone and the water table becomes more and more shallow, unless it can be channelled away using tile drains. Without tile drains and a place to dispose of the drainage water, the situation becomes problematic since salts can readily move back into the root zone as soon as the downward flux of water stops. Adhesive and cohesive forces result in the capillary rise of water from the shallow water table carrying salts in the ground water back into the root zone. The finer textured the soil, the greater the capillary rise will be, which can ultimately result in the substantial accumulation of salts at the soil surface in high clay soils. If this occurs, then crop yields will decline. In humid regions, rainfall normally results in enough leaching to flush the salt from the root zone. In subhumid and drier regions, irrigation water that exceeds the crop's water requirements may be needed to assure adequate leaching. Depending on the salinity control needed, leaching may occur continuously or at intervals of a few weeks to a few years.

The crop's water requirement and salinity control must be the prime considerations in places where salinity poses a hazard. Proper irrigation restores the soil's water deficit without a wasteful and potentially harmful excess. Over time, leaching must remove enough salt to prevent it from accumulating in the root zone beyond the crop's salt tolerance level. Chemical reactions in the soil, including cation exchange, dissolution, and precipitation reactions, affect the amount of salt leached, which may be greater than, equal to, or less than the amount of salt added by irrigation water.

Water and salt balance

The amount of irrigation water needed to meet the crop's water requirement can be calculated from a water balance of the crop root zone. The major flows of water into the crop root zone are irrigation, rainfall, and upward flow from groundwater, where the depths of each are expressed as the variables I_i , I_r , and I_g , respectively. The equivalent depths of water flow out due to evaporation, transpiration, and drainage are represented as O_e , O_t , and O_d , respectively. The difference between the water flowing in and the water flowing out must equal the change in storage, the depth of which is expressed as D_s . The water balance equation for storage change is as follows:

$$D_s = I_r + I_g + I_i - O_e - O_t - O_d \quad [4]$$

The equation for the change in salt storage, S_s , is expressed in Eq. [5]:

$$S_s = I_r C_r + I_g C_g + I_i C_i + S_m + S_f - O_d C_d - S_p - S_e \quad [5]$$

where C is the salt concentration; and the subscripts r , i , g , and d correspond to rain, irrigation, upward flow from groundwater, and drainage, respectively; S_m is the salt dissolved from minerals in soil; S_f is the salt added to soil as a fertiliser or amendment; S_p is the salt precipitated; and S_e is the salt removed in the harvested crop.

If $I_r + I_g + I_i$ is less than $O_e + O_t$ in Eq. [4], then the crop water demand is met by extraction from soil storage and reduced drainage in the root zone. As D_s is depleted, the soil dries, which reduces O_e , and O_t and the crop becomes water stressed. Initially, these processes bring water loss from the root zone in balance with the water supply at zero drainage. However, without drainage, salt stored in the root zone concentrates in the remaining stored water, which increases the osmotic stress on the plant, further reducing transpiration. If salts continue to increase in concentration, osmotic stress will reduce plant growth and may result in the plant dying. If the plant is salt-tolerant, transpiration may be reduced to the extent an irrigation results in excess water again being present in the profile and drainage occurs. This drainage carries salts out of the root zone. The resulting leaching fraction (LF) (i.e., the volume of drainage water divided by the volume of irrigation and precipitation) is the absolute minimum at which the crop can extract water from a saline root zone. This LF, however, is far less than that needed to prevent a reduced yield.

When a shallow water table exists, deficiencies in $I_i + I_r$ may be offset by I_g . When flow is upward from the groundwater, drainage is zero and salt will not be exported from the entire root zone. This situation cannot continue indefinitely. In the field, upward flow and drainage may take place alternately during the year. Typically, in arid and semi-arid regions drainage takes place when the water requirements of the crop are low, and rainfall or irrigation water applications are high. Upward flow takes place when water requirements are high, and rainfall and applications of irrigation water are insufficient to meet crop demand. If upward flow continues and sufficient leaching does not take place, soil salinity will ultimately reduce the crop's water consumption to the point that the crop dies. Temporary use of water from soil storage beyond that normally removed between irrigations or from shallow groundwater is a useful strategy for managing water. However, over the long-term and where salinity is a hazard, a net downward flow of water through the root zone is needed to sustain crop productivity. One management strategy in shallow groundwater areas is to utilise pre-plant irrigation to leach the root zone prior to planting (Ayars 2003).

Rarely will conditions controlling the water that flows into and out of the root zone last long enough for a true steady state to exist. As a result, the amount of salt stored in the root zone continually fluctuates. The goal of water management is to maintain this fluctuation within limits that neither allow excess drainage nor reduce crop growth.

Leaching requirement

The leaching requirement (LR) is based on the concept of LF, which is defined as the fraction of infiltrating water that moves beyond the root zone and is a measure of the level of leaching of salts. As the LF increases, the concentration of salts in the root zone decreases. Quantitatively, LF is defined by Eq. [6]:

$$LF = \frac{O_d}{I_a} = \frac{C_a}{C_d} = \frac{EC_a}{EC_d} \quad [6]$$

where O_d (mm) and I_a (mm) are the depths of drainage water and infiltrating applied water, respectively; C_a (mg/L) and C_d (mg/L) are the salt contents of the applied and drainage water, respectively; and EC_a (dS m⁻¹) and EC_d (dS m⁻¹) are the electrical conductivities of the applied and drainage water, respectively. The LR represents the minimum LF that will adequately leach salts in the root zone to a level that does not measurably reduce crop yield;

consequently, the root zone salinity level is the maximum permissible salinity level of EC_d (i.e., EC_d^*) that will still result in optimum plant growth. Quantitatively, the original LR model as defined by the U.S. Salinity Laboratory Staff (1954), which assumes steady-state conditions, is represented by Eq. [7]:

$$LR = \frac{C_a}{C_d^*} = \frac{EC_a}{EC_d^*} \quad [7]$$

where C_d^* is the maximum permissible salt content of the drainage water. Equation [7] must still include a relationship between plant response and the EC of the bottom of the root zone to be useful in determining the required leaching level.

It has generally been assumed that the plant responds to the linearly averaged root zone EC of the saturation extract (EC_e), (Shalhevet and Bernstein 1968; Shalhevet et al. 1969), which is an assumption derived from early salt tolerance experiments that were conducted at very high LF's resulting in fairly uniform salt concentrations throughout the root zone. Rhoades (1974) introduced an estimate of EC_d^* with Eq. [8]:

$$EC_d^* = 5EC_e^* - EC_i \quad [8]$$

where EC_e^* ($dS\ m^{-1}$) is the linearly averaged root zone EC of the saturation extract for a given crop appropriate to the tolerable degree of yield depression, usually 10% or less, and is equivalent to the plant salt tolerance threshold EC values as defined by Maas (1990) and Maas and Hoffman (1977). EC_i is the EC of the irrigation water ($dS\ m^{-1}$). Substitution of Eq. [8] into Eq. [7] yields Eq. [9], which ties LR to irrigation water salinity and crop salt tolerance and is referred to as the Rhoades LR model (also referred to as the Traditional LR model):

$$LR = \frac{EC_i}{5EC_e^* - EC_i} \quad [9]$$

Hoffman and van Genuchten (1983) developed a steady-state model that determined the linearly averaged mean root zone salinity by solving the continuity equation for the one-dimensional vertical flow of water through

the soil assuming an exponential plant water uptake function. The linearly averaged salt concentration of the root zone ($\bar{C}$) as a ratio of the salt concentration of the irrigation of water (C_i) is:

$$\frac{\bar{C}}{C_i} = \frac{1}{LF} + \frac{\delta}{Z \cdot LF} \ln[LF + (1 - LF)e^{-Z/\delta}] \quad [10]$$

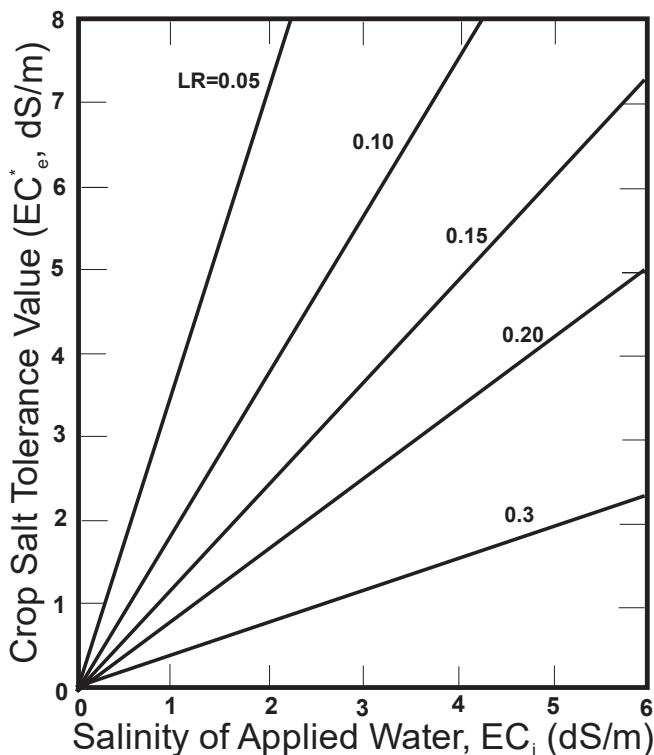


Figure 7.6 Leaching requirement (LR) as a function of the salinity of the applied irrigation water (EC_i) and plant salt tolerance threshold (EC^*_e). Modified from Hoffman and van Genuchten (1983) with permission.

LF is the leaching fraction; Z is the depth of the root zone, and δ is an empirical constant set to $0.2 Z$. Figure 7.6 shows the LR as a function of the salinity of the applied irrigation water and salt tolerance based on the

Hoffman-van Genuchten model. Other steady-state models of LR have been developed by Ayers and Westcot (1976) and Hanson et al. (1993). Hoffman (1985) compared calculated leaching requirements from the Rhoades, Hoffman-van Genuchten, and two other steady-state models. Of the four models tested, the Hoffman-van Genuchten model agreed well with the measured values throughout the range of LR of agricultural interest.

These models can be used not only to determine the LR but also to determine the maximum irrigation water salinity that can be applied to a crop for a specific LF. Table 7.1 provides a comparison of the Rhoades and Hoffman-van Genuchten LR models for the estimated maximum irrigation water salinity (i.e., EC_i) that could be used to grow tomatoes with an EC^* of 2.5 dS m^{-1} and LF's of 0.05, 0.10, 0.15, and 0.20. At all LF's, the Hoffman-van Genuchten model indicated that higher levels of salinity could be used for irrigation without loss compared to the Rhoades model. The Hoffman-van Genuchten model is in closer agreement with transient models than other steady-state models, which are too conservative in the quality of irrigation water that can be used without reducing yields.

The aforementioned LR models, including the Rhoades and Hoffman-van Genuchten models, only consider the salt tolerance of the crop grown and the salinity of the irrigation water while assuming steady-state conditions. However, steady-state conditions do not exist in most field situations due to the influence of rainfall, crop rotations, alterations to irrigation management, variation in irrigation water quality, and variations in soil profile water content and salinity resulting from variations in plant root water uptake.

Table 7.1 Estimated maximum irrigation water salinity that could be used to grow tomatoes with a salt tolerance threshold of 2.5 dS m^{-1} for various leaching requirements (LRs) as calculated from the Rhoades and Hoffman-van Genuchten LR models.

Leaching Fraction (LF)	Maximum Irrigation Water Salinity (dS m^{-1})	
	Rhoades LR Model (Eq. [9])	Hoffman-van Genuchten LR Model (Eq. [10])
0.05	0.6	0.7
0.10	1.1	1.3
0.15	1.6	2.0
0.20	2.1	3.0

In addition, the LR is influenced by numerous factors including the salinity of applied water, crop salt tolerance, precipitation-dissolution reactions, transient root water uptake distributions, preferential flow, climate, runoff, extraction of shallow groundwater, and leaching from effective precipitation, in addition to the questionable appropriateness of the assumption of steady-state conditions. Based on the exclusion of these factors from consideration, publications by Corwin et al. (2007) and Letey and Feng (2007) have shown that the steady-state LR models are conservative, suggesting that a new paradigm may be needed, particularly for research applications. Corwin et al. (2007) provided a detailed discussion of the appropriateness of transient LR models over steady-state LR models and demonstrated that transient models provide less conservative estimates (i.e., LR estimates are lower). Based on the work of Corwin et al. (2007) and Letey and Feng (2007), papers by Letey et al. (2011) and Corwin and Grattan (2018) suggested that new guidelines for LRs are needed that are developed from transient LR models, as these provide more accurate estimates of LR. The new guidelines are especially important for water-scarce agricultural areas.

Accounting for non-uniformity of irrigations to estimate LR has not been addressed to date. If the LR is not met everywhere in the field, salinity will increase wherever ET plus the LR is not met. Whether to apply enough water to assure that the LR is met throughout the field, or to accept some reduction in yield in parts of the field rather than over-irrigate most of the field, must be determined.

Adopting advanced irrigation technologies and implementing advanced management alternatives is needed to approach the goal of achieving LRs. Inefficient irrigation inadvertently results in excessive leaching that is costly and leads to a loss of water, energy, and nutrients, deteriorates the quality of water, and increases the need for drainage facilities. Consequently, knowing the LR of crops and striving to attain them is vital.

Steady-state vs transient leaching requirement models

The questionable ability of Eq. [7] to accurately calculate the LR stems from (1) the assumption of steady state, and (2) influencing factors that are not considered. Furthermore, osmotic and matric effects on roots will cause plants to take up water from that area of the root zone where the least energy is expended to extract water. The dynamic uptake of water by roots allows the plant to tolerate average root zone salinities higher than the plant's salt-tolerance values, which are experimentally derived from the linearly averaged root zone EC_e of high-LF experiments that produce a nearly

uniform root zone salinity. By accounting for the transient uptake of water by roots, the LR will be lower than that calculated by Eq. [7].

Other factors that influence the LR are also often not taken into account, including:

- Precipitation and dissolution reactions, which can change soil water salinity and either increase (dissolution) or decrease (precipitation) the LR.
- Climatic factors such as humidity, which can increase a plant's salt tolerance and lower the LR.
- Preferential flow, which influences the efficiency of salt leaching, resulting in an increase in LR.
- Runoff, which reduces the volume of infiltrating water and reduces the leaching of salts, thus raising the LR.
- Plant groundwater extraction, which can reduce the impact of salts accumulating in the root zone, thereby lowering the LR.
- Leaching from effective precipitation, which will lower the LR.

Furthermore, the estimation of LR does not include (1) the manner in which spatiotemporal variation in salinity within the root zone affects crop response and water uptake; (2) scale issues; (3) horizontal leaching and the subsequent redistribution of salts for cracking soils when flood irrigation is used; (4) basing the LR on the most salt-sensitive crop in a crop rotation; and (5) uncertainties in salt-tolerance data developed from experimental plots when applied to field situations. Some of these have been discussed in Rhoades (1999).

It is also noteworthy that the LR does not provide sufficient information concerning optimal irrigation because optimal irrigation is the amount of water that maximises profit, and maximum profit may not always coincide with maximum yield (Letey et al. 1985). The relationship between crop yield and the seasonal amount of water required is essential to determine the optimal irrigation management (Letey et al. 1985). For this reason, crop water production functions have been advocated as a means of determining the economically optimal amount of water that is needed to prevent an excessive accumulation of salts. Nevertheless, LR is still widely used by growers and irrigation management districts in the southwestern USA and many other irrigated arid and semi-arid regions of the world.

Transient models enable the simulation of complex processes with time-dependent variables. The development of transient models has primarily been facilitated by the development of high-speed computers. To evaluate the appropriateness of a steady-state approach for estimating the

LR, Corwin et al. (2007) compared a variety of steady-state and transient LR models to determine whether differences existed, the size of the differences, and the reasons for the differences, and to analyse the implications of the differences with respect to irrigation management and salinity control. Corwin et al. (2007) found that transient models estimated the LR to be lower than the traditional steady-state approach. The weakness of the traditional steady-state LR approach is that steady-state conditions seldom exist except over long time periods, and processes such as preferential flow and precipitation-dissolution reactions are not considered. The difference between the traditional steady-state and transient approaches is expected and adds credence to the recommendation that any estimation of LR should first consider the use of a transient model, particularly for research applications. The same general conclusion recommending the use of a transient over a steady-state approach for estimating LR was also found by Letey and Feng (2007) when focusing on the influence of plant water uptake using the transient *ENVIRO-GRO* model compared to two steady-state models. Letey et al. (2011) and Corwin and Grattan (2018) evaluated current soil salinity leaching requirement guidelines based on steady-state and transient models. Their analysis indicated that steady-state models overestimate LR and exaggerate the negative consequences of irrigating with saline waters. The overestimation results in an excessive application of irrigation water and increased salt loads to drainage systems and underlying aquifers.

Transient models provide improved estimates of LR but are not without their own problems. In comparison to steady-state models, current transient models used to estimate LR are (1) more complex to use and subsequently less user friendly; (2) require input parameters that are less readily available and more difficult to obtain; and (3) do not calculate LR directly so they must be reprogrammed. These problems should not preclude the use of transient models in place of steady-state models as long as transient models are not misused due to the lack of understanding and inability to accurately measure input parameters.

Site-specific irrigation management to control soil salinity

The estimation of a single LR for a field, even when using a transient model to estimate LR, is dubious because of the tremendous spatial variability in soil properties that influence the LR. To control field-scale soil salinity in the most water efficient manner, site-specific irrigation management units must be created to address the issue of spatial variability. Site-specific irrigation management units (SSIMUs) are spatial domains where the

applied irrigation water plus precipitation result in the same LF for the entire spatial domain.

Corwin and Lesch (2010) provided a thorough explanation of the process involved in developing SSIMUs and Corwin et al. (2003b) conducted the original research that led to SSIMUs. Corwin et al. (2003b) used EC_a -directed soil sampling to develop a crop yield response model for cotton that related cotton seed yield to edaphic properties, including salinity (i.e., EC_e), LF, pH, and water content. From the crop yield response model, SSIMUs were developed to control salinity and to reduce excessive leaching for water conservation (Corwin 2005; Corwin and Lesch 2010). The delineation of SSIMUs for the control of soil salinity by Corwin and colleagues (Corwin et al. 2003b; Corwin 2005; Corwin and Lesch 2010) requires that statistical correlations exist between EC_a and crop yield and between EC_a and salinity. The correlation between EC_a and crop yield indicates that EC_a is measuring a soil property or combination of properties that are influencing yield. The correlation between EC_a and salinity indicates that geospatial measurements of EC_a will spatially map salinity. If EC_a does not correlate with crop yield, then the edaphic properties measured by EC_a are not influencing the crop yield. However, other factors are influencing crop yield, such as biological, meteorological, topographical, and anthropogenic factors, and other edaphic properties not measured by EC_a . If EC_a is not correlated with salinity, then geospatial EC_a measurements cannot accurately map salinity. The LR is estimated for each SSIMU using any of the previously mentioned steady-state or transient models.

Conclusions

The control of soil salinity at field scale by leaching requires (1) geospatial knowledge of the LR to establish the amount of irrigation water to apply; and (2) geospatial distribution of soil salinity in the root zone to establish where to apply the irrigation water in the amount indicated by the LR. The application of EC_a -directed soil sampling to delineate SSIMUs is a means of establishing geospatial LR estimates using steady-state or transient models to control soil salinity by leaching. In instances where precipitation-dissolution reactions are occurring, the steady-state model WATSUIT (Rhoades and Merrill 1976) is the simplest model to establish the LRs. The transient model UNSATCHEM (Šimůnek and Suarez 1994; Šimůnek et al. 1996) will provide greater accuracy but requires considerably more investment in time to operate because of its complexity. In those cases where precipitation-dissolution reactions are not a concern, the transient model TETrans (Corwin et al. 1990; Corwin and Waggoner 1990a, 1990b)

is the simplest to establish the LRs. The geospatial approach to controlling soil salinity by leaching is considerably more rigorous and complex than the traditional approach that disregards soil spatial variability, the transient nature of the leaching process, and the occurrence of precipitation-dissolution reactions, but is substantially more accurate.

An aspect of soil salinity control that is not a focus of this chapter is the application of soil amendments to enhance the leaching of salts. These soil amendments include calcium chloride, gypsum, lime, sulfur, and organic matter. These soil amendments are used to improve the tilth and permeability of soils (e.g., application of calcium chloride, gypsum, and lime to sodic soils), to adjust the soil pH (e.g., sulfur applied to high pH soils), and to aggregate soils to improve soil structure (e.g., application of organic matter to soils with little structure) for the purpose of improving water flow to leach salts.

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Abbreviations

- ρ_b : bulk density
- CEC: Cation exchange capacity
- EC_a: Apparent soil electrical conductivity (dS m⁻¹)
- EC_{1:1}: Electrical conductivity of a 1 to 1 soil to water (gravimetric basis) saturation extract (dS m⁻¹)

EC _{1:2} :	Electrical conductivity of a 1 to 2 soil to water (gravimetric basis) saturation extract (dS m ⁻¹)
EC _{1:5} :	Electrical conductivity of a 1 to 5 soil to water (gravimetric basis) saturation extract (dS m ⁻¹)
EC _e :	Electrical conductivity of the saturation extract (dS m ⁻¹)
EC _p :	Electrical conductivity of the saturation paste (dS m ⁻¹)
EM _h :	Electromagnetic induction measurement in the horizontal coil configuration
EMI:	Electromagnetic induction
EM _v :	Electromagnetic induction measurement in the vertical coil configuration
ER:	Electrical resistivity
GPS:	Global positioning system
OM:	Organic matter
SP:	Saturation percentage
TDR:	Time domain reflectometry
Θ:	Water content

CHAPTER 8

ACIDITY

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Abstract

Soil acidity limits crop growth and reduces crop yields and agricultural product quality in many parts of the world. This chapter discusses the causes of soil acidification in agroecosystems, the distribution of acid soils worldwide, the adverse effects of soil acidity on crop production, and management options to remediate soil acidity and improve crop production. It also highlights areas requiring further research. The chapter provides useful references for sustainable utilisation of acid soil resources for crop production.

Keywords: Al toxicity; amelioration; crop production; lime; organic amendment; soil acidification.

The causes of soil acidity

Natural acidification

Soil acidification is a process accompanying soil formation and development (Krug and Frink 1983). When rainfall is greater than evaporation, leaching occurs, which leads to loss of soluble salts and alkaline substances from soil profiles. The protons in rainwater can also directly react with soil alkaline substances and consume them. These

processes lead to soil acidification.

Initially, the dissolution of soil carbonates is the main reaction to consume hydrogen (H^+) ions, until the pH drops to around 6.2 (Ulrich 1986). Below pH 6.2, the weathering of soil minerals and cation exchange reactions between soil exchangeable base cations (Ca^{2+} , Mg^{2+} , K^+ , and Na^+) and H^+ in soil solution become the main reactions to consume H^+ ions. The exchange reactions of soil base cations with H^+ , combined with heavy rainfall, enhance leaching losses of base cations, leaving H^+ ions to occupy soil exchange sites. The exchangeable H^+ on the surface of soil colloids is highly active and dissolves aluminosilicates to form exchangeable Al^{3+} . Therefore, soil exchangeable acidity exists mainly as exchangeable Al^{3+} in acid soils.

The protons causing natural soil acidification are mainly derived from the dissociation of H_2CO_3 and organic acids (Vanbreemen et al. 1984). However, the production of protons through these pathways is limited and thus the rate of natural soil acidification is very slow.

Acid deposition

Acid deposition occurs via acid rain and dry deposition. Acids form from SO_2 and NO_x in the atmosphere, released due to air pollution. Coal combustion produces large amounts of SO_2 and some NO_x . Vehicle exhaust also contains NO_x and pollutes the air. Acid rain brings not only protons, but also large amounts of SO_4^{2-} and NO_3^- into soils. These anions are highly mobile in soil, and they enhance the movement of base cations from soil to surface water, and thus accelerate soil acidification.

At the end of last century, acid deposition was reduced significantly and its negative effect on ecological systems became less in west Europe and North America (Reis et al. 2012). However, acid deposition in developing countries, including China, is still serious and more attention should be paid to its effect on soil acidification, agricultural production and ecological system stability in these countries (Yu et al. 2016).

Ammonium-based fertilisers

Excessive application of NH_4^+ -based fertilisers is the main cause of soil acidification in croplands. This has been verified by long-term field experiments around the world. For example, in Alberta, Canada, after 27 years of fertilisation with NH_4NO_3 at $168 \text{ kg N ha}^{-1} \text{ year}^{-1}$, soil pH was reduced to 4.10 in the 0-5 cm layer, while soil pH was 6.85 where no N was applied (Malhi et al. 1998). Similar decreases in soil pH induced by N

fertilisers were also observed in croplands in the United States, Australia and China (Cai et al. 2015; Schroder et al. 2011; Xu et al. 2002). Two nationwide soil surveys indicated that soil pH significantly decreased from the 1980s to the 2000s in the major cropping regions in China (Guo et al. 2010).

Nitrification of ammonium in soils is responsible for soil acidification induced by ammonium-based fertilisers. Its simplified reaction is



If nitrate from nitrification is fully taken up by crops, then half of the protons produced will be neutralised by crop release of OH^- during nitrate uptake. However, when the nitrate is leached out of the rooting zone, all the protons from nitrification contributed to soil acidification. Therefore, nitrification of ammonium fertilisers combined with the leaching loss of nitrate accelerates soil acidification (Ridley et al. 1999).

Crop nutrient uptake

Crop harvest and removal of crop straw and grain contributes to soil acidification. When plants take up more nutrients with one type of charge than the other, the plants extrude H^+ or OH^- to maintain electro-neutrality, leading to acidification or alkalinisation of the rhizosphere soil (Bolan et al. 1991; Hinsinger et al. 2003). As most plants generally take up more cations than anions (Curtin and Wen 2004; Noble and Randall 1999; Noble et al. 1996), they release net protons into the rhizosphere soil through their roots, while accumulating organic anions (alkalis) in their tissues (Yan et al. 1996). In natural ecosystems, when the plant dies and the residue decomposes, this accumulated alkalinity is returned into the soil. However, crop harvest and product removal change the acid-base balance in the plant-soil system, leading to soil acidification and the depletion of calcium (Ca^{2+}), magnesium (Mg^{2+}), and potassium (K^+) (Tang and Rengel 2003). The return of crop residues to the soil after crop harvest thus slows down soil acidification (Lesturgez et al. 2006; Yan et al. 1996).

Legumes generally absorb more cations from soils than non-legumes, leading to accelerated soil acidification when plant residues are removed offsite (Bolan et al. 1991; Yan et al. 1996). Extensive growth of legume pastures in Australia and New Zealand had resulted in considerable soil acidification in the last century (Bolan et al. 1991; Noble et al. 1997). When phosphorus (P) is deficient, N_2 -fixing legumes absorb more cations and extrude more protons (Sas et al. 2001), which may accelerate the acidification of P-deficient soils (Shen et al. 2004). Soil acidification

results in the production of exchangeable Al^{3+} (Figure 8.1) and thus Al toxicity to crops.



Figure 8.1 Processes of exchangeable Al^{3+} production during soil acidification (modified from Xu et al. 2018)

The distribution of acid soils worldwide

Globally, about 3.95 billion hectares of land have topsoil acidity ($\text{pH} < 5.5$), which is about 30% of the ice-free land area in the world (Sumner and Noble 2003). Such acid soils occupy about 6.1% of Australia and New Zealand, 9.9% of Europe, 16.7% of Africa, 26.4% of Asia, and 40.9% of the Americas. They cover a significant part of more than 48 developing countries, mainly in the tropical regions. Acid soils exist more frequently in Oxisols in Africa, and Oxisols and Ultisols in South America (Fageria and Nascente 2014).

Generally, climatic conditions are the main factor controlling the distribution of acid soils. Most acid soils are distributed in the tropical, subtropical, and temperate regions with humid climates. High temperature and high precipitation in the tropical and subtropical regions induce intensive mineral weathering and mass leaching of base cations (Fageria and Nascente 2014). Coniferous forests also occur widely in temperate regions. Organic compounds from incompletely decomposed tree debris not only promote mineral dissolution and leaching of base cations, but provide protons to acidify soils (Lundström et al. 2003). Acid sulphate soils, which are located in swamps, coastal marshes, or polder lands, can also generate large quantities of acidity if they are exposed to oxygen since the iron sulfide minerals they contain oxidise to produce sulfuric acid. Acid precipitation and fertiliser application have expanded the scale and distribution of acid soils to a great extent (FAO 2015). The most acidic topsoils ($\text{pH} < 3.5$) in the world are mainly found in South America due to intensive agriculture and deforestation, and some are distributed in river deltas covered with mangroves, such as the Amazon and Orinoco Deltas.

Effect of soil acidity on crop production

Al toxicity

Aluminium adsorption on root cation exchange sites is one of the mechanisms responsible for Al toxicity (Rufyikiri et al. 2003). The adsorbed Al leads to the rupture of root cells through decreased cell-wall relaxation, and Al then decreases root growth through its adverse effects on the plasma membrane, cytoplasm, and nucleus at a subcellular level (Kopittke et al. 2016). Negatively-charged functional groups on crop roots have been found to be responsible for Al adsorption (Liu and Xu 2015). The roots of Al-resistant crops carry fewer negative charges and functional groups than Al-sensitive crops and thus adsorb less exchangeable and complexed Al on the roots compared with Al-sensitive crops (Ahn et al. 2004; Liu et al. 2016; Lu et al. 2020). This is responsible for the greater yield of Al-resistant than Al-sensitive cultivars grown in a soil with subsurface acidity (Tang et al. 2003a). In addition, crops can also cope with the toxic effects of Al through exclusion and internal tolerance mechanisms (Kochian et al. 2015).

The extent of Al toxicity depends on the Al species in the soil solution. Dissolved Al mainly exists as Al^{3+} , hydroxyl-Al, and complexes with organic ligands, fluoride, and sulfate (Driscoll 1984). Al^{3+} and monomeric hydroxyl Al (Al-OH) are generally considered the most toxic species, while organic-Al, Al-F, and Al- SO_4 complexes are less, or not, phytotoxic (Kinraide 1997). It was found that the yields of sweet potato and its straw were correlated negatively with the concentration of Al^{3+} and Al-OH in soil solution, confirming that alleviating Al toxicity increased potato yield (Zhao et al. 2020a).

The harmful impacts of Al toxicity on crop growth and yield vary with crop species and their genotypes. For example, there was an overall 14% grain yield increase by growing Al-resistant wheat ET8 on an acid soil in Western Australia compared with Al-sensitive ES8 (Tang et al. 2003a). Great progress has been made on confirming Al-resistance genes. Resistance gene expression is higher in Al-resistant genotypes, and ALMT and MATE genes are linked with malate and citrate efflux induced by Al stress (Kochian et al. 2015; Ryan 2018).

Mn toxicity

Manganese is a micronutrient for plant growth (Marschner 1995). However, excess Mn supply can damage plants and reduce the quantity and quality of their products. Manganese toxicity to plants is the second most

important constraint to crop growth in acid soils following Al toxicity (Foy 1984; Marschner 1995). Manganese toxicity generally occurs under acidic conditions if soil total Mn contents are high, even when the pH is above 5.0 (Foy 1984).

Plant species differ in their responses to Mn(II) toxicity. In a solution culture, soybean showed decreased growth at 30 μM Mn(II), while white lupin, narrow-leaved lupin, and sunflower grew well even at 100 μM Mn(II) (Blamey et al. 2015). The responses of crops to Mn(II) toxicity also vary with their genotypes. The sensitive genotypes of soybean (such as Kahala, Sango, Verde, and Wilson 5) grown on an acid soil (pH 5.2) with a high content of Mn displayed the severe Mn(II)-toxicity symptoms of chlorosis and crinkle leaves, which decreased yield by 42-84% (Reddy et al. 1991).

Toxicity of Mn(II) to plants mainly results from its excess accumulation in leaves (Wissemeier et al. 1992), which leads to small dark spots, chlorosis, necrosis, distortion, and abscission in plant leaves (Heenan and Carter 1976; Wissemeier and Horst 1992; Fernando and Lynch 2015). In addition, adverse impacts on root growth also occur. Surface potential and negative charges on plant roots affect Mn(II) adsorption onto the roots and thus its toxicity to plant roots (Kopittke et al. 2011; Liu et al. 2019).

Soil infertility

Soil acidification increases the releases of base cations Ca^{2+} , Mg^{2+} , K^{+} , and Na^{+} from soil exchange sites into the soil solution due to cation exchange reactions with H^{+} and Al^{3+} ions. Rainfall then facilitates the leaching loss of these base cations from soils, which leads to deficiencies of Ca^{2+} , Mg^{2+} , and K^{+} . For example, it was found that the concentrations of available Ca was negatively correlated with exchangeable Al in North American forest soils, and that Al mobilisation was responsible for Ca deficiency (Lawrence et al. 1995).

In tropical and subtropical regions, acid soils normally have low cation exchange capacity and thus low retention capacity for base cations and ammonium (Yu 1997), which is one of the reasons for soil infertility. Acid soils in these regions also contain large amounts of Fe/Al oxides and hence have great adsorption capacity and strong adsorption affinity for phosphate, which leads to low P availability (Fageria and Nascente 2014). The precipitation of phosphate with Fe and Al further decreases P availability. This is the reason why P deficiency still occurs in some acid soils with high total P.

Soil bacteria are generally sensitive to soil acidity. Both the relative abundance and diversity of bacteria are correlated positively with soil pH

(Rousk et al. 2010) and the highest bacterial diversity is observed near neutral pH values (Tripathi et al. 2012). Decreasing soil pH decreases bacterial growth and activity and thus inhibits the decomposition of soil organic matter and the cycling of C, N, P, and S in acid soils (Fageria and Nascente 2014; Zhang et al. 2020), with flow on effects for crop growth and yield.

Crop yields

Overall, the combined impact of Al/Mn toxicity and nutrient deficiencies in acid soils can inhibit crop growth and reduce crop yield (Figure 8.2). For example, the results from a long-term fertilisation experiment showed that as the soil pH of an Ultisol decreased from 5.4 to 3.75 and 4.0, respectively, wheat yield decreased by 92 and 76 kg ha⁻¹ year⁻¹, while corn yield decreased by 167 and 94 kg ha⁻¹ year⁻¹, respectively (Zhang et al. 2009). Decreases in crop yield during 1995-2002 also coincided with large decreases in soil pH in a long-term experiment in Oklahoma, USA (Schroder et al. 2011).

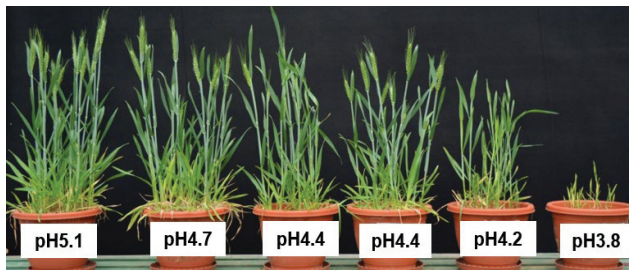


Figure 8.2 Effect of soil acidification on wheat growth

The critical soil pH is defined as the pH value below which lime application increases crop yield (Adams 1984). Below the critical pH, soil acidity shows a negative effect on crop growth and yield. The critical soil pH can vary with crop species and soil types. For example, the critical pH for cotton on the soils (Typic Paleudults) in the Interior Low Plateau Province, USA, is between 5.0 and 5.3. In comparison, the critical pH of the soils in the same Province is about 5.0 for corn (Adams 1984). The critical pH is about 5.5 for wheat grown on most soils in Alabama, USA (Adams 1984). In a field experiment, seed yield of winter canola decreased linearly with decreasing soil pH below 5.8 in an acid soil (Thermic Ultic Haplustalfs) in north-central Oklahoma, USA (Lofton et al. 2010).

The effect of soil acidity on crop yield also varies with crop genotype. Soil acidity has a greater negative effect on the growth and yield of acid-sensitive genotypes than acid-tolerant genotypes (Flores et al. 1988; Tang et al. 2001; 2003b). For example, Tang et al. (2003b) showed that 115 wheat genotypes differed substantially in their sensitivity to soil acidity with root length of 6 day old seedlings ranging from 66 mm to 350 mm when grown at soil pH 3.9. They also found that the yields of 12 genotypes were 52% to 89% when grown in an acid soil relative to the limed soil in the field.

Management options to remediate acidity and improve crop production

Amelioration of soil acidity

Liming materials

Liming is a traditional and effective practice for amelioration of soil acidity. Several liming materials are used as amendments, and their acid-neutralizing capacity follows the order of $\text{CaO} > \text{Ca(OH)}_2 > \text{CaMg(CO}_3)_2 > \text{CaCO}_3 > \text{CaSiO}_3$. The reaction formula below illustrates the reactions involved in acid soils:



where Al-X and Ca-X represent Al and Ca attachments, respectively, to a soil exchange site. The reaction binds the Al to hydroxyls, causing it to precipitate (Fageria and Nascente 2014).

The application of liming materials to acid soils has a number of favourable effects. The primary effect of liming is to eliminate the toxicity of H^+ , Al^{3+} , and Mn^{2+} , and to provide base cations of Ca^{2+} and Mg^{2+} . Liming also promotes the availability of Mo and P to crops through increasing solubility. Therefore, it can increase root activity and uptake of nutrients. Elevation in pH due to liming increases soil negative charges and thus retention capacity for nutrients (Bolan et al. 2003).

Biologically, liming can improve the growth environment for soil microorganisms, and enhance the diversity and activity of soil bacteria. An elevation in pH also enhances the activity of urease, catalase, acid phosphatase, alkaline phosphatases, polyphenol oxidase, and protease. Therefore, liming enhances mineralisation of organic matter and thus nutrient cycling in acid soils (Bolan et al. 2003; Fageria and Baligar 2008). Liming has often been effective to improve the structure of acid soils

through increasing soil pH and Ca concentration and thus aggregate stability. In addition, increased microbial activity and the subsequent production of extracellular polysaccharides in acid soils can act as a binding agent to increase aggregate stability (Chan and Heenan 1998; Bolan et al. 2003). Thus, liming improves the chemical, physical, and biological fertility of acid soils and thereby increases crop yields (Adams 1984).

Most plants grow well between pH 5.5-6.5 and liming normally aims to maintain soil pH in this range (Harter 2007). It should also be noted that the ameliorating effect is mainly confined to the topsoil, with lime less effective for treating subsoil acidity due to the low mobility of lime materials.

Gypsum and phosphogypsum

Gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) and phosphogypsum have also been widely used as acid-soil amendments (Shainberg et al. 1989). Because their acid-neutralizing capacities are normally less than 10% of CaCO_3 , their ameliorating effect on topsoil acidity is much less than that of CaCO_3 . However, they can improve conditions for plant growth by decreasing the Al/Ca ratio, releasing OH^- due to SO_4^{2-} adsorption; and decreasing Al^{3+} activity in the soil solution by increasing ionic strength, forming non-toxic Al-SO_4^{2+} complexes, and precipitating Al-OH-SO_4 minerals. Phosphogypsum also contains fluorine (F) and P, which are important ligands to complex with Al^{3+} and alleviate Al toxicity to plants (Shainberg et al. 1989; Fageria and Nascence 2014; Masud et al. 2015). In addition, because gypsum is moderately water-soluble, surface application of gypsum is proven to be the most effective method to ameliorate subsoil acidity.

Subsoil acidity is an important constraint in acid Oxisols and Ultisols, which cover an area of >800 million hectares and constitute more than 40% of potentially-arable tropical soils (Fageria and Nascence 2014). In addition, severe subsoil acidity also occurs in subtropical and temperate regions (Bolan et al. 2003). Dissolved gypsum can leach down the soil profile to increase subsoil Ca^{2+} and decrease toxic Al. Therefore, gypsum combined with lime has proven effective to ameliorate topsoil and subsoil acidity in highly weathered soils (de Oliveira and Pavan 1996).

Industrial by-products

Some industrial by-products that contain calcium and sulfate can be used as alternative amendments. Recent studies have shown that alkaline slag, an industrial by-product from the ammonia-alkali production of sodium carbonate, is effective in correcting soil acidity (Li et al. 2010; Pan et al. 2019). The main chemical composition of alkaline slag is CaCO_3 , CaSO_4 ,

and CaCl_2 , which can directly overcome soil acidity (Masud et al. 2014). Alkaline slag contains high amounts of soluble sulfate and chloride which are easily leached down the soil profile and promote the downward migration of base cations in the amendments to ameliorate subsoil acidity. Alkaline slag also contains significant amounts of Mg to enrich available Mg in acid soils. Compared with phosphogypsum, alkaline slag was more effective in correcting soil acidity, eliminating Al toxicity, and improving Ca and Mg availability in an Ultisol profile (Li et al. 2015a; 2015b; Masud et al. 2015). Thus, alkaline slag has the combined properties of lime and gypsum to correct soil acidity. The combined application of alkaline slag with organic materials was more effective than alkaline slag alone to correct subsoil acidity (Masud et al. 2014; Li et al. 2015b) and increase crop yield (Table 8.1).

Table 8.1 Effect of inorganic and organic amendments on the yields of canola seeds and sweet potato in an Ultisol (Pan et al. 2019)

Treatment	Soil pH (1:2.5 water)	Canola seed yield (t ha ⁻¹)	Sweet potato yield (t ha ⁻¹)
Control	4.37 c	0.38 h	6.11 e
Lime	4.62 b	2.31 c	7.38 abc
Peanut straw (PS)	4.41 c	0.84 f	6.45 de
Canola straw (CS)	4.38 c	0.61 g	6.11 e
Organic manure (OM)	4.40 c	1.63 e	6.70 cde
Alkaline slag (AS)	4.75 a	1.94 d	7.15 bcd
AS + PS	4.73 a	2.06 c	7.64 ab
AS + CS	4.75 a	2.43 b	8.07 a
AS + OM	4.82 a	2.61 a	7.55 ab

Means followed by the same letter within a column are not significantly different ($P < 0.05$).

Organic amendments

Some organic amendments (manures, crop straws, and biochar) have also been applied to ameliorate acid soils due to their alkaline nature (Naramabuye and Haynes 2006; Lauricella et al. 2020). The application of organic amendments also increases soil organic matter and cation exchange capacity, and thus increases soil pH buffer capacity (pHBC) (Nelson and Su 2010; Xu et al. 2012). Long-term manure application increased the pHBC of an Alfisol and an Ultisol by 60% and 66%, respectively (Shi et al. 2019a). Four crop-straw biochars increased the pHBC of an Ultisol by 85-200% (Shi et al. 2017). The organic anions, such as $-\text{COO}^-$ and $-\text{O}^-$, on the added manures or biochars are associated with protons to form neutral molecules ($\text{Biochar}-\text{COO}^- + \text{H}^+ \rightleftharpoons \text{Biochar}-\text{COOH}$), which slows down pH decrease

during soil acidification (Shi et al. 2019a).

Manure - Animal manures and composts are generally rich in nutrients (e.g., N, P, K, Ca, and Mg) and some are alkaline (Goss et al. 2013). Application of manures can neutralise soil acidity and increase soil pH directly. The ammonification of organic N and the decarboxylation of organic anions in manures and composts also contributes to pH increase (Yan et al. 1996; Khalil et al. 2005; Xiao et al. 2013). In addition, the increased soil organic matter after the application of manures and composts can alleviate Al toxicity through the formation of Al-organic complexes (Hagvall et al. 2015). In a pot experiment, Masud et al. (2020) showed that the application of 15 g kg⁻¹ poultry litter raised soil pH by 0.5 units and increased the biomass of maize by more than 10 times in an acid soil.

Crop residues - Crop residues are a tremendous natural resource containing nutrients and alkaline substances. Therefore, they are also used to ameliorate soil acidity. The increased pH after crop straw application is caused by (i) the association of organic anions with H⁺; (ii) the decarboxylation of organic functional groups; and (iii) ammonification of the organic-N in the residues. The relative contribution of the above processes is determined by the properties of crop residues and initial soil pH (Ritchie and Dolling 1985; Butterly et al. 2011). Generally, the pH increase after crop straw incorporation can be predicted by straw ash alkalinity and pHBC of the soil. In general, the ash alkalinity of legume residues is significantly greater than that of non-legume straws due to more base cations being absorbed by legumes than non-legumes (Tang and Rengel 2003; Wang et al. 2011). However, when legume residues are applied into acid soils, mineralisation and subsequent nitrification of the organic N in the residues releases H⁺ and counteracts the alkalisation effect (Wang et al. 2011). While, if residues are applied with a nitrification inhibitor, such as dicyandiamide, the nitrification is inhibited without stopping mineralisation, increasing the opportunity for plant uptake of mineral N, and decreasing the potential for NO₃⁻ leaching. The ameliorating effect of legume residues on soil acidity was enhanced greatly by the addition of a nitrification inhibitor (Mao et al. 2010).

Biochar - Biochar is produced from biomass pyrolysis under complete or partially hypoxia. Due to the abundant nutrients and alkaline substances, biochar has been regarded as an alternative amendment for correcting soil acidity and improving nutrient supply (Shi et al. 2019b). For example, a 4-year field experiment showed that canola seed yield and biomass production were significantly increased by crop-straw biochars due to their ameliorating effect on soil acidity and Al toxicity (Zhao et al. 2020b). A global-scale meta-analysis has also shown that biochar elicited a 25%

average crop yield increment in the tropics due to the amelioration of acidity (Jeffery et al. 2017).

The alkalinity of biochar, which exists mainly as carbonate and organic functional groups, determines their liming effect (Yuan and Xu 2011; Yuan et al. 2011). Feedstock and the pyrolysis process significantly influence the characteristics of biochar (Shi et al. 2019b). There is a strong correlation between the total equivalents of base cations in feedstocks and alkalinity in biochars (Fidel et al. 2017) because the base cations in biochar are accompanied with carbonates or organic anions. The alkalinity of the biochars derived from legumes is greater than that of the biochars derived from non-legumes (Yuan and Xu 2012). In addition, the biochars derived from plants growing in soils with more base cations generally contain more alkali substances (Dong et al. 2019).

Fertiliser management

The application of chemically- or physiologically-acid fertilisers, such as superphosphate, ammonium sulfate, and ammonium chloride, can accelerate soil acidification. Conversely, basic fertilisers can significantly decreased soil acidity. Thus calcium-magnesium phosphate is recommended as a P fertiliser for acid soils. The ability of different N fertilisers to acidify soil follows the order of $(\text{NH}_4)_2\text{SO}_4 > \text{NH}_4\text{Cl} > \text{NH}_4\text{NO}_3 > \text{urea}$ (Chien et al. 2008). Over-application of ammonium-based fertilisers and the subsequent leaching of nitrate are the main reason for accelerated soil acidification in croplands (Guo et al. 2010). Therefore, N fertilisers should be applied according to crop demand to minimise soil acidification. Application of ammonium-based fertilisers with a nitrification inhibitor may also decrease soil acidification (Jiang et al. 2014).

The uptake of nitrate-N by crop roots leads to alkalisation in rhizosphere soils. Application of $\text{Ca}(\text{NO}_3)_2$ improved the growth of wheat roots below 15 cm, which in turn increased N capture and thus alkalisied the subsoil of a Podzol (Weligama et al. 2010). The ameliorating effect of $\text{Ca}(\text{NO}_3)_2$ on subsurface acidity was also observed in a field trial (Conyers et al. 2011; Tang et al. 2011).

Crop selection

Growing acid-tolerant crops is an important strategy for economic food production in acid soils. Crop species and genotypes vary considerably in their tolerance to soil acidity and Al toxicity. Canola, barley, and wheat are generally sensitive while rice, lupin, sugarcane, rye, buckwheat, and oats

are tolerant to soil acidity (Fenton et al. 1996; Yang et al. 2005). Significant genotypic variation in tolerance to acid soils has also been observed in crops such as wheat, barley, and legumes (Ryan 2018). Growing acid-tolerant cultivars/genotypes to obtain higher crop yield in the presence of acidity has been confirmed as a viable strategy under field conditions (Flores et al. 1988; Tang et al. 2003a; 2003b). Screening acidity-tolerant crops and cultivars can thus be an important strategy to maintain productivity in low input management systems (Fageria and Nascente 2014).

However, if acidification of agricultural soils is allowed to continue over time, they will eventually become unsuitable for even the most tolerant species/cultivars. Thus, the amelioration of acidity and a reduction in acidifying agricultural practices are essential.

Areas requiring further research

Aluminium toxicity is the main constraint limiting crop production in acid soils. It is well known that soil exchangeable H^+ is not stable and will dissolve aluminosilicates to form exchangeable Al^{3+} during soil acidification. This leads to the mobilisation of soil Al and is responsible for Al toxicity in acid soils. However, it is not well understood how Al is liberated from Al-O octahedral structures in clay minerals during reactions with H^+ ions. The mechanisms of Al mobilisation from clay minerals need to be investigated in the future.

Critical values of soil pH and Al concentration are important soil parameters for the amelioration of soil acidity. However, information regarding critical values is limited. The critical values of soil pH and Al concentration vary with crop species and genotypes as well as with soil physicochemical properties (Baquy et al. 2018). Field experiments should be conducted to establish the relationships between crop yield and soil pH and Al concentration and then define critical values for main crop species and soil types. The effects of soil physicochemical properties on these critical levels should be investigated and predictive models developed.

Several methods have been used to determine lime requirements for the amelioration of soil acidity (Vogel et al. 2020). Soil pH buffer capacity (pHBC) is an intrinsic soil property that quantifies a soil's ability to resist pH change. It can also be used to estimate lime requirements to ameliorate soil acidity. However, current methods generally under-estimate actual pHBC and provide different values based on measurement procedures. Wang et al. (2015) recently attempted to optimise the methods to measure soil pHBC. Future investigation may be conducted to understand the mechanisms by which soils buffer pH changes and to evaluate the

contributions of various soil components to soil pHBC. Then a model may be developed to predict soil pHBC and hence lime requirement based on soil components and basic properties.

Anthropogenic activities have accelerated soil acidification and will continue to do so in the future. The fast re-acidification of limed acid soils has been observed. In the future, new management options need to be developed to combat soil acidification. First, management options should minimise proton inputs and production in soils to decrease soil acidification rates. Such management can include increasing the use efficiency of ammonium-based fertilisers to reduce application rates in acid soils. Second, farming practices, such as the application of manures, crop straws, and biochars, should be encouraged. Developing organic amendments with ample functional groups will also increase soil pHBC to enhance soil resistance to acidification. The application of liming materials combined with organic amendments will ameliorate soil acidity and improve soil fertility simultaneously, which is a promising agricultural practice. Inorganic-organic complexing amendments should be developed for amelioration of both topsoil and subsoil acidity in the future.

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CHAPTER 9

NUTRIENT DEFICIENCIES

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Abstract

Nutrient deficiencies in soils can manifest for diverse reasons, some of which are inherent or closely related to soil types or soil properties. This chapter reviews knowledge on (i) the extent, distribution, and characteristics of soils with inherent nutrient deficiencies; (ii) identifying and managing soil nutrient deficiencies; and (iii) improving crop production on nutrient deficient soils. It also identifies areas requiring future research. Definitions of nutrient deficient soils are dynamic, which in part reflects soil-plant-management interactions. Nutrient deficiencies can be diagnosed and addressed by traditional agronomic strategies and through the development of novel crop genotypes with higher physiological nutrient use efficiency. Classical plant breeding complemented with powerful molecular technology will continue to secure long-term breeding progress to ensure ongoing crop productivity gains in nutrient deficient soils. Rapid in-field soil and plant testing is also needed for diagnosing plant nutrient deficiencies. New tools and technologies will assist farmers to correctly apply plant nutrients, but outreach and knowledge transfer to farms around the world that is site and crop specific is of vital importance.

Keywords: Fertility, acidity, alkalinity, dystic, nutrient mining, plant breeding, Sprengel-Liebig.

Introduction

Nutrient deficiencies: a crop-based perspective of soils

In agricultural systems, nutrient deficiencies refer to one of the essential plant nutrient elements derived from soil being in sufficiently short supply that the normal growth and reproductive function of the plant or, more operationally, target yield is not achieved. Thus, for the purpose of this chapter, crop species are the perspective through which the availability of nutrient elements in soils is discussed. Essential plant elements are defined as being necessary for a plant to grow *and* complete its life cycle (i.e., reproductive functions). Of the 18 essential plant elements, three are derived from the atmosphere or water: carbon (C), oxygen (O), and hydrogen (H). The remaining 15 are classified by their relative abundance in plant tissues, and thus plant needs, as macronutrients (0.1-10.0% dry mass) and micronutrients (<0.1%, often mg kg⁻¹). The macronutrients are nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), and sulfur (S), and are often subdivided into primary macronutrients (N, P, K) and secondary macronutrients (Ca, Mg, S). By virtue of their greater requirement by plants, macronutrients are generally the most commonly deficient in soils and are added in agriculture via fertilisers. Micronutrients include iron (Fe), manganese (Mn), zinc (Zn), boron (B), copper (Cu), chloride (Cl), molybdenum (Mo), cobalt (Co), and vanadium (V). More recently, sodium (Na) (Brownell 1980), nickel (Ni) (Eskew et al. 1984; Brown et al. 1987) and silicon (Si) (Takahashi et al. 1990; Epstein 2001) have also been proposed as essential micronutrients. In some cases, elements such as Co have been argued to be “beneficial” but not “essential” (Bolle-Jones and Mallikarjuneswara 1957).

A salient challenge to identifying truly essential micronutrients is ensuring their full exclusion in plant growth media, especially for soils, but even in controlled *in vitro* conditions. Plants require these elements in such small quantities that it is difficult to keep them out of the plant growth medium and therefore identify essentiality. However, it can be argued that these micronutrients are rarely of practical (i.e., agricultural management) concern for most soils and crop species, though exceptions exist (e.g., highly weathered and/or derived from hyperfelsic parent material). It is also worth noting that some plant essential elements can be toxic to plants and soil flora at high soil concentrations (e.g., B, Cu, Cl, Co).

The “nutrient deficiency” status of a soil can also be highly dynamic. Deficiency can vary with plant species, soil, and soil-plant interactions influenced by the growing environment and – for crop species – management practices such as fertilisation and liming. For example, many

plants have adaptations for acquiring nutrients, ranging from root architectural plasticity or mycorrhizal mutualisms (Morgan 2013). In extreme cases, certain nutrients may or may not be essential, and thus deficient, for specific plant species, in particular silicon (Si) for rice (*Oryza sativa*) (Epstein 2001). Plastic traits such as rooting depth and external influences such as weather mean that there can be substantial variation in the type and degree of nutrient deficiency for a given soil.

In soil science, paedogenic classifications entail recognition of nutrient deficiencies. In World Reference Base (WRB) and United States Department of Agriculture (USDA) Taxonomy, soil nutrient status is captured by the modifiers of eutric and dystric. However, the taxonomic designations of eutric and dystric are not nutrient element-specific, but rather denote low base saturation: the proportion of cations on the cation-exchange complex as Ca, Mg, and K, as opposed to H and aluminium (Al). In WRB, the base saturation threshold of 50% is used to identify eutric soils (>50% base saturation determined by neutral ammonium acetate) and dystric soils (<50% base saturation). Since base saturation is often related to crop availability of other nutrient elements, in particular P, as well as to root growth due to Al toxicity, base saturation serves as a general proxy of nutrient sufficiency vs deficiency. Additional sub-categories of eutric and dystric are used in WRB to identify extremes of nutrient status, such as hypereutric (>80% base saturation) or hyperdystric (<20% base saturation). For example, epidystric soils have <50% base saturation within 20-50 cm depth, whereas endoeutric soils have >50% base saturation at 50-100 cm depth. In USDA soil taxonomy, thresholds and methods for dystric and eutric classification based on base saturation by sum of cations varies by soil order. For example, the dystric threshold is 60% base saturation for Entisols and Inceptisols. For Andisols, dystric is based on the sum of extractable bases by neutral ammonium acetate extraction in addition to KCl-extractable Al^{3+} being <15.0 cmol_c kg⁻¹ (Staff 2003).

History of research on soils with inherent nutrient deficiencies

The first conceptualisation of nutrient deficiencies in soils predates knowledge of nutrient elements, and largely focused on edaphic properties such as texture and organic matter content. These characteristics represent a pool of the mineralisable nutrient elements N, P, and S. Agricultural specialists in ancient Rome classified soils based on visual and tactile attributes that can be ascribed to texture, organic matter, and emergent properties that result from these, such as water-holding capacity. In *De Re Rustica*, the Roman agronomist Columella categorised soils on the basis of

three pairs of opposing traits: fat vs lean, loose vs compact, and moist vs dry. The first of these most closely align with current understanding of nutrient deficiencies (“lean”). In ancient Rome and China, knowledge that animal manure and green manures improved crop growth led to the notion that humus was the *direct* source of plant nutrition, a concept that persisted until the 19th century (Korcak 1992). With advances in chemistry in industrialising Europe, increasing knowledge of nutrient elements began to shape a reductionist approach to plant nutrition. In the first half of the 19th century, the agricultural chemist Carl Sprengel catalysed a shift from the humus hypothesis to an element-specific understanding that posited direct use of inorganic or “mineral” elements by plants. Sprengel also conceptualised the Law of the Minimum in 1828, which is unfairly credited to Liebig, who formulated it more explicitly in 1840. The Sprengel-Liebig Law of the Minimum provided a paradigm shift in addressing nutrient deficiencies because it states that plant growth is most limited by the most deficient nutrient (Liebig 1840; Sprengel 1828). This provided an order of operations to address nutrient deficiencies: the relative abundance of each nutrient element must be first identified, and then the most deficient element (relative to plant needs) must be increased in the soil.

Of course, there was still work to be done to disentangle the relative importance of the fifteen soil-derived essential plant nutrient elements. The task of the early 20th century then was to concretise the Sprengel-Liebig Law by quantifying the relative importance of “plant food elements”, how these varied by soil type and crop species, and how soil deficiencies could be economically addressed (Stewart 1920). The work of Cyril Hopkins at the University of Illinois at the turn of the 20th century, for example, focused on establishing the importance of N, P, K, and S for crop productivity and ameliorating element-specific nutrient deficiencies with inputs such as rock phosphate. Without soil tests to enable precise diagnosis of element-specific deficiencies, much of this work focused on total nutrient stocks, which in part motivated the first state-wide (Illinois) soil survey in the United States in 1899. However, total concentrations or stocks of a given nutrient element are poorly related to crop deficiencies. Soil testing to identify nutrient deficiencies did not begin in earnest until the second half of the 20th century for P and K (Hopkins 2015; Peck 1990).

Soils with inherent nutrient deficiencies

There are multiple reasons why nutrient elements are deficient in different soil types and how this impacts crop productivity. Ultimately, if the level of the available nutrients in the soil are below the required amount

for optimum crop growth, deficiency will occur, though symptoms may not always be visible. Unusually low or high soil pH levels can make certain nutrients unavailable to the plant. Poor growing conditions such as excessively wet or dry soils, cold weather, and soil compaction can reduce the root's ability to access sufficient nutrients. Fertilisation to directly add deficient nutrient elements and management practices to increase the availability and/or accessibility of soil nutrients to the plant are standard approaches to address nutrient deficiencies. Crop genetics and breeding can play a major role in increasing nutrient uptake and nutrient use efficiency.

By soil groupings

In general, soils can be grouped according to properties that generally result in deficiencies of certain nutrient elements. Such groups include low organic matter soils, coarse-textured soils, strongly acidic soils, and alkaline soils. These four groupings reflect (1) organic matter as a reservoir for organic N, P, and S, which become crop available via mineralisation; (2) low clay, and generally low CEC, with poor nutrient retention and high leaching risk; and (3) low; or (4) high pH, both of which reduce the crop availability of nutrient elements by precipitation or sorption to minerals.

Low organic matter soils.

Soils with low organic matter generally tend to be N deficient. A testament to this is the higher N fertiliser requirements for cereal grain crops on lower organic matter soils in the U.S. Corn Belt, which have lower provisioning of crop available N via mineralisation of organic N (Sawyer et al. 2006; Stanford 1973). Phosphorus and S are also present in organic matter, but unlike organic N, organic P and S are generally indirectly bound to carbon (C) via ester bonds (C-O-P, C-O-S). Hydrolysis of these organic bonds via extracellular enzymes is a key step to mineralise these nutrient elements into crop-available forms.

Coarse textured soils.

Soils with high sand and low clay content can lead to nutrient deficiencies due to their low CEC and high leaching potential. CEC generally increases with clay and organic matter content, although this depends on clay type: CEC of pure phyllosilicate clays can range from 3-15 cmol kg^{-1} for kaolinite to 80-100 cmol kg^{-1} for montmorillonite. For organic matter, CEC is generally thought to range between 200-400 cmol kg^{-1} (Sparks and Sparks 2003), though the strong influence of organic matter

composition and soil pH can mean soil organic matter CEC can be as extreme as 30-2800 cmol kg⁻¹ (Ross et al. 1991; Silber et al. 2010). Critically, the CEC of organic matter is pH-dependent, increasing with pH. Given the relative amounts of mineral particles versus organic matter, however, only for low (mineral) CEC soils can organic matter offer agronomically appreciable improvements in CEC.

The high infiltration rates of coarse-textured soils, which already tend to have lower CEC, increase the risk of nutrient losses by leaching. Over time, leaching tends to remove so-called “base cations” (Mg, Ca, K) from the soil profile, leading to deficiencies. Moreover, replacement of these nutrient cations on the CEC by H⁺ and Al³⁺ entail high exchangeable acidity, and thus low pH that can further reinforce nutrient deficiencies via effects on nutrient availability (e.g., P) and root stunting. As a result, coarse-textured soils sometimes exhibit multiple co-deficiencies in micronutrients. For example, in the southeast U.S. coastal plains, sand-dominated soils have very low CEC (< 6 cmol kg⁻¹) and marked Mn and B deficiencies (Brown et al. 1997; Haby 2002). Given the low nutrient cation retention capacity, inputs must be managed carefully (e.g., split application) to avoid their loss.

In general, increasing organic matter content and pH in combination can enhance CEC appreciably in coarse-textured soils with low CEC values. This can be achieved by liming, which has the additional benefit of displacing Al³⁺ and H⁺ from the exchange complex to increase the number of binding sites for nutrient cations (Fageria and Baligar 2008). Increases in CEC, which are due to the deprotonation of carboxylic acid functional groups and the surface hydroxyls of metal hydroxides, increase cation binding sites (Sparks and Sparks 2003). However, the chemistry of liming-induced pH changes are complex and do not necessarily produce expected changes in CEC and Al availability (Ross et al. 1991).

Acidic soils.

Soil acidity is a major constraint to crop growth globally and afflicts nearly one-third of all soils (Rengel 2003). Soil acidity directly drives deficiencies of nutrient elements present in anionic form, notably P as phosphate and Mo as molybdate. In contrast, N, K, and S are less directly affected by soil pH. The availability of cationic micronutrient elements such as Fe, Cu, and Zn is greater at acidic pH. A comprehensive evaluation of soil acidity impacts on plant nutrient (and non-nutrient) element uptake is provided by Tyler and Olsson (2001). It is important to note that nutrient deficiencies in acidic soils, largely focused on the macronutrient of P, are mediated by complex interactions centred on Al: Al interactions with P (direct) and Al toxicity to roots (indirect) (Haynes 1982). Thus, acidity does

not so much decrease P availability as it increases the solubility of Al and thus formation of Al-P complexes that in turn reduce P availability and limit crop productivity (Margenot et al. 2016; Barrow 2017).

Increasing pH by liming entails multiple and interacting processes that can collectively help alleviate nutrient deficiencies: decreasing Al solubility and thus immobilisation of P and root impairment; adding large quantities of Ca and, if dolomitic limestone is used, Mg; increasing the availability of Mo (Robinson et al. 1951); enhancing bacterial activity and thus mineralisation of nutrient elements from organic matter (Balloni and Favilli 1987); increasing biological N fixation (Alves et al. 2021); and decreasing the solubility of Al that otherwise can stunt plant roots (Friesen et al. 1980). Liming can also induce deficiencies in micronutrient elements, in particular metal cations such as Cu, Zn, Fe, and Mn, because the availability of these decrease at pH >6.5 (Sherman et al. 1942; Havlin et al. 2013).

Alkaline soils.

Alkaline soils are generally defined as soils with a pH surpassing 7.5, and form in arid environments in which evapotranspiration exceeds precipitation. Usually, such soils have high carbonate content, often as calcium carbonate. Alkaline soils often have deficiencies in Fe, Mn, B, Cu, and Zn due to precipitation of these as hydroxide and carbonate forms. At alkaline pH values, phosphate ions tend to react with Ca and Mg to form less soluble compounds and can sorb to calcite (Bertrand et al. 2003). Acidification of soils using elemental sulfur or sulfuric acid can be used to decrease pH and increase the availability of these nutrients (Kissel et al. 2020), analogous to rhizosphere acidification by some crop species (Li et al. 2007). Application of metal micronutrients in chelated forms can also mitigate immobilisation in alkaline soils (Riaz et al. 2020).

By region

The tropics, temperate, and boreal zones account for 38%, 50%, and 12% of the world's land area, respectively (Eswaran et al. 1992). Temperate soils are generally viewed as more favourable for agriculture than tropical soils. This is because of higher nutrient stocks due to a lower degree of weathering as a result of recent glaciation and a cooler climate. However, there are exceptions in both regions, with highly productive volcanic and fluvial soils found in the tropics, and poorly developed and infertile soils in temperate regions. These broad and climatically or geographically based characterisations of soils are unscientific (e.g., they ignore the complexity of soil-forming factors) and largely unhelpful for understanding nutrient

deficiencies.

A particularly illustrative example is the common misconception that all tropical soils are highly weathered, and thus generally nutrient deficient. This is equivalent to stating that all temperate soils are young and fertile (Sanchez 2019). Historical overgeneralisations of soil types and soil fertility in the tropics have led to widespread misconceptions of tropical soils. In an evaluation of the history of the term “tropical soils”, the misconceptions and outright ignorance of the tropical mantle being laterite persisted through the 1930s and can be traced to often-quoted articles by authors who had never studied soils in the tropics (Eswaran et al. 1992; Hartemink 2002). Davies Brian (2009) argues against the misconception that tropical soils are inherently low in fertility: “Sadly, the myth that all soils between the tropics of Cancer and Capricorn are nutrient poor and old and generally contain laterite dies hard.” Though it is true that most highly weathered soils occur in the tropics as a result of geological stable landforms, high precipitation, and temperature that entail intense and prolonged weathering, not all soils in the tropics are highly weathered.

Eroded soils

A universal threat to soil nutrient availability is erosion: loss of soil from a given location by wind and water, and often enabled by bare soil exposure and mechanical disturbance, namely tillage in agricultural systems. Soil erosion under agriculture is commonly an order of magnitude or more greater than baseline, especially on slopes (Montgomery 2007). Erosion can undermine efforts to correct nutrient deficiencies and can aggravate or create nutrient deficiencies due to loss of topsoil (usually A horizons) in which the majority of nutrient elements are located. From the perspective of managing nutrient deficiencies, erosion entails a nutrient transfer that can aggravate or attenuate nutrient deficiencies. Indirectly, the soil remaining after erosional loss (“desurfacing”) may have indirect constraints to nutrient availability or utilisation, e.g., by restricting root growth or having lower water holding capacity (Lal and Moldenhauer 1987). Roots may also be limited to soil horizons with root-restrictive features such as fragipans or Al toxicity, resulting in indirect limitation to nutrient uptake (Rhoton 1990). The economic loss from nutrient export by erosion, and the cost of rebuilding nutrient stocks and improving physical and chemical conditions so they are favourable to crop roots, can be substantial.

Managing soil nutrient deficiencies

Diagnosis

Two ways to identify soil nutrient deficiencies are to measure nutrient concentrations in the soil or in the plant grown in the soil. Generally, soil testing is performed to estimate the likelihood of a yield increase from adding a nutrient element. The second, and not mutually exclusive approach, is to evaluate the plant itself, either qualitatively via visual symptoms of deficiency, or quantitatively via foliar testing. Both approaches have made advances since their inception in the mid-20th century, and both would benefit from further improvement.

Soil testing

Testing soils for nutrient elements is an indirect and probabilistic way to assess whether a given crop species will be deficient in a soil-derived nutrient. Most soil tests used today are (1) probabilistic, and (2) correlative, meaning that they do not actually measure the crop-available nutrient pool but rather the likelihood of crop yield increases occurring with nutrient addition. Timing, source, placement, and rate of the nutrient application interact with site-specific conditions, as well as germplasm and other management practices, to determine the extent to which a yield response occurs. A common misunderstanding is that soil tests such as Mehlich-3 are a measure of the plant-available nutrient element in the soil and correspond to plant uptake of the element. This is not so: soil tests are operationally constructed to predict the probability of a yield response and are not designed to measure an *in-situ* soil pool or correlate with plant tissue element concentrations.

Visual symptoms.

Symptoms of nutrient deficiency may include stunted growth, death of plant tissue, or yellowing of the leaves caused by a reduced production of chlorophyll. However, visual symptoms can be confusing if there are multiple nutrient deficiencies. Symptoms associated with different nutrients may also resemble crop injury caused by herbicide damage, pest damage, and/or the effects of plant diseases. Without laboratory tests, nutrient deficiencies may be difficult to determine since these may appear in many ways in a plant, including stunting or reduced growth; off-coloured leaves that are often white, dull green to yellow, or purple; abnormally shaped leaves, stems, and roots; and stunting of plant organs, notably the root

system. *Chlorosis* is a type of deficiency symptom characterised by yellowing that can be generalised over the whole plant, localised over individual leaves, or isolated between some leaf veins (interveinal chlorosis). *Necrosis* is a type of deficiency symptom characterised by death of plant tissue, sometimes in spots.

Nutrient deficiencies generally appear first in the oldest leaves when N, P, K, and Mg are limiting. These nutrients move from one part of the plant to another as needed via *translocation*. Younger leaves and terminal buds show a deficiency when S, Fe, Ca, Zn, Cu, B, Mn, or Cl are limiting. These nutrients do not readily move about in the plant. As a deficiency becomes more severe, visual symptoms may spread to the whole plant, leaves may become more chlorotic or bleached in appearance, or stunting, deformity and death of plant parts may become more extensive. Deficiencies of certain nutrients, such as S and P in cereal crops, may manifest only early in the growing season because of immature root development or cold weather that hinders mineralisation.

Foliar testing.

Plant tissue testing is the most accurate of all diagnostic tools for nutrient deficiencies, particularly when paired samples are taken. Plants with possible deficiency symptoms should be compared with nearby plants that appear to be healthy. Tissue testing is also the only way to detect such “hidden hunger”, as mild nutrient deficiencies may not produce obvious visual symptoms such as leaf yellowing or chlorosis. Nutrient analysis of plant tissue at different growth stages is a major tool for determining which nutrients are limiting growth. Plant monitoring with tissue testing is especially important with high value crops. Plant nutrient content will vary depending on the type of plant being sampled, the specific plant part sampled, and the growth stage of the plant.

If specific sampling instructions are not found for selected crops, the rule of thumb is to sample recently matured leaves, which for most crops are the upper leaves, just before the beginning of the reproductive stage. The sufficiency level for a particular nutrient may depend on genetic and environmental factors. A sample from a healthy area of a field establishes the tissue nutrient levels needed for healthy plants given the genetics, soil type, and weather. If the observed symptoms are typical for a particular nutrient deficiency, and if the tissue levels are higher for that nutrient in the healthy area than in the affected area, then deficiency of that nutrient is likely. For example, if browning of the edges of older (lower) leaves is observed, and tissue levels of K are higher in healthy plants than in affected plants, then K deficiency is a reasonable diagnosis.

The value of in-field plant testing results has stimulated an interest in “rapid test kits” for determining plant nutrient deficiencies. These kits often have the advantages of being portable, less expensive than standard laboratory methods, and suitable for use by growers and consultants without extensive laboratory training. The usefulness of rapid tests for plant nutrient management depends on the availability of research-based information for interpreting nutrient concentrations in plant sap for specific crops. Disadvantages of many rapid test kits are a lack of quality control (e.g., use of standards and blanks), dependence on a single supplier for reagents and parts, and improper storage effects on the viability of reagents. Once a diagnosis of nutrient deficiency is made, the appropriate nutrient may be applied immediately, or for the next growing season, to alleviate the problem.

Correction for nutrient deficiencies

Once a nutrient deficiency is identified, there are two major soil-based approaches available. The first is increasing the amount of the nutrient in the soil. Application of fertilisers is the most common way to add nutrients to correct deficiencies, either in liquid or solid forms and via surface vs subsurface application. How and when fertilisers are applied can markedly influence how effective (magnitude and rapidity) the application is in correcting the nutrient deficiency. For a review of fertiliser application to soils, the reader is referred to Havlin et al. (2013). In some cases, soils can be circumvented altogether, and fertiliser can be applied directly to the plant as a *foliar application*. This can be particularly effective for micronutrients because many micronutrient elements are at risk of immobilisation via interactions with soil colloids (e.g., Fe in alkaline soils) and because the low requirement of crops for these elements means low amounts of nutrient per area are needed. Depending on the crop species and nutrient, foliar application can offer more rapid uptake and greater nutrient use efficiency than soil application.

The second soil-based approach is to modify soil properties that constrain the availability of nutrients, either endogenous or fertiliser, to increase crop availability. This is exemplified by managing pH, generally using lime to correct acidity, but also with sulfuric acid or elemental S to correct alkalinity. Other soil characteristics that can be modified include soil structure and compaction, which can limit root growth and thus access to nutrient elements. Increasing root accessibility to nutrients by improving soil structure is not necessarily based on inputs, however. For example, the use of organic matter inputs via manure or compost benefits soil structure

while also serving as a net input of nutrients. However, practices such as reduced/no-tillage and reduced traffic can gradually improve soil structure over time without the use of inputs. Similar modifications to increase root accessibility to subsurface soils, and thus nutrient elements, include tile drainage in subsurface saturated soils and ripping in soils with root-restrictive layers such as fragipans.

Improving global crop production in nutrient deficient soils

Nutrient deficiency can be addressed partially by traditional agronomic strategies and through the development of novel crop genotypes. Breeding for nutrient efficient crops has been promoted strongly for the past four decades and significant progress has been achieved in establishing genetic variation and genetic enhancement leading to release of nutrient efficient (N, P, Mn, Fe) and biofortified (Fe, Zn) crop varieties (Graham 1984; McDonald et al. 2013; Bouis and Saltzman 2017; Lynch 2019; Smith and Rao 2021). In recent years, researchers have contributed to future breeding strategies by identifying traits and genes that can increase yields on soils with restricted phytoavailability of essential mineral elements.

Conventional selection procedures consider plant performance in multiple environments over many years and are thus well-suited for adaptation to nutrient-limited soils and climate change. However, modern breeding technologies can help to accelerate the incremental accumulation of positive alleles for “invisible” physiological traits underlying plant adaptation to change in the environment. Major crops have been bred to give optimum yields under the conditions currently prevailing in the main areas of production. Conventional breeding has the goal of producing genotypes that are productive across a range of environments, but a significant genotype $\times$ environment (G $\times$ E) interaction arising from variations in the environment hinders this development (Cooper et al. 2020). Genotype $\times$ environment $\times$ management (G $\times$ E $\times$ M) interactions underpin many aspects of crop productivity. Both the environment and management interact strongly with the plant genotype, so that higher-order G $\times$ E $\times$ M interactions must be considered in both breeding and agronomy (Cooper et al. 2021).

The biggest increases in crop yields are likely to come through improvement of primary physiological processes such as C assimilation. However, any increase in C assimilation would cause a larger demand for N, P, K, and other nutrients, so the efficiency of nutrient use by crops must be maintained as a minimum requirement to enable increased yields to be achieved. Nutrient uptake will not only have to be maintained at the current

rate, but due to the dilution effect of the extra C assimilation, it will have to be enhanced to maintain concentrations of nutrients in crop products. Improvement in nutrient use efficiency may be realised in low-intensity agriculture by increasing the efficiency of uptake of nutrients by breeding for suitable root systems (Lynch 2019). However, in more intensive agriculture, increased nutrient use efficiency from remobilisation of macronutrients (particularly N) and micronutrients within the plants will be required. Prolonged root uptake and greater nutrient remobilisation are breeding targets, and there is sufficient plasticity in these characteristics in crop species for increases in efficiency to be achieved.

Two approaches for selecting characteristics to breed into crops to alleviate nutrient deficiency are the analysis of quantitative trait loci (QTLs) in closely related individuals or genome-wide association studies (GWAS) (Mitchell-Olds 2010). If QTLs for traits associated with plant nutrition are found, a database search for similar DNA sequences that have been identified previously can be performed to find out what genes are involved. These genes can then be manipulated in breeding programs using marker-assisted selection, where the single nucleotide polymorphism (SNP) is selected for. In GWAS, unrelated populations of a species have markers such as SNPs analysed to identify QTLs across the whole genome. Many QTLs have been identified in diverse genetic backgrounds, with significant phenotypic variance explained under various nutrient deficiencies in different crops (e.g., Ali et al. 2018; Magalhaes et al. 2018; Lynch 2019).

New tools like gene editing also provide breeders with completely new opportunities to accelerate the implementation and stacking of beneficial variants for important target traits (Mao et al. 2019). However, most major-effect genes for key adaptation traits are generally rapidly fixed in high-performing modern crop gene pools, and pleiotropic interactions associated with severe changes in plant phenology mean that breeders need to be cautious when promising huge returns from technological breakthroughs on a single-gene level. On the other hand, conventional selection processes in plant breeding driven by testing and selection in diverse environments over many years have proven well-suited for the gradual adaptation of modern cultivars to changing soil and climatic conditions. There is a need to reduce breeding cycle time, which could be achieved through the selection of appropriate parents, optimizing the breeding scheme, rapid fixation of target alleles, and combining speed breeding with breeding programs to optimise trials for release (Sinha et al. 2021).

The incremental accumulation of beneficial alleles for complex quantitative traits impacting yield and quality has been the fundamental basis of historical breeding progress in all crops, and this process can be

expected to also play a major role in breeding for nutrient deficient soils in the future. However, the retrospective breeding progress in most cases was simply the positive by-product of long-term, multi-environment selection for high yield and cultivar registration policies that encourage high yield stability (Snowdon et al. 2021). Breeders in the past did not have access to the resources and tools that today enable effective elucidation, genetic dissection, and targeted selection of key adaptive target traits with small, genome-wide effects under complex genetic control. An expanding phenomic and genomic database makes it increasingly feasible to exploit the vast power of new prediction methods based on artificial intelligence and deep learning. Even as this repertoire of breeding technologies expands, however, classical breeding theory and methodology will continue to secure long-term breeding progress to improve crop productivity in nutrient deficient soils.

Future research recommendations

Biological approaches to correcting nutrient deficiencies.

The use of biological agents such as microorganisms, enzymes, and microbial metabolites is a fast-growing sector of the agricultural input market. *Biostimulants* are specifically marketed for their beneficial effects on crop growth and/or yield specifically mediated by a non-nutrient addition effect (Du Jardin 2015; Sible et al. 2021). The 2018 U.S. Farm Bill defines biostimulants as “a substance or micro-organism that, when applied to seeds, plants, or the rhizosphere, stimulates natural processes to enhance or benefit nutrient uptake, nutrient efficiency, tolerance to abiotic stress, or crop quality and yield” (U.S. Congress 2018). At a fundamental level, this is similar to the long-recognised ecological services afforded by soil organisms like arbuscular mycorrhizae. However, biostimulants are generally used as *inputs* to agricultural systems, and often with effects limited to the growing season. Given the diversity of biostimulant products, it is difficult to generalise similarities and differences. This diversity also entails many hypothesised and often asserted mechanisms by which nutrient deficiencies may be ameliorated using biostimulants. In part to test not only advertised claims, but also to generate new biological approaches for increasing nutrient use efficiency, biostimulants present multiple research avenues.

Need for new tools and technologies.

Managing nutrient deficiencies at the field level is a challenging task and several methods individually, and in combination, are needed for efficient management. Several methods including soil testing, plant tissue testing, spectral response, fertiliser placement and timing, and vegetative indexes (e.g., leaf area index, normalised difference vegetation index) acquired through drones, handheld sensors, and satellite imagery are being tested and used, particularly for managing N deficiency (Sharma and Bali 2018). Precision agriculture tools, such as GreenSeeker, Holland Crop Circle, unmanned aerial vehicles (UAVs; also known as drones), and satellite imagery can compete with soil testing, but can only be used for a standing crop and are therefore limited to in-season N application methods (sidedressing). Tools that estimate N mineralisation, and thus soil N supply, based on soil organic matter content, soil properties, and forecasted weather are critical to inform N application rates. No one method can be used individually, but a combined approach will improve N use efficiency.

Sensors for in situ monitoring.

As a long-term prospectus, the use of optical sensors may help improve nutrient use efficiency and farmer profit only when precise data are collected from multiple sites considering soil, climate, and cultural practice variability and then a robust yield predicting model is developed. Sensor-based in-season nutrient management combined with a soil testing approach at the beginning of crop planting and a split application might be the answer to improve N use efficiency. However, sensor technology and algorithms need further development to create more reliable models. Precipitation data might help improve sensor models due to the economic and logistic impracticalities of soil moisture sensor data collection. Estimated leaf area index from sensors has shown promising results but needs more research trials for robust data collection. Additional wavelength ranges beyond red also need testing to confirm their applicability for high biomass crops where the red wavelength is saturated.

Genetic strategies.

The potential may be limited for single-factor genetic improvements to address nutrient deficiency and improve nutrient use efficiency, or other “single-factor” genetic technologies to pay off. Instead, there may be more promise in systematic agronomic approaches that harness Big Data and geospatial extrapolation frameworks to accelerate the process of optimizing

crop and soil management practices governing both yields and resource use efficiencies at the production scale (Cassman and Grassini 2020). Traditional, replicated field studies with two or three treatment variables are not up to the task of optimizing the 10–20 management factors (including selection of the optimal hybrid or cultivar) that affect fertiliser efficiency, crop yield, profit, and other key performance indicators. Newly available technologies, genomics rapid cycling (Crossa et al. 2017), high throughput phenotyping (HTP, phenomics) (Montesinos-López et al. 2017), and historical descriptions of environmental relatedness (enviromics) (Resende et al. 2021) are crucial to improving conventional breeding schemes and increasing genetic gain in developing nutrient efficient cultivars. “Breeder friendliness” is a key criterion when considering scaling out, because highly complex or unproven phenotyping methods are likely to be counterproductive (Reynolds et al. 2020).

Machine learning and other artificial intelligence approaches may play an important role in developing optimised, tailored, and site-specific management solutions, particularly once it becomes possible to move seamlessly from data to prescriptive analytics and automated decision making. The rapid spread of GPS-enabled smartphones presents a unique opportunity for directly reaching 2 billion people in smallholder farming households, provided the information is scientifically sound, offers actionable advice, and utilises feedback mechanisms to maintain rigor and enable improvement (Fabregas et al. 2019). Public access to high-quality, high spatial resolution data on soil properties, long-term weather records, and current and forecasted weather represent essential and relatively low-cost “public goods” to support the needed innovation and should be given a high priority by policy makers. To meet this challenge, there is a need for public investment and public access to high-quality data on soil properties, historical and real-time weather data, and water resources at a spatial resolution sufficient to drive innovation towards increasingly precise crop and soil management in farmers’ fields.

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CHAPTER 10

NUTRIENT AND ELEMENTAL TOXICITIES

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Abstract

Contamination of agricultural soils by potentially toxic elements (PTEs) is widespread all over the world due to the intensification of crop production and rising anthropic inputs in agriculture and the environment. Depending on the concentration and speciation of these elements in soil, especially in the rhizosphere, and on the plant species, significant amounts of toxic elements can be taken up by roots and allocated to plant tissues with deleterious consequences for crop growth, yield as well as quality, and safety of production. Understanding of element biogeochemistry at the soil-microorganism-root interface, as well as the fate of elements in plants, is thus necessary to develop agronomic practices to minimize the supply of PTEs in soil, reduce their uptake by crops, and enhance plant strategies to cope with the problem. This chapter focuses on limitations in crop production caused by the occurrence of hazardous levels of copper (Cu), zinc (Zn), cadmium (Cd), arsenic (As), and chromium (Cr) in soil. The issue is addressed starting from the diffusion and forms of such elements in soil, moving to the mechanisms of plant acquisition and allocation, then to the effects on crops, and concluding with the agronomic strategies required to tackle the problem. Research fields to explore to find innovative and sustainable solutions to the problem are also reviewed.

Keywords: Copper, zinc, cadmium, arsenic, chromium.

Introduction

Farmlands worldwide are affected by potentially toxic element (PTE) contamination. Some of these elements (e.g., Cr, As, Cd, Hg, and Pb) are toxic for plants at extremely low concentrations, whereas others (e.g., Mn, Cu, and Zn) are essential as micronutrients, but can become phytotoxic when present in soil at concentrations exceeding certain safety thresholds. The indiscriminate use of fertilisers and pesticides, soil amendments with organic materials rich in PTEs, and irrigation with wastewaters are the main sources of contamination. Emissions from industrial and mining activities may also contribute to farmland pollution as a consequence of toxic metal(loid) dispersion by wind or water runoff.

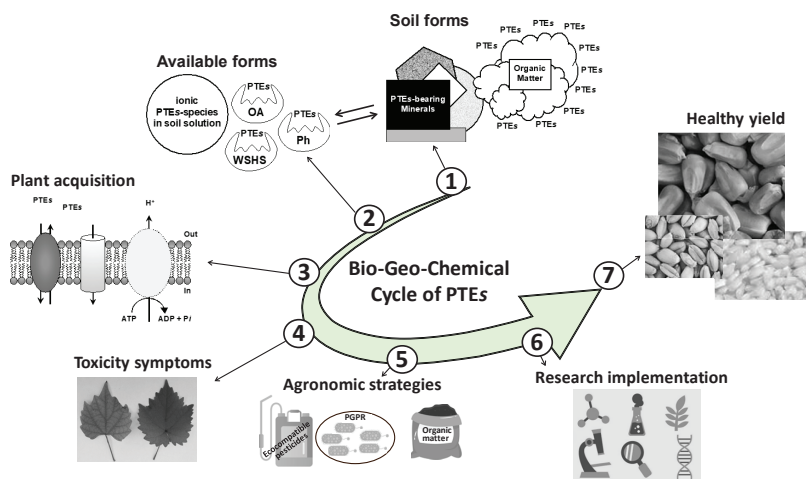


Figure 10.1 Representation of the approach used in the chapter to describe the dynamics of PTEs from the soil (1-2) to the plant (3-4), as well as the agronomic practices (5) and the scientific progresses (6) necessary to cope the problem and ensure the best crop performance (7).

The present chapter focuses on five PTEs selected for their prevalence in contaminated agricultural soils. Two of these are essential for plant growth (Cu and Zn) and three nonessential and hazardous (Cd, As, and Cr), with As and Cr also existing as anions in soil. As schematically depicted in Figure 10.1, for each element, an up-to-date and extensive overview is

provided examining the following key-aspects: (i) worldwide diffusion and extent of the contamination; (ii) element speciation and distribution in soil; (iii) mechanisms of plant/crop uptake and allocation; (iv) agronomic strategies to alleviate the impact on crop production; (v) required future research.

The case study of copper (Cu)

Copper is an essential element for living organisms and naturally occurs in soils at concentrations ranging between 2 and 50 mg kg⁻¹ (Oorts 2013). Soils rich in clay minerals and organic matter (OM) are characterised by higher background Cu values. Copper content may exceed background values in agricultural soils due to the application of Cu-based fungicides, sewage sludge, manure (especially pig slurry) and mineral fertilisers, as well as due to industrial activities like mining, smelting, and refining (Ballabio et al. 2018; Kumar et al. 2021). According to a recent study carried out on over 21,000 European agricultural soils, the highest Cu concentrations can be found in vineyards, with 15% of soils exceeding 100 mg Cu kg⁻¹, followed by olive groves and fruit orchards (Ballabio et al. 2018). The intensive use of Cu-based fungicides to control downy mildew, especially in organic vineyards exposed to warm-humid climatic conditions, can cause Cu accumulation in soils at levels even above 500 mg kg⁻¹ (Fernández-Calviño et al. 2009; Brunetto et al. 2016).

Copper mainly occurs in soil as Cu(II), and is distributed among the free, exchangeable, complexed, adsorbed, and coprecipitated forms. Organic matter content is the soil parameter most affecting solid-liquid Cu partitioning due to the high binding affinity between the metal and organic colloids (Kumar et al. 2021). Soil pH, texture, cation exchange capacity, and Eh also influence Cu speciation. Soluble Cu, usually at concentrations between 0.03 and 0.3 mg L⁻¹, is mainly (>90%) complexed by dissolved organic matter (DOM) (Oorts 2013; Blume et al. 2016). In acidic soils, Cu solubility and the free Cu²⁺ fraction increase (Oorts 2013). Although most Cu is adsorbed on OM, it can be also sorbed on oxides and clay minerals through specific or non-specific sorption (Brunetto et al. 2016). Less than 1% of total Cu is exchangeable, but this fraction can rise to 20% in strongly acidic soils (Blume et al. 2016). Copper precipitation is generally limited except in alkaline soils where Cu can precipitate as amorphous Cu(OH)₂, tenorite (CuO), or in calcareous soils, as Cu-carbonates. Due to its high affinity with OM, Cu is usually retained in the topsoil and is scarcely mobile. The increase of DOM concentration (for instance, after manure amendment or under rhizosphere conditions), associated with a high DOM

aromaticity, can also enhance Cu solubility and mobility under alkaline conditions (Araújo et al. 2019).

In plants, Cu can be acquired in its free ionic forms ($\text{Cu}^+/\text{Cu}^{2+}$), although the uptake of Cu complexes with organic molecules cannot be excluded, as recently demonstrated for Fe by Astolfi et al. (2020). After uptake, Cu is loaded into the xylem vessels, complexed with an organic ligand, and translocated to the shoot (Table 10.1).

High Cu bioavailability can impair root cell division and elongation and cause the development of thicker and shorter roots (Ambrosini et al. 2015; Marastoni et al. 2019; Cesco et al. 2021). As a consequence, nutrient and water acquisition is significantly impaired (Feil et al. 2020), leading to reduced growth and yield. In this respect it is interesting to note that appropriate nutrient contents, as well as a timely mechanism of leaf reallocation, appear crucial for the expression of response mechanisms against pathogens in resistant varieties of grapevine (Cesco et al. 2020). Once inside the plants, high Cu levels compromise the functionality of the photosynthetic system affecting both the pigment contents and the chloroplast structure and thylakoid membrane composition (Cambrollé et al. 2013; Rosa et al. 2020). Moreover, the nutritional disorder induces oxidative stress as a consequence of the imbalance between the neosynthesis of antioxidant enzymes and the increased concentration of reactive oxygen species (ROS) (Giroto et al. 2013; Tiecher et al. 2016a).

From an agronomic point of view, Cu toxicity can be managed with several approaches. Cover crops (such as *Avena sativa* L., and *Lolium perenne* L.) are a particularly promising practice, provided that there are no nutritional and water limitations. The sharing of the rhizosphere by two plant species lowers the metal availability, thus limiting the onset of toxicity. Furthermore, if the aerial part of the cover crops is removed from the field, it can decrease Cu accumulated in the soil over time (Cu phytoextraction). Nevertheless, species selection for phytoextraction is important as many plant species tend to accumulate significant amounts of Cu in the root system (Cu phytostabilisation) (Ambrosini et al. 2015; De Conti et al. 2019).

Soil microorganisms, including mycorrhizal fungi, can also reduce Cu availability to plants by metal chelation and sequestration at the cellular level (Cesco et al. 2021). In acidic soils, the application of alkalinizing compounds, such as lime, can also reduce bioavailability. Similarly, the application of OM (such as crop residues, manure, compost, biochar, etc.) with a low content of toxic metals can reduce the free Cu^{2+} fraction in the soil solution (Cu assisted phytostabilisation) (De Conti et al. 2019; Trentin et al. 2019).

Table 10.1 Summary of the mechanisms underpinning element uptake and translocation in plants. For a detailed description, please refer to the relevant review papers reported in the reference column.

Element	Bioavailable forms	Plant uptake	Translocation	Reference
Copper	Cu(I), Cu(II), Organic ligand-Cu complex	Cu Transporter Protein 1 (COPT1), Zn/Fe Permeases (ZIPs), Natural Resistance Associated Macrophage Proteins (NRAMP) & Iron Related Transports (IRT)	Xylem loading via Heavy Metal-transporting P-type ATPase	Cesco et al. 2021
Zinc	Zn(II), Organic ligand-Zn complex	ZIPs	Xylem loading via Yellow-Stripel-like (YSL) family, the heavy metal ATPases (HMAs), and the cation diffusion facilitator (CDF)	Gupta et al. 2016
Cadmium	Cd(II) Organic ligand-Cd complex	Apoplasmic way: H ⁺ /Cd ²⁺ exchange at root epidermis Symplasmic way: NRAMP, P-type ATPase, ZIP, low-affinity cation transporter (LCT) YSL	Chelated Cd is loaded in xylem vessels via YSL	Qin et al. 2020
Arsenic	As(III) As(V) MMA & DMA	Influx silicon transporters (Lsi1), nodulin-26-like proteins (NIPs), aquaporin channels Pi transporters (PHT) NIPs	Xylem loading via Lsi2 Xylem loading via PHT Xylem loading via unknown transporters	Abbas et al. 2018
Chromium	Cr(III) Cr(VI)	Passive diffusion Sulfate transporters (SULTR) and PHT	Cr is sequestered in vacuole of root cells. Cr(VI) can be allocated to the shoot exploiting long-distance transport mechanisms of Fe and S	Shahid et al. 2017

Future research to reduce the impact of Cu toxicity should aim to identify maximum Cu thresholds in soil based on the metal redox status, soil features, and plant species, to avoid toxicity phenomena. Further research to better understand the molecular and biochemical mechanisms underpinning Cu translocation and allocation in plant organs, as well as those activated in response to ROS generated by the excessive concentration of the metal, will also help identify agronomic strategies to limit toxicity. Additionally, a more in-depth knowledge of the mechanisms underlying the contribution of soil microorganisms to the biogeochemical cycle of the metal in the rhizosphere could contribute to the development of useful tools (microbial strains) aimed at increasing the tolerance of crops.

The case study of zinc (Zn)

Zinc is an essential plant micronutrient existing naturally in soils at concentrations between 10 and 300 mg kg⁻¹ (Kabata-Pendias and Mukherjee 2007). The highest values are found in soils rich in OM, clay minerals, and CaCO₃. However, values up to a few thousand mg kg⁻¹ have been measured in agricultural soils due to the increasing use of Zn-contaminated soil amendments (particularly manure, sewage sludge, and compost), and inorganic fertilisers (especially super phosphate and Zn salts). Industrial activities also contribute to Zn soil pollution via atmospheric deposition; for instance, Zn concentrations up to 3331 and 10,547 mg kg⁻¹ were found in soils near a smelting plant in Russia (Barsova et al. 2019) and a mining area in Greece (Angelone and Bini 1992), respectively. Although Zn soil pollution is a localised and often isolated phenomenon, the risks of toxicity cannot be ignored, especially in acidic and heavily amended soils. In acidic conditions, free Zn²⁺ is one of the most soluble and mobile trace metal cations, and among the most phytotoxic elements after Al³⁺ and Mn²⁺ (Stephan et al. 2008).

Sorption, complexation and, to a lesser extent, dissolution-precipitation mechanisms determine the solid-liquid partitioning of Zn in soil and, consequently, its mobility and bioaccessibility. Sorption is the main mechanism controlling Zn solubility, especially in neutral-alkaline soils where Zn is specifically adsorbed by OM and oxyhydroxides of Fe, Mn, and Al, forming stable inner-sphere complexes. Zinc non-specific sorption (ion exchange) on clay minerals and OM becomes more relevant under acidic conditions. Soluble Zn occurs mainly in an organically complexed form, and its concentration varies between 0.001 and 0.8 mg L⁻¹ (Blume et al. 2016) and can reach 22 mg L⁻¹ in severely polluted soils (Stephan et al. 2008). Other soluble forms are free Zn²⁺, Zn(OH)⁺ (the most prevalent in

alkaline soils), the ion pair ZnCO_3^0 , and in the presence of high sulphate and phosphate concentrations, ZnSO_4^0 and ZnHPO_4^0 ion pairs. Zinc precipitation in the form of oxides, carbonates, and silicates may occur in highly Zn-polluted soils (Mertens and Smolders 2013). Zinc mobility is strongly inhibited under reducing conditions.

Zinc is taken up mainly as Zn^{2+} , but organic ligand-Zn complexes can also be acquired by plant roots. After uptake, Zn is translocated in the shoot (Table 10.1), where it is mainly stored bound to chelating molecules, cell walls (Milner and Kochian 2008), or compartmentalised in the vacuole (Balafrej et al. 2020).

In soils with high levels of Zn availability, crops can acquire more Zn and less Mn, Fe, and Cu due to the ionic radius similarity among the divalent cations. Because of this nutrient competition, Zn concentrations in the order of 150–200 mg kg⁻¹ can be easily reached in vegetable tissues, impairing growth, particularly in sensitive species (Kabata-Pendias 2011). At the cellular level, these Zn concentrations are responsible for the enhanced production of ROS (Tiecher et al. 2016b), that in turn alter the selective permeability of the cell membrane (De Gara, Pinto, and Tomasi 2003). At the photosynthetic level, gas exchange, and thus CO₂ assimilation process, are significantly impaired compromising biomass accumulation and overall plant development (Cambrollé et al. 2013).

To treat Zn contamination, limestone applications are recommended for acidic soils to favour Zn adsorption by soil particles and increase the proportion of chemical species that are not bio-available, such as Zn(OH)^+ (Tiecher et al. 2016b; Trentin et al. 2019). The application of high-quality organic amendments can also help create Zn-organic ligand complexes and limit free ionic Zn forms in the soil solution (Trentin et al. 2019). Similarly, the release of exudates by cover crops can enhance Zn complexation. Root competition between multiple plant species for nutrient acquisition in the rhizosphere, and Zn accumulation at the shoot level of the cover crop plants, can also lower the levels of crop available Zn at the soil-microorganism-root interface (Tiecher et al. 2016b).

To prevent Zn contamination, superphosphate (which is rich in Zn and other metals) should be avoided in favour of mono- or di-ammonium phosphates, which have a lower Zn content and can reduce the availability of the metal in soil by complexation of the free ionic Zn and precipitation as Zn-phosphates (Alloway 2013). However, the acidification of the rhizosphere induced by ammonium phosphates and the consequent effects on Zn availability should be monitored in soils already high in Zn. More targeted and reduced use of Zn-based fungicides and fertilisers at the field scale should also be adopted to reduce Zn contamination.

Phytoremediation represents a promising method to remediate Zn-contaminated soils, although this is currently limited due to the low biomass of Zn-hyperaccumulator plants and the low bioavailability of Zn. In order to enhance the efficacy of phytoremediation, the use of microbes (i.e., PGPR, endophytic bacteria, and mycorrhizae) as biofertilisers can increase Zn mobility (Li et al. 2018). Taking advantages of recent genome editing tools, research efforts should focus on improving our knowledge of microbes-assisted phytoremediation (Basharat, Novo, and Yasmin 2018). New insights into both Zn hyperaccumulation and tolerance mechanisms in plants, as well as Zn-mobilizing microbes in soil will open new ways for soil decontamination technologies (Balafrej et al. 2020).

The case study of cadmium (Cd)

Cadmium is one of the most hazardous elements in soil due to its toxicity to all living organisms, its high mobility, and its potential bioavailability (Khanam et al. 2020). Except for soils derived from black shales or those rich in phosphorites, Cd geogenic concentrations are low and range between 0.06 and 1.1 mg kg⁻¹ (Kabata-Pendias and Mukherjee 2007). Elevated Cd contents in soil are associated with an abundance of clay, OM, carbonates, and Fe, Al, and Mn (oxy)hydroxides (Kubier, Wilkin, and Pichler 2019). Most anthropogenic sources of Cd in agricultural soils are derived from phosphate fertilisers, which contain Cd as an impurity at concentrations up to 120 mg kg⁻¹ P₂O₅ (Nziguheba and Smolders 2008). Mining, smelting, fossil fuel combustion, vehicular emissions, and agricultural use of contaminated sewage sludge and wastewaters are other anthropogenic inputs. Cadmium soil pollution is widespread worldwide and can even exceed levels of 500 mg kg⁻¹ (Kubier, Wilkin, and Pichler 2019). One third of Chinese agricultural soils are polluted with Cd, with average concentrations twofold higher than the national background value (0.097 mg kg⁻¹) (Yuan, Xue, and Han 2021). Conversely European farmlands are mostly free from Cd pollution (Tóth et al. 2016).

Up to 99% of soil Cd is associated with the solid phase (Christensen and Haug 1999), whereas only a small fraction (from 0.1 to 20 µg L⁻¹) exists in soluble forms (Smolders and Mertens 2013). From 55% to 90% of soluble Cd occurs as free Cd²⁺ (Kabata-Pendias and Mukherjee 2007), nevertheless other inorganic forms (i.e., CdCl⁺, CdOH⁺, CdHCO₃⁺, and ion pair CdSO₄⁰) and Cd-DOM complexes can also be found in the soil solution. Complexation usually enhances Cd solubility. Adsorption is the process mainly influencing Cd solubility, and it increases with increasing soil pH and, secondarily, OM content. Cadmium adsorption preferentially involves

organic colloids at pH <6.5 and, at higher pH values, (oxy)hydroxides and clay minerals (Smolders and Mertens 2013). Despite the formation of inner-sphere complexes, Cd adsorption is almost totally reversible since the metal forms weaker bonds with adsorption sites compared to other metals, such as Pb, Cu, and Zn (Kubier, Wilkin, and Pichler 2019). This makes Cd one of the most dangerous metal pollutants for plant uptake. Cadmium precipitation occurs only in the presence of high Cd^{2+} activity and under alkaline conditions. In flooded soils, Cd is immobilised due to the precipitation of CdS and adsorption by FeS and pyrite (FeS_2) (Khanam et al. 2020).

The mechanism underpinning Cd^{2+} entry into the root primarily involves apoplastic and symplastic pathways. The low molecular weight compounds released by roots (i.e., mugineic acids, MAs), can chelate Cd^{2+} forming metal ligand complexes, which can be translocated through the xylem (Table 10.1).

The presence of Cd in the environment, even in trace amounts, can cause serious problems for all organisms and its bioaccumulation in the food chain can be highly harmful. It severely affects plant growth and development by disturbing photosynthesis, carbohydrate metabolism, sulphate assimilation, and several enzyme activities (Sanità di Toppi and Gabbrielli 1999). The most typical symptom of Cd toxicity is leaf chlorosis, due to inhibited chlorophyll biosynthesis (Sanità di Toppi and Gabbrielli 1999). Cadmium also induces an oxidative challenge resulting in increased production of ROS, which trigger non-specific oxidation of proteins, lipid membranes, and nucleic acids (Sanità di Toppi and Gabbrielli 1999). Because roots are the first part of the plant affected by soil Cd, a drop in water and nutrient acquisition is also one of the first plant physiological changes. Furthermore, Cd interferes with the uptake of essential metals (Fe^{2+} , Cu^{2+} , Ca^{2+} , and Zn^{2+}), due to competition phenomena for their transporters (Connolly, Fett, and Guerinot 2002; Astolfi et al. 2012), and of nitrate by inhibiting anion uptake and nitrate reductase activity (Rizzardo et al. 2012).

Considering the strategies adopted at the field level to limit the toxic effects on plants and/or the entry of Cd into the food chain, it is interesting to note that several studies indicate a fundamental role of sulfur (S) metabolism in Cd homeostasis and detoxification in plants. In fact, the S-containing amino acid cysteine is the precursor of both phytochelatins and glutathione, which are involved in Cd chelation and antioxidant response, respectively (Sanità di Toppi and Gabrielli 1999). A role for glucosinolates, sulfur-rich secondary metabolites, in Cd detoxification was also suggested (Durenne, et al. 2018). Thus, ensuring an adequate supply of S in the growing medium is likely to reduce Cd stress. Furthermore, it was recently

suggested that potassium (K) could also mitigate Cd toxicity. Potassium can enhance Cd adsorption on the cell wall of root surfaces by increasing hemicellulose biosynthesis and also increase the expression of Cd detoxification related genes (Wu et al. 2020).

Cadmium toxicity, by causing DNA damage and cell wall alteration, impairs plant vegetative and reproductive growth. Therefore, research aimed at unravelling the mechanisms via which Cd exposure affects plant growth and development is crucial to enhance crop performance. Furthermore, considering the detrimental effect of Cd on pollen tube morphology and seed germination, the transgenerational effects involved in plant adaptation to Cd exposure represent a further research area useful for selecting tolerant plants to Cd contaminated soils. Cadmium toxicity can be alleviated using soil amendments (i.e., biochar and silicon-based fertilisers) as well as by inoculation with specific plant-associated bacteria (Bhat et al. 2019; Gutsch et al. 2019). Such observations emphasise the importance of using future research efforts to examine both plant traits and the plant microbiome in coping with soil heavy metal decontamination.

The case study of arsenic (As)

Arsenic is a metalloid present in soil under different oxidation states between -III and +V. Its concentrations range between 0.1 and 55 mg kg⁻¹, with an estimated global mean between 5 and 7.5 mg kg⁻¹ (Matschullat 2000). Background values exceeding several hundred mg kg⁻¹ are linked to the abundance of shales, phosphates, and clays in soil parent material. Arsenic soil pollution is principally caused by the historical presence of mining and smelting activities, reaching values up to 20,000 mg kg⁻¹ (Smith, Naidu, and Alston 1998). Irrigation with As-polluted waters, phosphatic fertilisation, use of As-based pesticides (nowadays banned in several countries), and biosolids amendment are the main causes of pollution of agricultural soils. The paddy soils of Bangladesh, India, and other Asian countries have As concentrations significantly higher than background values due to irrigation with heavily As-polluted groundwater (Khanam et al. 2020). The trend of As pollution in the agricultural soils of China has gradually increased in the last three decades (Gong et al. 2020), and 0.8% of European farmlands suffer from As pollution (Tóth et al. 2016).

Arsenic is scarcely mobile in soil and possesses an estimated retention time of 1000-3000 years (Matschullat 2000), thus it tends to accumulate, especially in the topsoil. Most As is associated with soil solid phases, principally adsorbed on mineral surfaces and/or coprecipitated with secondary minerals, such as Fe-(oxy)hydroxides in aerobic soils and

sulphides in anaerobic soils (Wenzel 2013). The main process affecting the solid-liquid partitioning of As is its specific adsorption on Fe and Al (oxy)hydroxides by ligand exchange reactions, which are favoured by the low crystallinity and higher specific surface area of the oxides' structures. In Fe-poor soils, adsorption involves mainly Al(III) and Mn(III/IV) (oxy)hydroxides, as well as clay minerals and OM. The latter, however, plays a controversial role, since it can also enhance As solubility by occupying sorption sites on Fe-(oxy)hydroxide surfaces (Wenzel 2013). Layered double hydroxides (LDH) are also effective in reducing As availability (Sommella et al. 2015). The inorganic oxyanions arsenite [As(III)] and arsenate [As(V)] prevail in the soil solution, with the first form more soluble, mobile, and toxic than the latter. The methylated forms monomethylarsonic acid [MMA(V)], dimethylarsinic acid [DMA(V)], and trimethylarsine oxide (TMAO) are soluble forms formed by bacteria and fungi metabolism. Arsenic speciation in the soil solution is mainly affected by soil Eh and pH: arsenous acid (H_3AsO_3) occurs in reduced soils, whereas arsenic acid (H_3AsO_4) and its deprotonated forms prevail under oxidizing conditions. Soil fertilisation with phosphates can enhance As solubility since they compete with sorption sites on soil organic and inorganic phases (Violante and Pigna 2002).

Arsenic is taken up by roots in its inorganic As(III) and As(V) oxyanion forms thanks to the activity of different protein transporters. Once acquired by roots, the As(V) form is rapidly reduced to As(III) and complexed with phytochelatins, and therefore compartmentalised in the vacuoles. However, it is also possible that both As(V) and As(III) are loaded into the xylem vessels and then translocated to the shoot (Table 10.1). Despite representing a minor fraction of As in soil, MMA and DMA can also be acquired and readily translocated into the xylem vessels.

Arsenic concentration and speciation in soil determine its toxicity to plants. In some plant species, organic forms as MMA and DMA have been suggested to be more toxic than the inorganic arsenate AsO_4^{3-} and arsenite AsO_3^{3-} . For instance, in rice the order of toxicity is $\text{MMA} > \text{arsenite} > \text{arsenate} > \text{DMA}$ (Marin, Masscheleyn, and Patrick 1992). Furthermore, interconversion between As species might be triggered within the rhizosphere due to continuously changing chemical, biochemical, and microbiological interactions.

Being a redox active metalloid, As inhibits plant growth by inducing oxidative stress (i.e., accumulation of ROS and malondialdehyde - MDA), resulting in the degradation of important biomolecules like lipids, nucleic acids, and proteins (Tripathi et al. 2012). Furthermore, As binds to sulfhydryl (-SH) groups of enzymes and proteins, hindering several important

cellular functions (Meharg and Hartley-Whitaker 2002). Finally, As(V) mimics phosphate, preventing oxidative phosphorylation and ATP production (Tripathi et al. 2007). The consequent impact on crop yield is plant species and tissue concentration dependent. For instance, tomato is naturally resistant to As and avoids its accumulation in plant tissues (Hettick et al. 2015), while rice accumulates high As concentrations, posing great hazards to human health (Williams et al. 2007). Furthermore, very low As tissue concentrations (up to 4 mg kg⁻¹) reduce the yield of plants like soybean or cotton, while more tolerant species, like rice, show an inhibited yield only with As concentrations around 100 mg kg⁻¹ in aerial biomass and 1000 mg kg⁻¹ in roots. Arsenic tolerance and stress response also differ for each As species. For example, MMA and DMA have been suggested to be responsible for the As-associated-straighthead disease, a physiological disorder affecting rice (Rahman et al. 2008).

From an agronomical point of view, intermittent irrigation in crops like rice could limit As dissolution and, thus, the As soil-to-plant transfer, without impairing crop yield compared to constantly flooded rice fields (Hu et al. 2013). The application of Fe or silica (Si)-based soil amendments has also been shown to reduce the As plant uptake (Meharg 2004; Punshon et al. 2017). Iron oxides limit As bioavailability, particularly in soils with low pH values and low phosphate concentration, mainly thanks to the enhanced sorption sites and reduced arsenite/phosphate competition (Punshon et al. 2017). Silica-based amendments mitigate As crop uptake by competitive inhibition, besides supplying an essential nutrient in Si depleted soils (Penido, et al. 2016). The limited sustainability of this practice due to its high cost might be overcome by exploiting Si-rich harvest residues such as rice husk. Furthermore, low As accumulating cultivars might be selected by traditional breeding techniques, gene editing or alterations in specific gene expressions fostering also plant-microbiome interactions that both limit As plant uptake and improve directly and/or indirectly the plant health (Fitz and Wenzel 2003). Finally, plants have developed several mechanisms to cope with As toxicity, including phytochelatins-based detoxification (Tripathi et al. 2012). The activation of phytochelatin production is finely coordinated with thiol metabolism and, thus, highly dependent on increased sulfate uptake and reduction pathways (Rausch and Wachter 2005), as reported in rice (Tripathi et al. 2012). Thus, S fertilisation could play an important role in controlling As toxicity in plants.

Future research efforts should be aimed at understanding the effects of As on lipid, protein, and DNA metabolism. Several studies have suggested that the exogenous application of proline, salicylic acid, and inorganic phosphate might reduce As stress symptoms in hydroponically-grown

plants (Abbas et al. 2018). In this context, the development of strategies aimed at reducing toxicity symptoms, as well as As uptake and translocation to the edible parts of soil-grown agricultural crops, should be investigated.

The case study of chromium (Cr)

Chromium is a PTE that occurs naturally in soil as a result of bedrock weathering, at concentrations ranging between 0.5 and 250 mg kg⁻¹ (Gonnelli and Renella 2013). Higher Cr concentrations characterise silty and loamy soils (from 30 to 1100 mg Cr kg⁻¹; Kabata-Pendias and Mukherjee 2007), and soils originated from mafic and ultramafic rocks. Chromium of anthropogenic origin is derived from the discharge of wastes from the metallurgical, industrial, mining, and refractory industries. Concentrations up to 25,573 mg Cr kg⁻¹ have been found in polluted soils located in mining and industrial districts (Dhal et al. 2013). In agricultural soils, Cr-pollution is principally linked to soil amendment with sewage sludge and irrigation with wastewaters. Values up to 3142 mg kg⁻¹ of total Cr and 87 mg kg⁻¹ of Cr(VI) were detected in agricultural soils irrigated with tannery effluents (Paul et al. 2015), and total Cr concentrations up to 5160 mg kg⁻¹ were measured in soils amended with tannery sludge (Gattullo et al. 2020). Chromium pollution affects 1.1% of European agricultural soils (on 22,000 sites investigated; Tóth et al. 2016) and 4.3% of Chinese farmlands (on 1625 sites; Li et al. 2020).

Chromium hazardousness depends on its speciation between the trivalent [Cr(III)] and hexavalent [Cr(VI)] form. While Cr(III) is slightly toxic and scarcely mobile in soil, Cr(VI) is highly toxic to living organisms, mobile, and bioavailable. Oxidation-reduction reactions driving the complex Cr biogeochemistry may take place simultaneously and are affected by soil pH, Eh, OM content, and the availability of Mn and Fe (oxy)hydroxides (Bartlett 1997). Chromium is mostly present in the solid phases, such as in primary minerals (i.e., chromite), secondary minerals (as a substituent of the octahedrally coordinated Al), precipitated forms (mainly Cr(OH)₃ in neutral-alkaline soils), adsorbed, and complexed forms. In addition to Cr(OH)₃, Cr(III) exists as Cr(OH)²⁺ and other hydrolytic products adsorbed on clay minerals and OM, with adsorption strength increasing with increasing soil pH (Gonnelli and Renella 2013). Moreover, Cr(III) may be complexed by organic and inorganic ligands. Complexes with low molecular weight organic molecules usually enhance Cr(III) solubility. Hexavalent Cr, occurring in soil as an oxyanion (CrO₄²⁻ and Cr₂O₇²⁻), can be partially adsorbed on Fe and Al (oxy)hydroxides and clay minerals by ligand exchange. Nevertheless, under both acidic and alkaline

conditions, Cr(VI) is highly soluble and easily bioavailable. Soil OM reduces Cr(VI) to Cr(III), therefore in soils rich in OM Cr(III) is quite stable and its reoxidation to Cr(VI) is very limited (Gattullo et al. 2020).

Root acquisition of both Cr(III) and Cr(VI) has been clearly demonstrated. The uptake of Cr(III) is a passive mechanism, whereas that of Cr(VI) is an active process (Table 10.1). Among the heavy metals, Cr is reported to be the least mobile in plants, although a limited fraction of Cr(VI) can be also allocated to the shoot (Table 10.1).

Chromium is highly toxic to plants in both its oxidation states [Cr(III) and Cr(VI)]. Except for Cr-hyperaccumulating plant species, Cr is poorly translocated to the shoots and is mainly adsorbed in the root tissues (Shahid et al. 2017). Nonetheless, Cr induces severe toxicity symptoms in plants at a morphological, biochemical, molecular, and physiological level, inhibiting plant growth and development and inducing complete cell damage (Ertani et al. 2017). Several factors might cause Cr-induced plant growth deficiency and, consequently, limit crop yield: decreased water and unbalanced nutrient uptake, impairment of photosynthesis, reduced cell division, displacement of essential elements from physiological binding sites, enhanced generation of ROS with consequent oxidative stress, inhibition of enzymatic activities, lipid peroxidation, protein oxidation, and mutagenesis (Sharma et al. 2020). It is important to highlight that Cr toxicity is highly plant species and concentration dependent, with observed yield impairment in the micromolar to millimolar range (Sharma et al. 2020).

As for phytoremediation of Cr-polluted soils, since the translocation of Cr is rather limited, some applications are aimed at triggering Cr plant uptake by the addition of organic acids (e.g., citric, oxalic, aspartic) (Ertani, et al. 2017), biological soil amendments (e.g., arbuscular mycorrhizal fungi – AMF) (Singh et al. 2013), or by limiting the sulfate concentration in the soil solution (Schiavon et al. 2008). Depending on the fungal species and the host plant, AMF can enhance plant resilience to Cr stress by either enhancing Cr uptake or by inhibiting Cr accumulation in plant roots, thus protecting the plant against Cr toxicity. In grasses like wheat, barley and rice, Si supply was able to counteract Cr toxicity by reducing oxidative damage and root-to-shoot translocation, yet without hindering Cr accumulation at the root level (Sarkar et al. 2020).

Although some data are available, the mechanisms determining Cr toxicity at the protein and molecular level are still unclear. Indeed, the acquisition of knowledge concerning Cr tolerance and homeostasis will be fundamental in order to identify possible strategies to limit root acquisition of Cr, as well as contain its toxic effects in crops once inside the plant. These aspects are relevant in the context of more sustainable crop production and food safety.

Conclusions

Soil contamination by PTEs can be a serious issue in agricultural areas since, besides damaging the environment, it can reduce both crop yield and quality. A deeper knowledge of the biogeochemistry of these elements in soil, and specifically, at the soil-microorganism-root interface, is essential to develop sustainable agronomic strategies able to (i) reduce the supply of PTEs to the soil, (ii) reduce plant uptake and translocation without compromising nutrient acquisition, and (iii) promote plant defense mechanisms against toxic elements. Greater attention should be given to organic amendment and phosphate fertilisation since, depending on how they are managed, they can enhance or reduce element toxicities for crops. Organic amendments are often used to immobilise hazardous metals by adsorption or, in the case of Cr, to favour the formation of the less harmful reduced form, Cr(III). Nevertheless, one of the main causes of unintentional soil contamination can be the soil application of PTE-rich organic materials, therefore only high-quality organic amendments must be used. Phosphate fertilisation should be used cautiously in soils polluted with As and Cr, in order to avoid the solubilisation of these elements, and superphosphate use should be in any case avoided due to its high content of Zn, Cd, and other PTEs. Conversely, phosphate fertilisation can reduce the availability of Cd, Zn, Pb, and other dangerous metals. A limited use of PTE-containing pesticides is also advisable to reduce the risks of soil contamination. Additionally, a balanced nutrient supply is desirable to improve plant-defence mechanisms against toxic metals (particularly S supply, which is essential for the synthesis of metal chelators, like phytochelatins and glutathione, by plants). The selection of metal tolerant/resistant crops and microorganisms able to protect plants from metal toxicity is also required. The coexistence in contaminated farmlands of more than one PTE (also including Pb, Ni, and Hg), as well as organic pollutants, often makes management highly complicated and, therefore, additional research efforts in this area are needed.

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PART D:

BIOLOGICAL CONSTRAINTS

CHAPTER 11

SOILBORNE DISEASES

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Abstract

Reduced crop performance, increased production cost, and potential damage to the natural environment are significant problems caused by soilborne pathogens in agricultural production worldwide. In many parts of the world, methyl bromide was used extensively to control these pathogens before it was banned. This led to the development and use of site-specific chemicals. However, even with chemical treatments, pathogens can be difficult to control due to their production of resting structures (chlamydospores, microsclerotia, oospores, sclerotia, or nematode cysts) and the development of chemical resistance. This means alternative and integrated approaches are required to reduce economic damage. This chapter will discuss the currently available disease management options, their advantages and disadvantages, and how they can be effectively integrated. The approaches examined will include sanitation, legal methods, soil solarisation, anaerobic soil disinfestation, soil amendments, plant nutrients, use of resistant cultivars and grafting, and chemical and biological methods.

Keywords: Soilborne diseases, disease management methods, chemical control, biological control.

Common soilborne diseases

Soilborne diseases are considered one of the major limitations to crop production. Soilborne plant pathogens like *Fusarium* spp., *Phytophthora*

spp., *Pythium* spp., *Rhizoctonia* spp., *Sclerotinia* spp., *Verticillium* spp., and nematodes are major pathogens of concern. These soilborne pathogens can cause more than 50-75% yield loss for many crops such as wheat, cotton, maize, vegetables, fruit, and ornamentals (Lewis and Papavizas 1991; Panth, Hassler, and Baysal-Gurel 2020). In the United States alone, soilborne diseases are responsible for about 90% of the major diseases of principal crops (Lewis and Papavizas 1991; Mokhtar and El-Mougy 2014).

Soilborne pathogens generally survive for a long period of time on host plant debris, soil organic matter, or as free-living organisms. Pathogens can also form resistant structures like microsclerotia, sclerotia, chlamydospore, or oospores, and nematode eggs, cysts, or juveniles can persist in soil for extended periods. An interesting feature of soilborne diseases is they are difficult to identify and control. Generally, by the time aboveground plant parts start to show symptoms, the underground part is already heavily compromised and control measures will either be late or ineffective. In addition, similarities in the symptoms of many soilborne pathogens (blackening of roots and crown, root rot, damping-off of seedling, stunting, wilting, yellowing, twig or branch dieback, bark cracking, and defoliation) make them difficult to diagnose (Astrom and Gerhardson 1988). All these factors make soilborne diseases a major limitation to crop production.

Conventionally, synthetic chemical fungicides or nematicides with a short interval are used to control outbreaks. However, the negative effect of these synthetic chemicals on ecology, human health, aquatic life systems, and beneficial microorganisms cannot be ignored. Consequently, significant work has been undertaken to identify how chemical control methods can be more safely and effectively used to control soilborne diseases and combined with (or replaced by) alternative and more ecologically friendly management approaches. Chellemi et al. (2016) proposed the system-based approach of disease management, which focus on avoidance, exclusion, and management of soilborne pathogens using natural systems. This approach combines the practices that promote natural soil disease suppressiveness with integrated disease management options. However, with all disease management strategies, regular monitoring of the crop is necessary to identify effective control options.

Soilborne disease management strategies

Advances in disease management in recent years have shown that the active and integrated management of soil microbial communities could be an effective way to naturally suppress soilborne plant pathogens while improving crop production. The logical and careful inclusion of chemical

methods along with other available techniques would provide better disease management. Here, we will summarise the existing techniques of soilborne disease management.

Chemical control

Chemical control methods are effective for controlling soilborne plant pathogens in agricultural crops. Non-chemical options can be slow and less effective against very aggressive soilborne pathogens. Therefore, growers are attracted to chemical options. The use of chemical control methods is especially preferred among larger producers because of their rapid effect and ease of use. Labrada (2007) argued that non-chemical alternatives are not currently sufficient to fulfil grower needs, which may lead them to search for chemical alternatives. The chemicals from dicarboximide, benzimidazole, and triazole are known to effectively control many soilborne pathogens (Matheron and Porchas 2004; Budge and Whipps 2001). For example, Bubici et al. (2006) reported the use of strobilurins for the effective control of *Verticillium* wilt of eggplant in greenhouse and field conditions. Similarly, cyprodinil and fludioxonil based fungicides were also used against *S. sclerotiorum*. Baysal-Gurel and Kabir (2019) used pyraclostrobin, oxathiapiprolin, and mefenoxam for the effective management of *P. nicotianae* of hydrangeas. Panth et al. (2021) used strobilurin, oxathiapiprolin, and mefenoxam for the effective management of oomycete pathogen, *Phytophthora vexans*. Similarly, chemical nematicides have been used singularly or in combination with other chemicals since the late 19th century. Talavera-Rubia et al. (2020) used milbemectin to effectively control root-knot nematodes by reducing egg hatching and immobilising juveniles.

While chemical methods of disease control are easy, quick, and effective, they can disturb human health and environmental systems, damage aquatic life, harm pollinators, and may even reduce beneficial microorganisms in the soil (Panth, Hassler, and Baysal-Gurel 2020). Fungicides/nematicides of dicarboximide, benzimidazole, chlorpicrin, azoxystrobin, metalaxyl, dimethyl sulfoxide, 2,3-D, and triazole groups have been used to control soilborne diseases caused by fungi, oomycetes, or nematodes. However, the effect of these chemicals on crop productivity, microbial community, and resistance development needs to be addressed (Panth, Hassler, and Baysal-Gurel 2020; Wang et al. 2006). The development of resistance against chemicals is also a major concern. The random or repetitive use of chemicals of the same Fungicide Resistance Action Committee (FRAC) code leads to pathogen resistance. Therefore, FRAC recommend the

rotation of chemicals, adherence to the recommended dose, and use of biorational products, which are effective and less harmful to the environment, to combat these pathogens (Brent et al. 1998). The use of host plant defence inducers to develop immunity against the disease is a promising option for the future. Similarly, the use of biocontrol agents such as *Bacillus*, *Coniothyrium*, *Gliocladium*, *Pseudomonas*, *Rhizobium*, *Serratia*, *Streptomyces*, and *Trichoderma* are being put into use (Panth et al. 2020). Soil physical and chemical properties, biochemical processes, and the sustainability of the environment will always be in danger from the long-term application of chemicals.

Generally, copper and sulfur-based fungicides are effective against a wide range of diseases caused by fungi, however, they must be reapplied often. Carbon based organic fungicides are very specific in their mode of action, so proper diagnosis of the disease is important. Chemical pesticides can be used to kill alternate hosts to manage polycyclic disease. As the overuse of chemicals can develop chemical resistance in pathogens, a rotation of chemical options is recommended. While there is no best fungicide available for any disease, effective application of selected fungicides can help reduce disease severity. Application of host plant defence inducers, a good number of fertilisers, and biological control agents, in combination with less harmful chemical options, can also be effective choices. The use of chemical options during the sensitive growing period when the chance of pathogen attack is highest, followed by integrated disease management options during the rest of the growth period is most desirable. Generally, chemical fungicides should be used as a last resort. However, further research is required to effectively develop a model to use chemical options more effectively, in combination with other disease management options, to reduce the negative impacts of chemicals.

Biological control

Biocontrol agents suppress soilborne plant pathogens through parasitism, production of antagonistic chemical compounds, competition for nutrients, or providing resistance to the plants against the pathogens (Shafique et al. 2016). Some commonly employed biological control agents are *Bacillus*, *Trichoderma*, *Pseudomonas*, *Rhizobium*, *Streptomyces*, *Gliocladium*, and *Coniothyrium* (Mazzola and Freilich 2017). *Trichoderma* species are known to produce fungi-toxic chemicals, which can be used against soilborne pathogens including nematodes. *Trichoderma* species have been used to control root-knot nematodes on diverse crops such as tomato, okra, mung bean, and bell pepper (Meyer et al. 2001). Mycoparasite

C. minitans was used to effectively manage *S. sclerotium*. *C. minitans* produces cell wall degrading enzymes such as chitinases and glucanases and restrict the colonisation of *S. sclerotium* (Tomprefa et al. 2011; Smolińska and Kowalska 2018). Bacterium *B. amyloliquefaciens* can produce antifungal substances, which can inhibit the mycelial growth of some plant pathogenic fungi such as *Botrytis cinera*, *S. sclerotium*, *P. grisea*, *Penicillium digitatum*, *Alternaria panax*, and *C. orbiculare* (Ji et al. 2013; Panth, Hassler, and Baysal-Gurel 2020).

Not only antagonistic fungi and bacteria, but also arbuscular mycorrhizal fungi can be used to control soilborne diseases. These fungi cover the root by forming a mat and protect the plant against pathogens. The most common type of mycorrhizal association, vesicular arbuscular endomycorrhizas, were found to be effective in protecting plants against *Rhizoctonia* in several plants, *Verticillium* in cotton, and *Fusarium* and *P. nicotianae* on tomato (Liu and Liu 1995; Cordier, Gianinazzi, and Gianinazzi-Pearson 1996).

In general, the development of disease in any host is the exception. The three components of disease development (susceptible host, virulent pathogen, and favourable environment) need to be perfectly matched to produce disease. The use of chemical fungicides or nematicides generally targets a specific population of microbes in the soil, which unbalances the natural ecosystem and aids in resistance development. The use of biological control agents not only checks the pathogen proliferation but promotes the ecological health by promoting a healthy soil microbial population.

Sanitation

Sanitation of working areas and tools can be the first step towards avoiding the introduction of disease-causing pathogens to new host plants. Plant debris and soil organic matter can provide shelter to pathogens or their survival structures. Therefore, it is critical to remove infected plant debris from growing areas to avoid pathogens. The spread of pathogens can be further prevented by disinfecting tools and equipment, as well as washing hands (Baysal-Gurel, Gardener, and Miller 2012).

Soil tillage can also be used to expose the soil to the sunlight and incorporate plant debris into the soil to amplify decomposition. This can help to reduce disease inoculum levels to a certain extent. Farmers who use minimum tillage leave crop residues undisturbed, which can help to minimise soil erosion, increase soil moisture, and may increase yield. However, those crop residues can act as a home for pathogen survival structures (Bockus and Shroyer 1998). Therefore, additional control

methods are recommended for farmers practicing zero or reduced tillage. Working with plants when they are wet can also favour pathogen persistence and may transfer the pathogen from the diseased plant to a healthy one (Panth, Hassler, and Baysal-Gurel 2020). This can be managed by working during the daytime when the plant canopy is not too wet. Infected plant parts can also be removed in the early stages of infection to avoid further disease development in some crops. Sanitation of fields and/or equipment is an easy but important method which, within an integrated disease management approach, can prove beneficial.

Legal methods/quarantine

This involves legislative control to ban the transportation of plants or plant materials to stop pathogen spread. For example, the first U.S quarantine law was passed in 1912 to prevent international travellers bringing fresh plant material into the states to prevent overseas pathogen introduction. Local quarantine is also sometimes established to prevent the spread of pathogens across state boundaries. The introduction of plant pathogens can remain below economic threshold levels but then suddenly emerge as an epidemic (Crooks 2005), which can be minimised with effective quarantine methods. However, even with quarantine, pathogen spores can travel a long way through air or water to contaminate agricultural land across states or even countries. Pathogens can also be transported by people or food.

Resistant cultivars/varieties and grafting

The use of disease resistant cultivars or varieties can also be an effective way to avoid the heavy loss caused by pathogens. Propagative plant parts like corms, bulbs, and tubers are very likely to carry soilborne pathogens on their surface. Thus, careful measures should be applied to prevent contamination from mother stock to the new plants. Grafting with disease resistant rootstock is the traditional practice that has been widely adopted with many crops. The successful use of resistant rootstock in tomato against *F. oxysporum* f. sp. *lycopersici*, *F. oxysporum* f. sp. *radicis-lycopersici*, *Pyrenochaeta lycopersici*, and *Meloidogyne* spp. has been reported (Mihajlović et al. 2017). Corsini and Pavek (1985) successfully used potato clones resistant against *Verticillium* wilt. The use of genetically engineered crops with greater resistance also has potential for improving resistance, however, community acceptance of such crops can limit their use. There are varieties of asparagus, beans, potatoes, cucurbits, spinach, corn, and

tomatoes that are resistant to diseases (NWSIPM 2021). When used with appropriate crop rotation methods and other IPM strategies, the use of resistant cultivars will help reduce the pathogen level long term. Generally, the disease resistance may not be complete, but the disease will be less severe with resistant cultivars.

A few researchers reported the successful management of soilborne diseases such as bacterial wilt, root knot nematode of solanaceous crops, and Fusarium wilt of cucurbits (Bruton 2005; Rivard and Louws 2006). *V. dahliae* is one of the common soilborne plant pathogens, which can be effectively controlled by grafting (King et al. 2008). Also, combining grafting with soil sterilisation improves its reliability (Ioannou 2001). Eggplant is commonly grafted to control Verticillium wilt; and successful control has also been observed in cucumber, watermelon, tomato, and melon (Paplomatas et al. 2002; Lahkim and Nachmias 1995; Bletsos, Thanassouloupoulos, and Roupakias 2003). Similarly, grafting has been proven to be effective against bacterial wilt caused by *Ralstonia solanacearum* in tomato (Grimault and Prior 1994).

Successful grafting with higher compatibility may also increase crop performance, tolerance to abiotic stress, and water and nutrient utilisation, which ultimately improves the yield, quality, and nutritive value of produce (Lee and Oda 2003; Cohen et al. 2007; Rabinowitch and Cohen 2012, Katan 2017; Bertin et al. 2017). This research explains the importance, as well as potential success, of the resistant rootstocks and grafting. Using this method in an integrated approach helps to reduce the damage caused by soilborne pathogens.

Cropping system

Mixed cropping, intercropping, and crop rotation are important cultural practices that help to avoid or reduce pathogen build up in the soil. Mixing the crops with cover crops or some trap crops may also help to reduce pathogen levels. In addition, crop rotation is associated with enhanced soil fertility, better water and nutrition management, as well as erosion control (Ball et al. 2005; Mihajlović et al. 2017).

The biology of the pathogen is important for the management of disease. Crop rotation is particularly useful for those pathogens that overwinter in the field and infect plants in the next growing season. The rotation with non-host crops will starve the pathogen and reduce the inoculum level. However, rotation may not be a feasible option for perennial cropping systems or be useful for pathogens with a wide host range, such as *Phytophthora* spp., *Fusarium* spp., *Botrytis* spp., and root-knot nematode or those producing

resting structures such as sclerotia, chlamydospores, or oospores (Umaerus et al. 1989). Generally, rotations based on the plant family are a better option than individual crops. For instance, tomato should not be rotated with other members of the same family (chili, brinjal, potato), but can be rotated with legumes or brassica crops. Cover crop rotation with main crops such as oat-potato, barley-potato, and clover-potato has been effective in reducing the inoculum and severity of *Rhizoctonia solani* (Larkin, Griffin, and Honeycutt 2010) as well as lower nematode recovery (Johnston et al. 1994). The intercropping of cover crops also helped in reducing root rot disease severity caused by *P. nicotianae*, *Phytophthium vexans*, and *R. solani* in woody ornamental production systems (Panth et al. 2020) (Figure 11.1).

Pathogens that are soil inhabitants thrive well in the soil with or without their host. So, the management of those pathogens becomes difficult. However, soil invaders can be managed through crop rotation as their population decreases rapidly in the absence of host plants. These cultural practices are often incommensurate when applied alone, however, they will be of significance in an integrated approach.



Figure 11.1 Intercropping of the cover crops crimson clover (*Trifolium incarnatum* L.) and triticale ($\times$ *Triticosecale* W.) in a woody ornamental production system to manage soilborne diseases.

Soil solarisation

Soil solarisation is the trapping of solar radiation under plastic sheets to create a greenhouse effect and increase the temperature to kill or reduce

pathogens. This can be an effective way to manage the primary inoculum level, as the higher temperature can kill most important pathogens. Soil solarisation can be adopted only in specific regions where sunshine is abundant and the temperature is higher (Baysal-Gurel, Gardener, and Miller 2012).

Soil solarisation changes soil physical and chemical properties and the microbial ecosystem (Panth, Hassler, and Baysal-Gurel 2020). When the solarisation is performed in hot summer months, it increases the temperature sufficiently to destroy many soilborne plant pathogens including *Verticillium dahlia*, *Fusarium* spp., *Sclerotinia* spp., *Agrobacterium tumefaciens*, *Streptomyces scabies*, some nematodes, insects, and even weeds (Katan et al. 1976; Elad, Katan, and Chet 1980; Baysal-Gurel, Gardener, and Miller 2012). Long term application of soil solarisation has also been reported to reduce populations of *F. oxysporum* f.sp. *vasinfectum* (Bennet 2012), while several researchers also reported the sensitivity of *Phytophthora* species to high temperatures from solarisation (Juarez-Palacios et al. 1991; Pinkas, Kariv, and Katan 1984). A complete elimination of *P. cinnamomi* was reported in South Africa after six weeks of soil solarisation (Barbercheck and von Broembsen 1986). Similarly, soil solarisation has been used to reduce the impact of *P. cambivora* in cherry production in Australia (Wicks 1988).

Although soil solarisation can be effective, it is a slow, expensive process for larger areas, and a climate dependent measure, which limits its application. Similarly, some soilborne pathogens, such as root-knot nematodes, are heat tolerant and may not be managed by this method alone. Integration of biofumigation with soil solarisation may help in this case. Cabbage yellow disease caused by *F. oxysporum* f. sp. *conglutinans* was eliminated in plots where brassica residues were incorporated into the soil before solarisation (Ramirez-Villapudua and Munnecke 1987). The sulfur containing fungitoxic volatiles may have helped eliminate disease in this study. Baysal-Gurel, Kabir, and Livanapathirana (2019) successfully used biofumigants such as arugula, mustard green, and turnip in combination with soil solarisation to reduce the Rhizoctonia root rot of flowering cherry.

Biofumigants

Soil fumigation involves the disinfestation of soil by incorporating volatile chemical compounds. Biofumigants are plants that produce chemical compounds that help reduce the inoculum of plant pathogens but are safe for the soil ecosystem. The brassica crops, which include cabbage, broccoli, radish, canola, rapeseed, cauliflower, kale, turnip, and other

mustards, contain sulfur compounds that can be effectively used to control soilborne diseases. These sulfur compounds are known as glucosinulates, and release the biologically active compound “isothiocyanates,” which is toxic to many soilborne plant pathogens (Larkin and Griffin 2007; Baysal-Gurel, Liyanapathiranaige, and Mullican 2018; Larkin and Honeycutt 2006.). Although the mechanism of disease control is not fully understood, these biochemical compounds may reduce the growth of harmful plant pathogens or create a favourable environment for beneficial microorganisms, resulting in antagonism between the microorganisms (Panth, Hassler, and Baysal-Gurel 2020). Baysal-Gurel, Kabir, and Livanapathiranaige (2019) successfully used brassica cover crops to reduce the disease severity caused by *R. solani*. Auger et al. (2004) reported the use of crushed *Allium* spp. containing thiosulfonates and disulfides as effective pesticidal plants.

The effectiveness of biofumigant cover crops against major soilborne plant pathogens depends upon the type of soil, season, and the climatic conditions in the area. Care must be taken as the biofumigant cover crops can also be phytotoxic in nature (Baysal-Gurel, Liyanapathiranaige, and Mullican 2018). Similarly, they can reduce nutrient and moisture availability to the principal crop and may act as secondary hosts for some diseases. Therefore, sound cropping practices, intercropping or mixed cropping with other cover crops, and integration with soil solarisation is recommended for better outcomes.

Soil amendments

Soil amendments are generally added into the soil to improve productivity; however, they can also improve disease suppressiveness of the soil. The addition of organic amendments can improve soil physical and chemical properties, support beneficial microorganisms, and help control soilborne pathogens. An increase in the population of beneficial microorganisms may increase microbial competitiveness, thus reducing harmful pathogens (Panth, Hassler, and Baysal-Gurel 2020).

Composts and liquids enriched with organic acids, phenolic compounds, and essential oils, which have biocidal properties, could be effective against soilborne diseases (Paret et al. 2010). Although the use of organic amendments for the biocontrol of soilborne diseases is rare, successful control of soilborne pathogens such as *Fusarium* spp., *Phytophthora* spp., *Pythium* spp., *Sclerotium* spp. *R. solani*, *Thielaviopsis basicola*, and *V. dahliae* has been reported with the application of organic amendments (Bonanomi, Antignani, and Scala 2007; Shafique et al. 2016). The use of compost extract rich in *Rhizobacteria*, *Trichoderma*, and *Pseudomonas*

species can also enhance crop production (Welke 2005), and produce plant growth regulators and antagonistic chemicals, such as phenols and tannins, that can control various soilborne plant pathogens. Similarly, biochar can be incorporated to improve disease suppressiveness. Several studies have shown that the incorporation of biochar improves soil pH, increases cation exchange capacity, increases nutrient and moisture retention, and promotes the growth of beneficial organisms (Panth, Hassler, and Baysal-Gurel 2020). Brtnicky et al. (2019) reported an improvement in beneficial microorganisms when biochar was applied with organic manure. Similarly, Jaiswal et al. (2019) also reported that biochar is an effective method to reduce disease severity in cucumber seedlings caused by *P. aphanidermatum*.

The application of organic amendments to improve the disease suppressiveness of soil is promising. However, care must be taken to avoid phytotoxicity and harbouring plant pathogens in undecomposed plant materials. Improving the efficacy of organic amendments by integrating them with other disease management options such as solarisation, crop rotation, or biofortifying the organic amendments with beneficial organisms to improve the rate of decomposition will be valuable future research areas.

Anaerobic soil disinfestation

The addition of easily decomposable materials into the soil to make it anaerobic and control pathogens is known as anaerobic soil disinfestation. The materials that are generally used for anaerobic soil disinfestation are rice bran, fresh crop residues, soybean flour, and vinasses (de Cara 2015). The soil amendments used here will provide the base for rapid microbial growth. At the same time, irrigation will be provided, followed by plastic mulch, which creates anaerobic decomposition. This anaerobic decomposition leads to the formation of many toxic chemicals, which will directly impact the pathogens in the soil. Some of those chemicals include acetic acid, butyric acid, and other volatiles (Panth et al. 2020). Anaerobic soil disinfestation was mainly found to be effective against *Verticillium* wilt (*V. dahliae*), bacterial wilt (*R. solanacearum*), *A. tumefaciens*, *P. ultimum*, and root-knot nematodes (*Meloidogyne* spp.) (Shrestha, Wszelaki, and Butler 2014; Fernando et al. 2019; Poret-Peterson et al. 2020). Song et al. (2020) used maltose as an organic amendment, and they sealed it with totally impermeable film to create anaerobic decomposition, which suppressed >96% of *Fusarium* spp. and >91% of *Phytophthora* spp. The carbon source provided in the form of organic amendments also increases the beneficial microbes in the soil, which can be helpful to control soilborne pathogens via antagonism. This technique is being commercially used for strawberries

in California, cut flowers, tomatoes, and melons in Japan, and strawberries, asparagus, and tree nurseries in the Netherlands (Panth, Hassler, and Baysal-Gurel 2020).

Anaerobic soil disinfestation is a promising method that can be used alone or in combination with other methods, such as biological control, for better efficiency. However, care must be taken on the number of organic amendments to prevent phytotoxicity. This method is also cost and labour intensive and takes a longer period of time.

Steam heat sterilisation

This is the process of heating the soil (known as soil pasteurisation) with steam to kill pathogens. This method was first used in Germany before we had soil fumigants (Baker and Olsen 1960). Fungi, bacteria, oomycetes, weeds, a few viruses, and even nematodes can be killed using hot steam. This method of soilborne pathogen control was reported to be even better than methyl bromide and chloropicrin (Tanaka et al. 2003). Pathogens such as *S. scabiei*, *Spongospora subterranea*, *Fusarium* spp., *R. solani*, and *Colletotrichum coccodes* were effectively controlled in potato fields using steam application (Afek and Orenstein 2002). Root-knot nematodes were controlled in cut flower production in Florida with steam application and this was as effective as applying methyl bromide (Fennimore et al. 2014).

However, the length, as well as temperature, of disinfection may vary according to the pathogen present. Gutierrez et al. (1997) reported the elimination of *R. solani* by steaming trays of tobacco at 80 °C for 30 minutes. However, Minuto et al. (2004) reported the elimination of *F. oxysporum* and *R. solani* after 3 hours of steam application at 80 °C in sandy-loam soil at 16 cm depth. Even with the high heat, there is a possibility of some fungal survival structures, which may overcome the heat. Integrating this method with some other treatment can be useful. Luvisi et al. (2014) used potassium hydroxide or calcium oxide along with soil steaming to control tobacco mosaic virus.

This method can be very feasible and effective to sterilise indoor potting mixes; however, it is very labour intensive and economically expensive for large fields. Further research on the development of easy and cost-effective tools for steam injection is required.

Soil fertility and plant nutrients

Soil physical and chemical properties, such as structure, integrity, density, porosity, pH, cation exchange capacity, and nutrients such as zinc,

nitrogen, etc. can play a role in disease management. Integrated nutrient management can not only provide better growth and yield of crops, but also better protection or resistance to pathogens. Potassium, phosphorus, magnesium, and other micronutrients are involved in maintaining cell structure and function. A balanced application of nutrients may thus help in reducing the disease severity caused by soilborne pathogens. Although, disease resistance is mostly genetically controlled, the management of nutrients to improve physiological and biological processes will help to combat disease. However, care must be taken while applying the chemical nutrients. For example, while nitrogen is considered an important nutrient for plant vegetative growth, an overdose of nitrogen can increase plant susceptibility to disease. The level of nitrogen negatively affects the physical defences and production of anti-microbial compounds such as phytoalexins (Sun et al. 2020).

There are a number of examples of where appropriate nutrient management has helped with disease control. The application of nitrate nitrogen to basic soil growing tomato helped to control Fusarium wilt (Woltz and Jones 1973). Woltz and Ebgehard (1973) reported that the use of nitrate nitrogen reduced Fusarium wilt on chrysanthemums, king asters, and carnations to the lowest level. Potassium is also involved in reducing disease severity. A higher level of potassium reduced Fusarium wilt in tomato (Foster and Walker 1947) and wilt of cotton (Dick and Tisdale 1938). In addition, calcium increases the structural integrity and resistance of the middle lamella, a component of cell walls. The application of calcium reduced the disease caused by *Cylindrocladium crotalariae*, *F. solani*, *P. myriotylum*, *R. solani*, *S. minor*, and *S. rolfsii* (Huber and Haneklaus 2007). Mayers and Campbell (1985) reported the control of clubroot of crucifers (*Plasmodiophora brassicae*) with the application of lime. Other nutrients such as manganese, boron, and zinc are also very important elements that can minimise disease severity.

Soilless culture

Planting in soilless culture is an effective way of avoiding the soil inhabitants or invaders that come with it. Soil is the abundant growing media that provides support to the plant via nutrients, minerals, water, and anchorage. However, soil is also the home for disease causing fungi, oomycetes, bacteria, and nematodes (Panth, Hassler, and Baysal-Gurel 2020). Growing the plants in soilless media removes the barriers of disease-causing pathogens present in the soil and the production and maintenance of phytosanitary measures in the soilless media is easily achieved. The

higher yield in soilless media with more effective control over growing conditions can make it attractive. For example, Engindeniz and Gul (2009) reported a 25% higher net return from cucumber grown in a perlite and geolite mixture compared to a soil-based cultivation system.

Having said that, instances of *Phytophthora* spp. and *Pythium* spp. in soilless media were previously reported (Favrin, Rahe, and Mauza 1988; Rafin and Tirilly 1995; Hutton and Forsberg 1991). If the temperature is not maintained, and the dissolved oxygen is low, instance of infection with that pathogen is higher. However, filtration of water, UV-treatment, as well as use of disinfectants can provide better phytosanitation. The use of biocontrol agents such as *Pseudomonas* spp., *Burkholderia* spp., *Bacillus* spp., *Penicillium* spp., and *Trichoderma* species can also improve biological interaction in the soilless media (Vallance et al. 2011). This system of growing plants in soilless media is new and is adopted in-vitro and in greenhouse conditions. Further research in the natural field setting is recommended for better optimisation.

Conclusion

There is an increasing trend to integrate different methods of disease management for crop protection. Integrated disease management is the use of all available tactics for the management of diseases, including cultural, physical, legal, biological, and chemical (if urgent) approaches. This method is gaining popularity as the impact on the environment due to harmful pesticides is no longer tolerated. Combining different methods can also produce additive or synergistic effects. The major goal of integrated disease management is to apply those measures that are more efficient, healthier, eco-friendly, and which reduce the number of pesticides used.

Recently, a systems-based approach has also been applied, whereby different disease control practices are combined with appropriate cropping patterns to aid natural disease suppression. This system aims to 1) prevent the introduction and spread of pathogens; 2) reduce pathogen inoculum levels to the point where they can be effectively managed via natural systems; 3) increase soil disease suppressiveness; and 4) integrate management approaches with the life cycle of pathogens and minimise the use of chemicals (Chellemi et al. 2016).

Conventional disease management options will likely continue to dominate due to the prevalence of high input, conventional production systems. The increasing food demand of an ever-increasing population will create an urgent demand for quick and effective disease management options, which is not possible in a quick succession. The use of a system-

based approach to create better disease suppressive soils for the future and research to optimise the non-chemical management options are imperative.

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CHAPTER 12

DECLINE IN SOIL FERTILITY AND BIOLOGICAL DIVERSITY

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Abstract

Soil fertility and biological diversity are in decline across the world's cropping systems, owing to intensive agricultural practices including frequent tillage, application of agrochemicals, and soil nutrient mining. Intensive cropping practices generate significant declines in soil organic matter, which has deleterious effects on soil health through losses of soil structure, organic nutrient pools, and the soil biota that underpin nutrient cycling processes. The consequences of these declines are manifold and necessitate increasing input levels to stem concomitant declines in crop productivity. To halt and reverse soil fertility and biological diversity decline requires adoption of agricultural practices that decrease soil disturbance and increase the quantity and quality of organic matter inputs. Such practices include reduced tillage, cover cropping, residue retention, diversifying crop rotations, crop-pasture rotations (ley pasture), and improved fertiliser management. The impact of these practices is context dependent, and the selection of appropriate practices requires consideration of environmental constraints. Further research is required to unpack the breadth and complexity of soil biodiversity, the ecosystem services provided by soil biodiversity, and how soil functional biodiversity can be manipulated through management. The world's dryland cropping systems are particularly deserving of greater research attention given their global extent and susceptibility to soil fertility decline.

Keywords: Cropping system, soil health, soil management, soil organic matter, soil nutrients, soil biota.

Introduction

The productivity of the world's cropping systems is underpinned by soil fertility, defined as the capacity of a soil to provide crops with essential nutrients. Nutrients are present in soils in both inorganic and organic forms, and their availability to crops is affected by a range of factors including soil type and pH, soil organic matter (SOM) content and composition, and soil water status. Soil organic matter represents the great reservoir of soil nutrients, and it is SOM that is the primary source of nutrients for soil biota. As soil biota decompose SOM, nutrients held within SOM are released into the soil solution in inorganic forms that can be easily taken up by plants. This biological process, known as mineralisation, is fundamental to soil fertility because it determines the availability of inorganic nutrients and consequently the potential productivity of a soil for crop production.

The process of SOM decomposition involves a great diversity of organisms. Detritivores play an important initial role by fragmenting larger units of SOM, such as leaf litter and roots, into smaller units. This increases the accessibility of SOM to the microbial decomposers – bacteria and fungi – that convert SOM into new chemical forms that are readily available for plant uptake. In turn, the increase in microbial decomposers supports greater numbers of microbial grazers that further accelerate the decomposition rate. Crop production often exerts significant direct negative influence on the diversity and functioning of soil biota, as well as indirect influence via negative impacts on SOM content and composition. These influences have important implications for soil fertility and hence feedback loops exist between crop production practices, SOM dynamics and soil biological activity, and crop productivity.

This chapter details the extent of the decline in soil fertility and soil biological diversity in cropping systems and outlines the crop production practices driving these declines. It then discusses the impacts of declining soil fertility and biological diversity on crop productivity and presents key strategies for reversing these trends. Finally, areas requiring future research are identified.

Soil fertility and biological diversity decline in cropping systems

Soil fertility decline

The conversion of native vegetation to conventional cropping, defined by frequent applications of tillage and inorganic fertilisers for the cultivation and harvest of annual crops, has led to significant declines in soil fertility. Globally, soil stocks of carbon (C), nitrogen (N), phosphorus (P), and sulfur (S) have declined by 43%, 42%, 27%, and 33%, respectively, following conversion of native vegetation to conventional cropping (Kopittke et al. 2017). Soils are estimated to have lost ~133 Pg C to the atmosphere due to land-use change for agriculture (Kopittke et al. 2021). These losses are primarily driven by the greater frequency and intensity of soil disturbance in cropping systems compared with native vegetation, which results in increased levels of SOM mineralisation (Grandy and Robertson 2007). In addition, nutrients are routinely exported in agricultural produce and often not replaced in sufficient quantities by fertiliser inputs (Tilman et al. 2002).

Soil organic matter is a crucial building block of soil aggregates, serving to bind and stabilise soil microaggregates (53-250 μm size class) that coalesce into macroaggregates (250-2,000 μm size class) (Tisdall and Oades 1982; Bronick and Lal 2005). Development of soil aggregates results in physical encapsulation of SOM, protecting it from decomposers (Schmidt et al. 2011; de Nijs and Cammeraat 2020). However, accrual of soil macroaggregates occurs over multiple years and macroaggregate persistence is highly sensitive to soil disturbance. Consequently, conversion of previously uncultivated native vegetation to cropping via tillage disrupts soil aggregates and exposes SOM to microbial breakdown, resulting in significant and rapid loss of SOM throughout the plough layer (Grandy and Robertson 2006, 2007). The decline in SOM after conversion to cropping follows a negative exponential curve, with the greatest declines occurring relatively soon (<50 years) after initial conversion, before reaching a relatively steady state (David et al. 2009; Dalal et al. 1991; Dalal et al. 2021). Overall, SOM loss is greatest in the top 0-50 cm depth of soil, i.e., that most disturbed by ploughing, and is affected by both the frequency and intensity of tillage, with mouldboard ploughing generating the greatest and steepest declines compared with reduced and no-till practices (Schwartz et al. 2015; Karlen et al. 2013). Importantly, while no-till has been shown to reduce rates of SOM decline compared with more disturbance-based approaches, it is insufficient by itself to reverse SOM loss (Page et al. 2013; Kopittke et al. 2017).

Once nutrients held within SOM are mineralised into inorganic forms, they are readily incorporated into crop biomass. Once a crop is harvested, a substantial proportion of these nutrients is exported as grain, with the remainder left either in the soil as roots or on the soil surface as crop residues. For example, the grain of a maize crop yielding 12.5 t ha^{-1} can hold 298 kg ha^{-1} of N, 55 kg ha^{-1} of P and 247 kg ha^{-1} of potassium (K) (Hartemink 2006). These nutrients are exported off-site during harvest. If crop residues are either burnt or harvested to feed livestock, nutrient exports increase further.

In soils with sufficient SOM, exported nutrients can be readily re-supplied to the following crop via mineralisation (Hartemink 2006). However, in the absence of adequate nutrient inputs to balance nutrient exports, SOM declines. This pattern of soil fertility decline, known as soil nutrient mining, has been particularly prevalent in subtropical and tropical parts of the world due to chronic under-fertilisation (Vanlauwe et al. 2015; Kintché et al. 2015; Lal 2009). In Australia, nutrient mining by crops is estimated to have removed up to a quarter of the original soil N stock (Angus and Grace 2017). As nutrients such as N, P, and S are mined, SOC is also lost due to the stoichiometric relationships of SOM (Kirkby et al. 2011). This leads to negative impacts on soil aggregation and increased greenhouse gas emissions (Guo et al. 2012).

Soil biological diversity decline

Soils are among the most complex of ecosystems. A single teaspoon of soil from a fertile agricultural field is estimated to contain tens of millions of bacterial cells, tens of kilometres of fungal hyphae, and a diverse array of micro-, meso- and macro-fauna (Stirling et al. 2016). These organisms are connected by complex food web interactions and together generate a multitude of ecosystem services critical for sustainable crop production.

At the base of the soil food web are soil microbes: bacteria and fungi. Soil microbes are the primary decomposers driving the cycling of nutrients between organic and inorganic pools (Bardgett 2005). Above the soil microbes are the micro- and meso-fauna, which includes major organism groups such as the protozoans, tardigrades, and nematodes (microfauna) as well as the collembolans, mites, and larger nematodes (mesofauna) (Coleman, Callahan Jr, and Crossley Jr 2018). Microfauna occur exclusively in water films and feed primarily on bacteria and fungi. This close trophic relationship with soil microbes, and with bacteria in particular, means microfauna can exert a strong positive influence on soil nutrient availability through their excreta (Stirling et al. 2016). Mesofauna predominantly occur

in the topsoil where crop residues are abundant and soil pores are large enough to facilitate ease of movement. Collembolans and mites encompass a variety of trophic strategies, but many are bacterial and fungal feeders. Through their consumption and digestion of microbial biomass, they play a key role in SOM decomposition processes and soil nutrient cycling (Coleman, Callaham Jr, and Crossley Jr 2018). The soil macrofauna comprises larger organism groups visible with the naked eye, such as ants, termites, dung beetles, and earthworms. Many macrofauna are ecosystem engineers that can directly alter soil structure (Coleman, Callaham Jr, and Crossley Jr 2018).

The management of agricultural systems has significant impacts on soil biological diversity (Figure 12.1). Conventional tillage practices, which cause intensive disturbance and bury crop residues, have particularly detrimental effects and can result in distinct shifts in soil community structure (Tsiafouli et al. 2015; Hedlund et al. 2004). Larger-bodied soil organisms with longer generation times (*K*-selected species), such as earthworms and predatory mites, are highly susceptible to soil disturbance and suffer reduced abundance and diversity (Postma-Blaauw et al. 2010, 2012; Briones and Schmidt 2017). Soil fungi, which can develop extensive hyphal networks, are also highly susceptible to disturbance and are well-documented to suffer declines in terms of both biomass and diversity under conventional tillage (Helgason et al. 1998; Bowles et al. 2016; Kabir 2005). In contrast, smaller-bodied soil organisms with rapid generation times and high dispersal capacity (*r*-selected species), such as bacteria, become more dominant (Pianka 1970; Wardle 1995; Verbruggen and Kiers 2010; Hedlund et al. 2004). However, even organism groups that increase in dominance under conventional tillage still suffer from reduced abundance and diversity (Postma-Blaauw et al. 2012, 2010; Strickland and Rousk 2010) due to declining C substrate availability. The overall result of these shifts is a simplified soil food web structure comprised of fewer functional groups with smaller body-sizes (Tsiafouli et al. 2015). The extent of decline is related to the intensity of management and soil disturbance: annual cropping drives the greatest declines, while more extensive systems that incorporate ley pastures can help limit the decline (Williams and Hedlund 2014; Tsiafouli et al. 2015).

Another characteristic of intensive cropping systems is the frequent application of inorganic fertilisers and pesticides. Long-term use of inorganic fertilisers has been found to reduce microbial biomass, richness, diversity, and functioning relative to organic fertilisers (van der Bom et al. 2018; Hartmann et al. 2015; Manoharan et al. 2017). However, soil microbial communities are highly sensitive to changes in soil pH (Rousk et al. 2010;

Geisseler and Scow 2014), and it remains unclear whether changes in soil microbial communities are a direct result of inorganic fertilisers or whether the changes indirectly stem from decreases in soil pH as a consequence of inorganic fertiliser use (Geisseler and Scow 2014; Zhao et al. 2014). Fertiliser application can also influence soil biodiversity through its positive effects on plant growth and biomass production, though this can also reduce soil microbial activity through increased crop consumption of water (Cavigelli, Maul, and Szlavecz 2012).

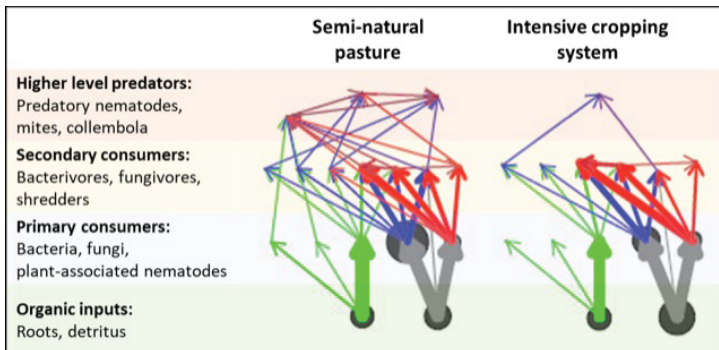


Figure 12.1 Schematic soil food webs showing functional groups (circles) and linkages (arrows) between trophic groups. In semi-natural pastures there is greater diversity of trophic links and more channels through which organic resources flow to higher level predators. In intensive cropping systems, the number and diversity of trophic links is reduced, as is the biomass at each trophic level. Circle size indicates trophic group biomass. Arrow thickness shows the flow of C between groups; green arrows show the root C channel; grey arrows show the detritus C channel; red arrows show the bacterial C channel; blue arrows show the fungal C channel. Data from the SOILSERVICE project (see Tsiafouli et al. 2015 and <https://cordis.europa.eu/project/id/211779/reporting>). Food webs were developed by Elisa Thébault.

Pesticides exert direct and indirect impacts on soil biodiversity. Direct impacts arise when particular organism groups are targeted by an agrochemical, e.g., the application of nematicides leads to significant reductions in total nematode abundance as well as shifts in soil nematode family structure (Chelinho et al. 2011). Indirect effects arise through alterations to soil food webs and SOM dynamics (Turbé et al. 2010). Pesticides can also cause direct non-target effects on soil biodiversity

(Bünemann, Schwenke, and Van Zwieten 2006). For example, application of glyphosate can decrease the abundance of beneficial microbes including arbuscular mycorrhizal fungi (AMF) and rhizobia, the latter resulting in reduced levels of legume root nodulation (Kremer and Means 2009).

Impact of declining soil fertility and biological diversity

Declines in SOM and soil biological diversity have numerous effects on crop production, the most significant of which is the loss of soil health via losses of soil structure, soil nutrients, and the biota that drive soil nutrient cycling processes. Given that SOM and soil organisms are critical components of soil aggregate stability (Zibilske and Bradford 2007; Rillig and Mummey 2006), their loss results in less stable soils with poorer structure (Figure 12.2). Loss of soil aggregates also reduces infiltration and soil water holding capacity (Franzluebbers 2002; Zibilske and Bradford 2007). These factors can negatively impact crop production by increasing the likelihood of soil cultivation producing suboptimal seedbed conditions with poor aeration and a greater probability of water logging (Schjønning et al. 2018), and reducing the capacity of the soils to supply moisture to crops during precipitation shortfalls (Williams, Kane et al. 2016; Williams, Hunter et al. 2016). Moreover, depletion of SOM releases CO₂; this process converts soils from a C sink to a significant C source and is a major contributor to global climate change (Kopittke et al. 2021).

The loss of SOM also affects cropping systems through loss of soil fertility, which reduces the supply of nutrients to crops (Lal 2009). This is particularly noticeable in regions with low inherent soil fertility and a history of soil nutrient mining where long-term crop productivity is strongly linked with SOM status (Kintché et al. 2015; Lal 2009). As SOM declines, there is increasing reliance on inorganic fertiliser inputs to make up nutrient shortfalls to maintain crop productivity (Angus and Grace 2017). However, crops in SOM-depleted, low fertility soils show limited responses to inorganic fertiliser inputs (Kintché et al. 2015; Tittonell and Giller 2013; Brady et al. 2015). This can result in ever increasing inputs of fertilisers in an attempt to close yield gaps, and increases the risks of environmentally damaging nutrient losses (Tilman et al. 2002; Robertson and Vitousek 2009). The SOM-dependency of crop responses to fertiliser is likely partly due to the distribution of available nutrients within the soil profile. Where SOM is distributed throughout the soil profile, mineralisation processes ensure nutrients are available across the entire root system. However, where fertilisers are stratified in the topsoil or concentrated in shallow bands, crop access can be both spatially and temporally limited (van der Bom, Williams,

and Bell 2020). Thus, fertiliser inputs are unable to fully substitute for SOM mineralisation.

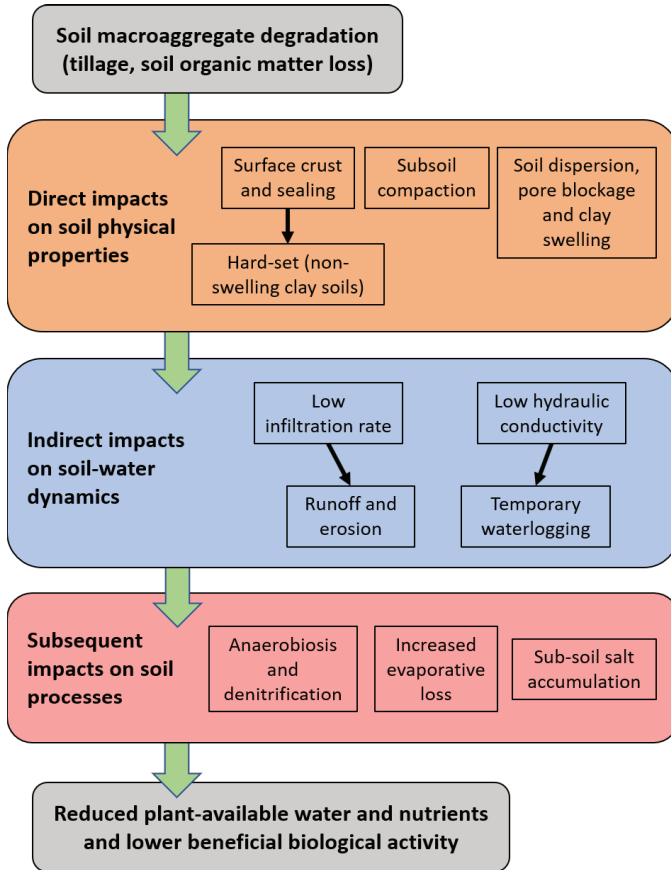


Figure 12.2 Soil aggregate breakdown from soil fertility decline affects interrelated degradative processes (runoff and erosion, nutrient loss, poor air-water relationships, and reduced plant-available water stored in the soil profile).

The loss of soil microbial biomass, diversity and richness in intensively managed cropping systems has consequences for soil nutrient cycling, plant-soil interactions, and crop nutrition and health (Wagg et al. 2014).

Fungal biomass is lower in intensively managed soils subject to repeated tillage and enriched with inorganic fertilisers compared with soils subject to less disturbance and greater inputs of organic nutrients. Although soil fungal biomass remains greater than bacterial biomass in intensively managed systems despite these declines, the relative importance of the bacterial channel increases (Figure 12.1), leading to more rapid nutrient cycling and greater rates of nutrient loss compared with less disturbed soils with greater fungal biomass (Manning 2012; de Vries and Bardgett 2012; van der Bom et al. 2018). This has consequences for global greenhouse gas emissions and pollution of waterways (Robertson and Vitousek 2009; Foley et al. 2011). Moreover, shifts within organism groups due to long-term fertilisation can affect plant-soil interactions and crop nutrition. For example, declines in AMF diversity from long-term (55-years) inorganic N fertilisation is associated with reduced mutualism, where with increasing levels of N fertiliser the crop host has to invest increasing amounts of photosynthate-C for diminishing returns of mycorrhizally-sourced P (Williams et al. 2017). Soil fungi also play an important role in building and maintaining soil structure through the enmeshment of soil particles in hyphal networks and through the production of fungal compounds such as glomalin (Rillig and Mummey 2006).

The loss of larger soil organisms, such as earthworms, also has consequences for soil fertility and crop performance. Earthworms play an important role in soil structure development by generating organo-mineral complexes that contribute to stable soil aggregates, as well as by creating soil macropores that are important for soil aeration and water infiltration through their burrowing behaviour (Bertrand et al. 2015). The action of earthworms also supports soil nutrient cycling processes. Earthworms pull crop residues down into the soil profile, increasing residue-soil contact and facilitating microbial colonisation and decomposition (Bertrand et al. 2015).

Root pathogens become increasingly abundant in intensive cropping systems due to the erosion of self-regulatory ecosystem processes (Stirling et al. 2016). This is a result of declining soil biodiversity due to tillage, the application of agrochemicals, and the preponderance of monocultures and simplified annual crop rotations across much of the world's cropping systems (Geisen, Wall, and van der Putten 2019; Angus et al. 2015). For example, across a range of crop types and environments, continuous cultivation of a single crop over consecutive years leads to accumulation of crop-specific pathogen loads and concomitant crop yield declines (Porter et al. 2001; Li et al. 2014; Blair and Stirling 2007). Even in conservation agriculture systems that combine no-till with stubble retention, the dominance of cereal crops results in a greater incidence of cereal-related

stubble-borne diseases (Flower et al. 2019).

Strategies to build soil fertility and biological diversity

Several management options are available to help stem, and potentially reverse, declines in soil fertility and biological diversity. These options focus on decreasing soil disturbance and increasing the quantity and quality of organic matter inputs, and include practices such as reduced tillage, cover cropping, residue retention, diversifying crop rotations, crop-pasture rotations (ley pasture), and improved fertiliser management. These options should not be viewed as being applied independently. Rather, the appropriate options should be selected based on environmental context and applied in concert as part of a coordinated strategy to build soil health.

Conversion of cropping systems from conventional tillage to reduced tillage or no-till is an important tool for stemming loss of SOM and soil biota (Haddaway et al. 2017; Williams, van der Bom, and Young 2020). Compared with conventional tillage, no-till produces a stable, undisturbed soil environment that allows the development of extensive fungal hyphal networks and diverse communities of micro-, meso-, and macro-fauna (Williams, van der Bom, and Young 2020 and references therein). The increase in fungal biomass is of fundamental importance; fungi tend to exhibit greater C-use efficiency than bacteria, resulting in more efficient resource utilisation and generation of new microbial biomass per unit of CO₂ produced (Kallenbach et al. 2015). This microbial biomass and subsequent necromass contribute positively to the production of mineral-associated SOM that is critical for long-term SOM accumulation (Lavalley, Soong, and Cotrufo 2020). Moreover, soil aggregates and aggregate stability show strong dependence on fungal hyphae, which help enmesh soil particles and produce hyphal exudates (Caesar-TonThat and Cochran 2000; Rillig and Mummey 2006). Consequently, aggregate turnover in no-till systems is reduced compared with conventional tillage systems, resulting in increased formation of stable macroaggregates that can store SOC (Six, Elliot, and Paustian 2000; Somasundaram et al. 2017). For these reasons, no-till systems have been consistently demonstrated to hold greater topsoil concentrations of SOC compared with conventional tillage systems (e.g., Grandy and Robertson 2007; Varvel and Wilhelm 2011).

Despite these potential benefits, numerous studies have demonstrated that if implemented in isolation, the impact of no-till is limited. Implementing no-till in conjunction with other practices, such as crop residue retention, increased cropping frequency, and optimised fertiliser inputs, can improve no-till outcomes in terms of building SOM (Luo, Wang,

and Sun 2010; Alvarez 2005). Interestingly, the SOM-benefits of crop residue retention and optimising fertiliser inputs appear to be inter-dependent. That is, residue retention increases SOM only when adequate fertiliser N is applied and *vice versa* (Alvarez 2005; Luo, Wang, and Sun 2010). This is likely due to the stoichiometric nature of stable SOM, which shows consistent C:N:P:S ratios across a wide range of global soils (Kirkby et al. 2011). The importance of maintaining these ratios through fertiliser additions has been demonstrated under field settings, where for a constant input of C via crop residues, there was a net difference of 8.7 t SOC ha⁻¹ after five years between a treatment that added N, P, and S in conjunction with crop residues and a treatment that provided crop residues with no additional nutrients (Kirkby et al. 2016).

The planting of cover crops – non-harvested crops planted between cash crops – has been shown to have positive effects on soil fertility and soil biodiversity. Across a range of studies from different environments, cover crops have been shown to increase SOM compared with conventional bare fallows (Jian et al. 2020; Poeplau and Don 2015; Williams et al. 2018). By maintaining ground cover, cover crops protect soil and reduce erosion, which is a major pathway of SOM loss (Mohammed et al. 2021; De Baets et al. 2011). Also, cover crops represent a fresh source of organic matter inputs into soils during a traditionally uncropped period. In particular, cover crop roots and their associated rhizodeposits represent significant organic matter inputs that contribute to stable SOM (Austin et al. 2017; Kätterer et al. 2011; Poirier, Roumet, and Munson 2018). Cover crops have also been demonstrated to increase soil N, including within the mineral-associated fraction that is critical for building long-term soil fertility (Jilling et al. 2020). Improvements in soil N stocks and N cycling dynamics are further improved by planting leguminous cover crops (Büchi et al. 2018; Tonitto, David, and Drinkwater 2006). Cover crops can also aid in the recovery of soil biodiversity and biological function (Kim et al. 2020). For example, cover crops have been found to increase AMF root colonisation of subsequent cash crops by ~30% compared with traditional fallowing (Bowles et al. 2016). Such findings have immediate relevance to addressing long fallow disorder in crops with strong mycorrhizal dependency (Thompson 1987). The inclusion of cover crops within a crop rotation can also lead to increased macrofauna density and biomass, e.g., of earthworms, as well as higher densities of predatory and bacterial-feeding nematodes and lower densities of phytophagous nematodes (Blanchart et al. 2006; DuPont, Ferris, and Van Horn 2009).

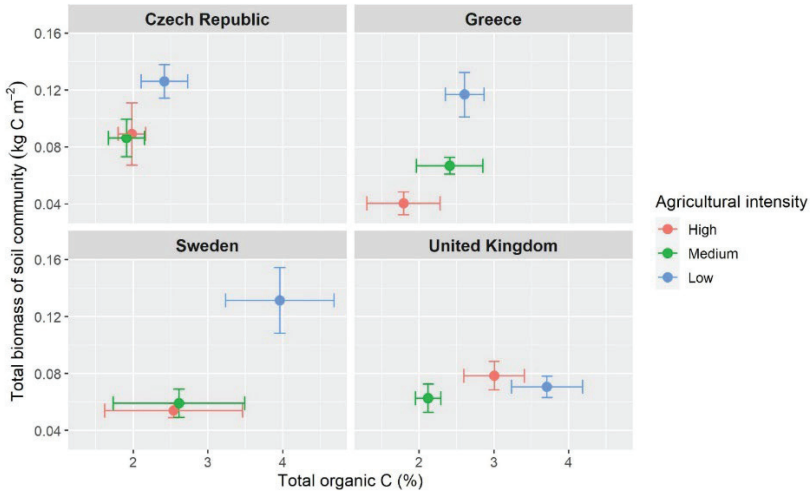


Figure 12.3 Relationship of soil community biomass with SOC and agricultural intensity across Europe. Low intensity agricultural systems (semi-natural pastures) can support greater SOC and soil biology compared with high intensity cropping systems. Medium intensity systems are cropping systems that incorporate a minimum of one-year of ley pasture within a five-year rotation. Error bars show mean $\pm$ 1 SE. Data from the SOILSERVICE project (see Tsiafouli et al. 2015 and <https://cordis.europa.eu/project/id/211779/reporting>).

Historically, crop rotations were much more diverse than today (e.g., Lawes and Gilbert 1894). However, with the advent of highly efficient mechanised agriculture, crop diversity has dwindled to the extent that it is not unusual for a single crop to be grown in a particular field year after year. The absence of crop diversity affects soil biodiversity by increasing the abundance of crop pests and pathogens. However, it also causes overall reductions in soil biodiversity with negative impacts on soil nutrient cycling (Venter, Jacobs, and Hawkins 2016; McDaniel, Tiemann, and Grandy 2014). Across a range of environments, increasing the diversity of crop rotations, e.g., by including three or more crops in a cropping sequence, results in distinct soil microbial communities associated with increased microbial biomass and activity (Tiemann et al. 2015; McDaniel, Tiemann, and Grandy 2014; Williams and Bell 2019). Diverse crop rotations, especially those that include legume crops, also develop greater concentrations of SOC and N compared with conventional low diversity crop rotations (Tiemann et al. 2015; Bell et al. 2017), showing their value for reversing soil fertility

decline. Incorporating ley pastures within crop rotations can also be beneficial by supporting more abundant soil biology and promoting SOM accrual (Figure 12.3) (Tsiafouli et al. 2015). Increasing crop rotational diversity can also lead to subsequent crop yield improvements by helping to suppress crop pathogens (Angus et al. 2015; Peralta et al. 2018), providing the rotation includes crops that do not host the same suite of pathogens (Flower et al. 2019).

Outlook

Soil fertility and biological diversity are strongly interlinked, underpin the concept of soil health, and are critical to the long-term future of the world's cropping systems. Conventional crop management practices are by now well documented to cause severe declines in both. This is untenable given the ever-growing demand for food and feed from a rapidly expanding human population (Tilman et al. 2011; Foley et al. 2011). Consequently, significant changes to how cropping systems are managed are needed to meet the twin challenges of increasing cropping systems productivity and minimising negative impacts on soil fertility and biological diversity. Achieving this will require much greater understanding of soil biodiversity, the ecosystem services provided by soil biodiversity, and how soil functional biodiversity interacts with management interventions (Bender, Wagg, and van der Heijden 2016; Williams, van der Bom, and Young 2020).

Fundamentally, to improve soil fertility and sustain greater soil biological activity, inputs to soil of nutrients in organic matter need to increase beyond what is exported in harvested produce and via loss pathways. Methods such as manure application are widely used in some regions and cropping systems, but the bulk transport of manure is unfeasible at scale for a global cropping solution (Araji, Abdo, and Joyce 2001). Moreover, surface application of manure does not consistently lead to beneficial SOM or soil microbial outcomes (Lai et al. 2017; Keel et al. 2019). Current evidence indicates that plant-derived C inputs, and particularly inputs from roots that undergo efficient microbial processing, are critically important for generation of stable, mineral-associated SOM (Poirier, Roumet, and Munson 2018; Sokol and Bradford 2019; Austin et al. 2017). Recent research has also uncovered the importance of N-rich residues for the formation of new mineral-associated SOM (Cotrufo et al. 2019; Kopittke et al. 2020). These factors point to the importance of maintaining more continuous living plant cover within cropping systems, i.e., reducing the extent and frequency of bare fallow periods, as well as including more crops within rotations with narrower C:N ratios than cereal

crops, e.g., by including more legume and brassica crops. This can be achieved by diversifying crop rotations, including extensifying rotations by planting ley pastures, and by greater planting of cover crops.

The use of cover crops has been widely demonstrated to generate improvements in both soil fertility and soil biological function. However, further research is needed to identify optimal cover crop establishment and termination methods that maximise desired soil improvement outcomes. The appropriate composition of cover crop plantings to generate rapid and sustained improvements in soil fertility and biodiversity is also needed. Recent evidence indicates that careful attention to cover crop functional traits, especially of root traits, is required (Poirier, Roumet, and Munson 2018; Saleem et al. 2020). Additionally, the potential of temporal intensification—increasing the number of crop harvests per year—to generate the same or similar improvements in soil fertility and biodiversity as cover cropping while still producing harvestable crop yield should be explored (Williams, Kane et al. 2016). Furthermore, the vast majority of cover crop research has been conducted in temperate and continental climates with sufficient rainfall. This risks over-generalisation of the applicability of cover crops to all the world's cropping systems. In particular, the feasibility of widespread adoption of cover crops across the world's dryland zone is unknown. Drylands cover ~40% of the world's terrestrial surface, are home to ~2.5 billion people, and are highly susceptible to soil fertility decline; moreover, the global extent of drylands is forecast to rise dramatically by the end of the century due to climate change (Právělie 2016). Consequently, further research on the role of cover crops in dryland cropping systems is urgently needed.

Lastly, growing more root biomass per crop may be a useful approach for enhancing soil functional biodiversity and building SOM (Kell 2012; Paustian et al. 2016). There is currently much research aimed at developing deep-rooting crop genotypes that can access deep stores of soil moisture to improve crop productivity and reliability in rainfall constrained regions (van der Bom, Williams, and Bell 2020). It is possible that crop germplasm that grows larger root systems, i.e., that has wider root:shoot ratios than prevailing crop varieties, may generate greater contributions to SOM through increased root biomass and rhizosphere microbial activity. This is an area greatly deserving of research, as if proven effective its adoption would entail no changes to current cropping practices or infrastructure.

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PART E:

SOIL CONSTRAINT MANAGEMENT

REGIONAL CASE STUDIES

CHAPTER 13

SOIL CONSTRAINTS AND MANAGEMENT PRACTICES: SOUTHERN AFRICA

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Abstract

The soil resources in Southern Africa are diverse due to geological and climatic variations and this determines the nature and extent of constraints posed on crop establishment, growth, yields, and productivity. The principal constraints include moisture stress, soil erosion, compaction, infertility due to depletion and imbalances of nutrients, acidity, salinity and sodicity, loss of organic matter and associated biological processes, soil borne pests and diseases, and contamination by agricultural practices and industrial by-products. Researchers and farmers in the region have developed and adopted management practices and strategies to overcome these constraints and improve crop productivity. These include use of organic and inorganic sources of nutrients, adopting integrated nutrient cycling practices, conservation agriculture, and bioremediation. The areas that need further research and development as a basis for improving sustainable crop production are highlighted.

Keywords: Soil fertility management, nutrient cycling, conservation agriculture, smallholder farmers, agroforestry.

Introduction

Southern Africa is defined as a group of twelve countries including Angola, Botswana, Eswatini, Lesotho, Malawi, Mozambique, Namibia, South Africa, Zambia, and Zimbabwe (Tulley et al. 2015). It is located along the tropic of Capricorn on the continent of Africa and geographically stretches from the northern point of Tanzania to the southern tip of South Africa. The region covers a land mass of some 6.8 million square kilometres with an estimated population in excess of 140 million people (Stringer and Reed 2007). Most people (up to 70%) live in rural areas and depend on subsistence agriculture for their livelihoods (van Rooyen 1997). A large area in the region falls within sub-humid and semi-arid agro ecological zones (Stringer and Reed 2007). Agriculture remains the linchpin of the economies of many countries, offering the most feasible strategy to promote poverty reduction, food security and economic development (CAADP 2003). Smallholder farming forms the main production system in the region, although commercial farming is also present resulting in dualistic and bimodal production systems. Mixtures, intercrops, or polycultures characterise crop production among most small-scale farmers while monoculture predominates in the large-scale commercial farming sector.

The soils of Southern Africa are characterised by considerable variability in terms of their mineralogical, chemical, and physical properties. However, for purposes of this chapter, the major soil groups in the region will be summarised in broad terms to include: Alfisols, Ultisols, Oxisols, Histosols, Andisols, Inceptisols, Aridisols, Mollisols, Histosols, and Vertisols (Table 13.1). Most of these soils have some kind of constraint to crop production, but the nature and magnitude of the constraint varies depending on the type of a particular soil resource base (Du Preez, Kotze, and Van Huyssteen 2018). It is not the purpose of the chapter to detail the properties of each soil group, but to highlight the principal constraints. The chapter begins by providing an overview of the major soil groups of the region and identifies the physical, chemical, and biological constraints that impact crop production. It then appraises the management practices that have been developed and used to manage these constraints in order to improve farm productivity. The chapter concludes by identifying areas that require further research and draws some implications for sustainable agriculture in the region.

Table 13.1 The major soil taxonomic groups used for crop production in Southern Africa and their constraining characteristics

Climatic zone	Soil taxonomic groups	Country of occurrence	Constraining characteristics
Very humid to Humid Forest zone	Lithosols, Ferralsols, Nitosols, Arenosols, Cambisols, Gleysols	Swaziland, Angola, Mozambique, Malawi, Zambia, South Africa	Low CEC, weak retention of bases, fixation of phosphates, deficiencies of Ca, S, and trace elements, subject to waterlogging and flooding during the rainy season, leaching during rainy season, moderate to high risk of erosion, high acidity and Al toxicity
Humid to sub-Humid wooded savannah	Luvisols, Ferralsols, Arenosols, Acrisols, Nitosols, cambisols, Lithosols, Fluvisols, Gleysols, Vertisols, Histosols	Angola, Mozambique, Swaziland, Malawi, South Africa	Moisture stress, sensitive to erosion, high base saturation, low P availability, micro-nutrient deficiency, low aggregate stability and surface sealing, shallow depth, permanent waterlogging, low bearing capacity, weak anchorage for plants, subsidence upon drainage, frequent micro- nutrient deficiencies and irreversible shrinking of the organic material upon drying
Semi-humid wooded to Semi-arid savannah	Xerosols, Luvisols, Cambisols, Arenosols, Lithosols, Rendzinas	Angola, Botswana, Swaziland, Namibia, Zambia, Zimbabwe, South Africa	Moisture stress, low fertility, salinity and alkalinity, deficiencies of Fe and Zn
Semi-arid to Arid savanna zone	Yermosols, Xerosols, Lithosols, Regosols	Namibia, South Africa, Angola	Extremely low inherent fertility, susceptible to wind erosion, devoid of moisture throughout the year

Sources: Sant-Anna (1985); FAO (2000)

Soil related constraints impacting crop production within Southern Africa

Physical constraints

Crop production in most parts of Southern Africa is constrained by a lack of continuously available water during the growing season (FAO/AGL 2000). Soil moisture stress is not only a function of the low and erratic precipitation in the region, but also of the inability of the soil to hold and

release moisture. The Oxisols, Alfisols, Ultisols, and other sandy soils of the region are particularly prone to moisture stress owing to their very low available water holding capacity (NRC 2009). High soil temperatures also produce high water loss through evaporation from the soil surface (Du Preez, Kotze, and Van Huyssteen 2018).

Wind and water erosion is extensive in many parts of Southern Africa and considered the most common form of soil degradation among the major soil groups in the region (Stander et al. 2021). High rates of erosion mainly occur in the semi-arid and sub-humid areas. Erosion causes loss of fertile land, declines in plant nutrients, damage to standing crops, and siltation of dams (Stander et al. 2021). Consequently, erosion results in the degradation of soil physical characteristics such as infiltration rate, soil structure, and crusting. Besides decreasing the efficient use of fertilisers by increasing nutrient losses, the physical removal of surface soil by erosion is a serious problem in shallow gravely soils as it compromises their effective rooting depth (Parwada and Van Tol 2016).

Vertisols and vertic soils occur extensively in many countries of Southern Africa, especially in dry semi-arid tropical climates (Jutzi et al. 1987). These soils are potentially a highly productive group if properly managed. However, Vertisols have a high (at least 30%) smectite clay content that makes them very hard and almost intractable when dry, but sticky and slippery when moist. Consequently, they cannot easily be cultivated with a hand hoe or an ox-drawn plough (Fey 2010). Low trafficability results in the disruption of farm operations, especially where farming is mechanised (Jutzi et al. 1987). Another feature associated with some Vertisols is the exhibition of a “*gilgai*” micro-relief, which makes seedbed preparation difficult, produces patchy growth, and is prone to soil erosion (Verster, de Villiers, and Scheepers 1973). Due to their very low hydraulic conductivity and infiltration rates, Vertisols easily get waterlogged during the rainy season. This leads to loss of nitrogen by denitrification, high alkalinity, and cation imbalances, with higher exchangeable magnesium (Mg) than calcium (Ca) (Fey 2010).

Hardsetting, crusting and sealing are forms of land degradation associated with poor aggregation and structural instability that cause the surface macrostructure of some soils in the region to collapse during wetting and harden sufficiently during drying to prevent seedling emergence (Mullins et al. 1990). Hard setting soils also experience other physical problems associated with unstable structure, including poor aeration, poor infiltration, runoff, and erosion (Mullins et al. 1990). In some soils, the dispersion of soil aggregates by raindrop impact also leads to the formation of surface crusts when the soil dries (Nciizah and Wakindiki 2014). The

problems of hard-setting and surface sealing in soils are widespread in Southern Africa and are associated with poor seedling emergence and crop establishment, together with enhanced runoff, which can accelerate erosion (Chikwari et al. 2015).

Soil compaction occurs in many parts of Southern Africa where agricultural mechanisation and grazing are practised (van Antwerpen, Meyer, and Meyer 2000). Prolonged cultivation of moist soils at the same depth exposes them to pressure induced by heavy machinery and gives rise to layers with high mechanical impedance, also called “tillage pans” (Esser 2016). Intensive tillage can also lead to subsoil compaction in some areas (Materechera 2018). In some soils of the region, high soil strength layers also occur naturally (Bennie and Krynauw 1985). Layers with high mechanical impedance restrict crop establishment, root growth, nutrient uptake, water infiltration through the soil, soil moisture storage capacity, growth, and yield of crops (Bennie and Krynauw 1985). Effective soil depth is a problem in more than 50% of the soils in Southern Africa and this reduces the potential of the soil for crop production (Msangi 2007). There are reports from several parts of the region that indicate that the effective soil depth of many soils is a problem because it reduces the volume of soil for root exploitation, thus reducing the potential of the soil for crop production (Nyamapfene, Hussein, and Asumadu 1988).

Chemical constraints

The predominant soils in Southern Africa are derived from granitic materials with low inherent fertility (FAO 2000). Poor soil fertility is recognised as a fundamental biophysical cause for declining food security among smallholder households in Southern Africa (Pardo et al. 2012a). The supply of plant available nitrogen (N), phosphorus (P) and sulfur (S), which are the most limiting nutrients, is low due to critically low soil organic matter (SOM) content in most of the regions’ environments (Msangi 2007). The potential of some of the soils to fix P is another problem, especially in sesquioxide rich Oxisols and Ultisols (Nyamapfene, Hussein, and Asamadu 1998). Furthermore, most of the soils have been subjected to leaching for a long time, resulting in deterioration of fertility, an increase in acidity, and sometimes the toxic effects of nutrient imbalance (FAO/AGL 2000). In terms of potassium (K), wide variations in the contents of micaceous clays and other K-bearing minerals imply significant variability in “slowly-available” K reserves for crop growth (Fey 2010). Toxic levels of aluminium (Al) species and other cations [particularly iron (Fe) and manganese (Mn)] occur in highly weathered, low pH red soils that are

dominated by oxide mineralogy (Nyamapfene, Hussein, and Asamadu 1988). The high levels of these cations interfere with nutrient uptake, sugar phosphorylation, and DNA synthesis when they enter the plant.

Soil infertility due to acidity is a serious constraint to the productivity of soils in Southern Africa, especially for the highly weathered and leached soils of high rainfall areas (Chinene and Lungu 1986). Many of the Ultisols, and some Alfisols with poor buffering capacity, have acid surface and subsurface horizons. Although the problem of soil acidity is largely natural, excessive use of nitrogen fertiliser and declining SOM have increasingly contributed to soil acidification in the region (Chinene and Lungu 1986). According to Haynes and Mokolobate (2001), the major fundamental factors that limit the fertility of acid soils include: nutrient deficiencies, especially P, Ca, and Mg, and the presence of phytotoxic substances such as soluble Al and Mn.

Salt affected (saline-sodic) soils are naturally found in pans and depressions of areas with low rainfall and high temperatures across Southern Africa (Van Huyssteen and Nell 2014). In other areas, salinity/sodicity has been induced by use of unsuitable irrigation water and poor management practices (Mutiro and Lautze 2015). It is estimated that about 24% of land in arid and semi-arid parts of the region are affected by salinity and alkalinity problems (Msangi 2007). Salinity has a negative osmotic effect on water uptake by plants, especially during emergence, while sodicity reduces the infiltration and internal drainage rates in the topsoil leading to waterlogging and erosion (Gotosa 1994). Sodicity also induces deficiencies of Ca and Mg (Fey 2010). Salinity and sodicity have been shown to affect not only chemical and physical properties, but also reduce microbial activity (Rietz and Haynes 2003).

Biological constraints

One of the most important threats to the sustainability of smallholder crop production in Southern Africa is the decline in soil fertility associated with falling levels of organic matter and soil nutrients (Pardo et al. 2012b). The major causes of declines in SOM are deforestation, continuous conventional cultivation involving ploughing, removal and burning of crop residues from arable land, and mono-cropping (Swanepoel et al. 2016). The effects of agricultural land use and management on SOM across the region have been reported widely (Du Preez, Kotze, and Van Huyssteen 2018).

The size, diversity, and abundance of communities of soil biota contribute to the soil processes that regulate nutrient cycling and decomposition of organic residues (Sanginga, Mulongoy, and Swift 1992).

However, some agricultural land use and practices in the region have been shown to decrease the abundance and diversity of faunal communities, thereby resulting in the degradation of soil properties (Graham, Haynes, and Meyer 2002).

Soils in Southern Africa also suffer from a number of soil-borne pests and diseases that cause loss to crop yields (Mapula, Mapotso, and Mamphiswa 2020). Soil-borne pathogens are those that undertake a major part of their life cycle in the soil. These include fungi, bacteria, viruses, and nematodes that infect roots or stem bases of crops and cause diseases. Furthermore, infection of roots by some pathogens, such as *Rhizoctonia solani*, may interfere with plants' ability to absorb water and nutrients (Mapula, Mapotso, and Mamphiswa 2020). In addition to field diseases, some of the common soil inhabitants, such as members of the genera *Aspergillus* and *Fusarium*, may cause postharvest decay of root crop such as groundnuts or cassava (Hillock and Waller 1997).

Pollution and contamination of soils

Soils in Southern Africa are polluted by agricultural practices, such as the application of pesticides, fertilisers, feedlot sludge, irrigation with polluted water, and addition of organic amendments, such as sewage sludge (Oberholster et al. 2012). Contamination of irrigation water and the agricultural environment by mining activities has also been reported as threats to soil resources in South Africa (Vermeulen and Usher 2009). In particular, the application of waste and effluent from piggeries and feedlots has resulted in salinisation and elevated levels of heavy metals in soils (Korentajer 1991). Fertilisers containing heavy metals and organic soil ameliorants may increase or decrease the bioavailability of heavy metals in the soil (Van Der Waals and Claassens 2001). These contaminants have degraded soil physical, chemical, and biological properties, which affects crop growth and yields (Van Der Waals and Claassens 2001).

Management practices used to overcome soil constraints and improve crop productivity in Southern Africa

Use of inorganic and organic sources of nutrients, and crop system management to improve soil fertility

Mineral fertilisers have been used as a source of soil nutrients in Southern Africa, mostly on cash crops (Morris 2007). In some cases, especially among smallholders, these fertilisers are applied in combination

with other nutrient resources, such as animal manure (Mugwira, Nyamangara, and Hikwa 2002). Most countries in Southern Africa have formulated fertiliser recommendations for all their major crops, sometimes with regionally specific adaptations (Mthimkhulu et al. 2018). However, the amounts of inorganic fertilisers applied are seldom sufficient to meet crop demands due to their high costs and uncertain availability, especially among smallholder farmers (Morris 2007).

Manure is a widely used resource for managing soil fertility in Southern Africa, especially among smallholder farmers (Mkhabela 2006). Manure from livestock such as cattle, goats, sheep, and donkeys is commonly available among households due to the integration of livestock in the farming system (Rufino et al. 2007). Consequently, extensive research has been undertaken on the various aspects of manure utilisation and management in the region. Smallholder farmers in Southern Africa have developed local knowledge on the use and management of animal (kraal) manure for improving and maintaining soil fertility. Smallholder farmers use old kraal sites as microenvironments for growing niche crops while taking advantage of the nutrients and superior soil properties in these locations (Materechera and Mkhabela 2001). Long-term application of manure has shown to increase, besides SOM, the soil microbial biomass and bacterial count (Belay et al. 2001). Green manuring is practiced by some farmers when space allows, however, because of small landholdings, most farmers practice green manuring only inadvertently when an early crop fails and has to be ploughed under before replanting (Scoones et al. 1996). Green manure's effectiveness depends on plant N and lignin content and the amount of biomass incorporated (Bloem and Barnard 2001). A number of green manure species, especially legumes, have been tested as cover in parts of Southern Africa (Mukumbareza, Muchaonyerwa, and Chiduza 2015).

Most farmers in Southern Africa have also adopted cropping systems with a multicanopy structure that provide continuous vegetative cover throughout the year in order to protect the soil against raindrop impacts, wind, and reduce run-off and erosion (Edje, Semoka, and Haule 1988). These systems include intercropping, rotations, and agroforestry. Intercropping, especially to increase soil fertility, has been used over a long time (Rasinamhodzi et al. 2012). Leaf litter has also been used as a source of nutrients for plants (Nyathi and Campbell 1993). The litter is collected from natural woodlands, crop fields, and households, then applied to the soil where it decomposes and release nutrients. However, because of its limited quantity and quality, the use of leaf litter has been limited and is mostly confined to areas where substantial amounts can be collected (Mafongoya and Bationo 2006). The systematic rotational sequence of crops on cropped

land has been a strategy adopted by smallholder farmers for enhancing soil fertility and exploiting residual nutrients (Thierfelder, Cheesmann, and Rusinamhodzi 2013; Njaimwe et al. 2016). Legume-based rotations have been advocated as a possible contributor to the nitrogen economy of a maize-based cropping system (Makoi, Chimphango, and Dakora 2009). Studies from the region have shown that, beside nitrogen, legumes are also a good supplier of Mg, Ca, and K (Mapfumo and Giller 2001). Crop rotations have also been used to benefit the humus content of the soil and protect against erosion due to their canopy cover and biological nitrogen fixation (Mukumbareza, Muchaonyerwa, and Chiduzza 2015).

Agroforestry has been promoted as an appropriate and sustainable response to climate change that is also capable of increasing crop productivity (Sheppard et al. 2020). In this system, fast growing leguminous trees, such as *Sesbania sesban*, *Gliricidia sepium*, *Tephrosia vogelli*, *Tephrosia candida*, and *Cajanus cajan*, are planted to replenish nutrients and allow more intensive and continuous cropping (Buresh 1998). These species, also known as “fertiliser trees”, recycle nutrients, fix nitrogen, reduce soil erosion, lower soil surface temperature and provides organic mulch and humus to the soil (Ajayi et al. 2011). In some dry areas of the region, indigenous agrosilvo-pastoral system takes advantage of the areas under leguminous trees to exploit the spatial and temporal variations of nutrients and moisture availability provided by *Faidherbia* (formerly *Acacia albida*) (Weil and Mughogho 1993). The tree sheds its leaves at the onset of the wet season, permitting enough light to penetrate for the growth of sorghum and millet, yet still providing enough shade to reduce the effects of intense heat. Consequently, smallholder farmers in dry areas of the region selectively retain *Acacia species* in their fields (Murovhi and Materechera 2006).

Soil amendments and conditioners

In acid soils, agricultural lime is applied to offset acidity (Haumann 2020). Haynes and Mokolobate (2001) have also shown that organic waste products can raise soil pH and reduce soluble and exchangeable Al. Industrial slags such as fly ash, coal ash, and phosphogypsum have also been used on acidic soils both to improve soil pH and to add essential nutrients (Harper and Mbakwe 2020). However, these by-products contain higher concentrations of hazardous and phytotoxic elements such as copper (Cu), arsenic (As), barium (Ba), bismuth (Bi), boron (B), zinc (Zn), Mn, and molybdenum (Mo), which may contaminate the soil (Van Der Waals and Claassens 2001). The effectiveness of locally available deposits of rock phosphates has also been investigated and technologies for its exploitation

have been developed (Dhliwayo 1998). Some studies have suggested that mixing of rock phosphates with organic matter of relatively wide C/N ratio can increase its solubility (Chinene and Lungu 1986).

Composting of both household and community waste has been investigated and used at various scales (Korentajer et al. 1999). The benefits of both digester and heat-treated sludge compost for soil quality and crop yields have been demonstrated in both pot and field studies (Hughes, Noble, and Scott-Brown 1993). However, the negative effects of sewage sludge application can be a concern due to the presence of toxic substances and pathogenic organisms (Korentajer et al. 1999). Vermicomposting, a combination of biological processes and techniques used to culture large quantities of certain earthworm species to speed up the stabilisation of organic wastes, has been used to generate high grade compost for agriculture (Reinecke, Viljoen, and Saayman 1992). The worms ingest the organic wastes, grind, and digest them with the help of some microorganisms, before being converted to fine, aggregated, stable, and humified microbially active faecal material called casts, which contain more soluble and available nutrients than the parent waste (Materechera 2002).

Termitaria, the soil around and within termite mounds, has also been used by smallholder farmers to manage soil fertility in some communities of the region (Nyamapfene 1986). Termitaria has been shown to contain high exchangeable bases (Ca, Mg, K) and micronutrients (Zn) (Nyamapfene 1986) and some farmers apply it in “dollop” form close to the base of the plant in order to improve nutrient supply (Edje, Semoka, and Haule 1988). Termitaria has also been shown to enhance soil structure, water holding capacity, and reduce soil erodibility, particularly in sandy soils (Watson 1976). Planting of crops on termite mounds has been practised as an in-situ strategy to utilise termite mound soil fertility by some smallholder farmers (Watson 1976).

Soil and water conservation and management practices

Considerable attention has been given to rainwater harvesting and management through the collection of surface runoff from micro-catchments in upslope areas, or through in-situ conservation of rainfall to mitigate against the problem of soil water scarcity in semi-arid areas of the region (Reij, Scoones, and Toulmin 1996). According to WOCAT (2010), the most commonly used in-situ rainwater harvesting and management practices include ridging, mulching, various types of furrowing and hoeing, and conservation tillage. In-situ water harvesting improves soil water availability by maximising the infiltration of water, which reduces surface

runoff and reduces evaporation (Botha, Anderson, and Van Staden 2015). Some factors that affect the adoption of water conservation technologies by smallholder farmers in the region have been reviewed by Perret and Stevens (2006).

Small-scale irrigation has widely been promoted in Southern Africa as a means of increasing crop production and adapting to climate change and climate variability in smallholder farming (Burney and Naylor 2012; Mutiro and Lautze 2015). Consequently, there are many irrigation schemes across the region whose performance and success appear to be linked to management style (Moyo et al. 2020). A wide variety of irrigation methods are employed under these schemes including flood, sprinkler, and drip (Mwendera and Chilonda 2013). However, other traditional forms of low-cost water control systems are also used by smallholder farmers, including rainwater harvesting, flood recession and flood water spreading, river diversion, treadle pumps, watering cans, porous jars, and small reservoirs (Bjornlund et al. 2020). Large-scale irrigation with modern technological investments is used on large commercial farms. The systems here involve sprinklers, centre pivots, surface/canal, drip, furrow, and other modern water distribution technologies. The application of fertiliser using drip irrigation has proved to be technically feasible and economically viable in crops such as sugarcane and horticulture (Howell and Conradie 2013).

Conservation agriculture and alternative tillage practices

In Southern Africa, conservation agriculture (CA) is increasingly being promoted as a climate smart approach that can help arrest or reverse the processes of soil degradation, promote water conservation, and mitigate against climate change (Thierfelder et al. 2013). Many studies have shown that, relative to conventional agricultural practices, implementing CA can increase soil organic carbon (SOC) storage by reducing the SOC turnover rate and increasing residue input rates (Cheesman et al. 2016). The integration of CA with high biomass cover crops can also increase water use efficiency and crop yields (Dube, Chiduza, and Muchaonyerwa 2017). However, uptake of CA by smallholder farmers is generally low in the region due to the low degree of mechanisation, lack of soil fertility management options, and competition for crop residues with other end uses (Mango et al. 2020). Some progress has been made with planters and sprayers, but a great deal more needs to be achieved in the immediate future (Thierfelder et al. 2013). According to Mango et al. (2020), conservation agriculture has a potential to be used for restoring soil quality on degraded cropland in most of Southern Africa.

Soil tillage has widely been used in the region as both a land preparation and conservation strategy (Willcocks and Twomlow 1993). Tractors, ox drawn implements and hand hoes have been used by different sectors of farmers. Ridging has widely been used by farmers to increase the depth of the soil above a water table, or an impeding soil horizon in the case of a perched water table, or on soils with slow internal drainage rates (Myburgh and Moolman 1991). Apart from improving water relations, other benefits of ridging include improved soil nutrient availability, weed control, and reduced compaction (Nyamangara et al. 2013). Other forms of tillage used to minimize soil compaction in cropped fields include controlled (zonal) traffic and precision strip tillage where mechanical cultivation is restricted to a strip of say 20-30 cm (Willcocks and Twomlow 1993).

Deep ripping, or subsoiling, in order to loosen compacted sub-soil layers has also been shown to improve water infiltration, root penetration, and crop yields (Chimbombi 1994). Despite its cost, it can also improve soil quality, health, and yields including the ability of plants to resist diseases (Gill, and Lungu 1988). Biological loosening of compact soils has been applied in the region by incorporating deep-rooted perennial cover crop species within cropping systems (Phophi et al. 2017). Apart from increasing SOM and nutrients, such cover crops have been shown to increase micro porosity, infiltration rate, moisture retention, and saturated hydraulic conductivity and decrease bulk density of soils (Chirwa et al. 2004).

Bioremediation and biological control of soil borne pests and diseases

The role of indigenous microbial communities in bioremediation of petroleum and heavy metal contaminated soils have been explored in the region (Lees et al. 1995). For example, the remediation of mining tailings using phytoremediation is being explored in Botswana, South Africa, and Zambia (Bruneel et al. 2019; McBain et al. 1996; Mghazli et al. 2019). The use of microbial processes to convert environmental contaminants into harmless substances has also been explored due to increasing concerns over contamination of soils and groundwater from improper disposal of hazardous industrial wastes (Tanner, Annandale, and Rethman 1999).

Similarly, the effects of land management on the size, composition, and properties of soil faunal communities, such as earthworms (*Eisenia fetida*), and their influence on soil physical, chemical, and microbial properties have been studied extensively (Dlamin, Haynes, and van Antwerpen 2001). The biological control of soil borne pathogens and diseases has been considered as an option, and augmentation and manipulation of the soil microflora,

such as by biofertilisation with cyanobacteria, has been shown to improve SOM and quality (Pardo et al. 2014). Crop rotations have also been shown to control soil borne pathogens that cause root rot in maize (Craven and Andre 2017).

Areas requiring future research

The management of soil constraints and the intensification of crop production in Southern Africa requires understanding of the capabilities and limitations of the soil resource base. In many countries of the region, surveys, classification, and evaluation of the soil resource base has been undertaken, but at scales too large for detailed planning to address specific soil constraints on crop productivity. There is also need for information on the mineralogy of the clay and clay fractions of most of the soils in the region.

The most appropriate use of tillage throughout the region is also one of the areas that is not fully understood and requires research attention. The location and crop specific nature of tillage implies that the characterisation of soil types will continue to be the guiding point for the selection of machinery and equipment tillage packages suitable for crop production in the different agro ecological zones of the region. Both compaction and shallow effective rooting depth continue to be severe constraints, especially on Ultisols and Alfisols, and attention regarding the most appropriate equipment and ploughing depth in these soils is required. Similarly, the development of appropriate agronomic systems and tillage practices to deal with the problem crusting on Inceptisols and Alfisols in semi-arid areas needs attention. Draught power is an important factor to consider, especially within the smallholder farming sector of the region

It is well-known that the maintenance and management of SOM is critical for improving crop yields. However, there are still many questions related to the role of SOM in sustaining the productivity of different soils in the region. For example, there are issues around the quantity and quality of organic inputs and their contribution to the build-up of the different SOM fractions depending on management practices. Process models should be used to translate the knowledge and predictive capacity generated through strategic research on soil biological processes into affordable on-farm options that improve the nutrient use efficiency from organic inputs. Development of manure and crop residue management options continues to be an important consideration, exacerbated by the changing climatic and economic circumstances of farmers, which demands the adoption of carefully planned specific mechanisation and soil management practices for the different soils. The interaction between SOM and other nutrient sources

also warrants attention.

The depth of understanding of the role of biological processes to maintain soil fertility in agricultural systems is also still superficial in the region. Research on the relationships between soil organisms and soil fertility is required to enable manipulation of processes that can improve productivity and soil properties. Soil biota management will involve the development of practices that manipulate specific populations of organisms to improve agricultural production. The opportunities offered by molecular biological techniques for genetic manipulation of organisms make this a particularly interesting area of research at the current time. The benefits of vesicular-arbuscular mycorrhizal fungi species remain largely unexplored, especially for drought prone areas. There is a need to quantify the wealth of indigenous knowledge and systems of soil fertility management that exist among communities of the region, including the balance of nutrients under various available options, especially in the smallholder farming sector. Because of the low quality and declining availability of locally derived nutrient sources, there is a need to determine the limit to which such farmer knowledge and innovation can be economically exploited for sustainable soil fertility management and the management of soil constraints to support the intensification of crop production.

Conclusion

Southern Africa has an extremely wide range of soils and climatic conditions. The major soil groups of the region include Alfisols, Ultisols, Oxisols, Histosols, Andisols, Inceptisols, Aridisols, Mollisols, Histosols, and Vertisols. Crop production under both smallholder and large commercial farming in Southern Africa is negatively affected by various constraints posed by the soil resource base. The majority of soils in the region are poor with low fertility, organic matter content, and water retention. The soils are also vulnerable to water and wind erosion, leaching, and salinisation when under irrigation. The nature and extent of the constraints vary depending on the mineralogical and morphological characteristics of the soils. Pollution by agricultural and industrial activities have also exacerbated the degradation of the physical, chemical, and biological properties of the soils. A wide range of management practices have been developed and adopted as strategies to overcome soil constraints under smallholder and commercial farming sectors. Using these approaches, the potential exists to improve the productivity of agricultural soils in the region, however, there are still areas that need further research to enhance our understanding and utilisation of these processes.

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CHAPTER 14

SOIL CONSTRAINTS FOR CROP PRODUCTION IN SOUTH-EAST ASIA

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Abstract

In South-East Asia (SEA), crops, such as paddy, oil palm, rubber, cocoa, cassava, and vegetables support both smallholder farmers and the plantation sector. Socially and financially, agriculture forms the backbone of society in many developing SEA nations. However, particularly since the Green Revolution in the 1950's, agricultural activities have led to soil degradation, which affects crop yield. This chapter highlights selected soil constraints affecting agricultural activity in SEA including soil erosion, soil acidification, soil salinity, and waterlogging. In the SEA region, Malaysia and Indonesia are well known to be net exporters of palm oil, subsequently generating a by-product known as palm oil mill effluent (POME). The recycling and reuse of POME as a soil amendment is a viable practice. Paddy cultivation often requires liming to alleviate soil acidity throughout SEA. In northeast Thailand, soil salinity is the main constraint for producing cassava, which can be alleviated through irrigation. Meanwhile, in the Philippines, soil erosion is a major soil constraint due to a combination of steep topography, deforested uplands, and heavy rainfall (>3000 mm year⁻¹). The way forward to mitigate these constraints will involve community engagement with governmental level coordination to recycle-reuse POME, manage water resources (irrigation systems), reduce chemical fertilisers, promote organic amendments, and finally, practice crop rotation/mix-crop cultivation.

Keywords: soil-crop constraints, erosion, acidity, plant nutrient depletion, soil degradation, pollution.

Introduction

South-East Asia (SEA) is located in the tropics and subtropics (Figure 14.1), and mainly falls within *Af* (tropical monsoon) and *Am* (monsoon) classifications under the Köppen system. Soil formation and development dating back to the Holocene period (~10,000 years ago) is highly variable. Malaysia was formed mainly from raised tectonic shift, as was the Indonesian archipelago, although this area also has active volcanoes. Thailand and its neighbouring countries are affected by a vast river system, now referred to as the Mekong Delta. South-East Asia's tropical and subtropical climate has created many soil types, depending on the moisture regime and drainage conditions. In well drained upland areas where soil erosion is minimal, Ultisols and Oxisols are the dominant soil orders based on USDA soil taxonomy. Meanwhile, the low-lying areas of the coastal region are dominated by hydromorphic soils that are acidic in nature – i.e., peat soils (Histosols) and acid sulfate soils (Entisols and Inceptisols). Elsewhere along the coast are found very sandy soils (Entisols and Spodosols), which have difficulty sustaining crop production. Spodosols, Inceptisols, or even Entisols occasionally occur in the highlands.

The region produces many important crops, including rice, corn, and oil palm. Crop production often relates to farm size and their respective practices. Average farm size ranges between 1-3 ha. One-third of farms (rice and cash crops) are less than 0.5 ha in size, one-half less than 1 ha, and three-quarters less than 2 ha (Barker et al. 1985). Rice is the staple food, and flooded rice paddies dominate most agricultural landscapes. This region produces 15% of all the rice grown in Asia. Rice is particularly prominent in floodplains and alluvial valleys (i.e., Malaysia), but terraced rice (Indonesia, Thailand, Laos) is also common in upland areas wherever irrigation water is available. Aerobic rice is also cultivated, but to a lesser extent, except in Vietnam (winter rice). Irrigated rice land generally yields much more than rain-fed rice.

Continuous cropping is typically practiced, with five seasons over a two-year period. In Indonesia, for example, sequential cropping of rice followed by corn or soybean is practiced. Sometimes complex intercropping of annual field crops is practiced in countries like Myanmar and Vietnam. One rice variety in Malaysia, Bario rice, has obtained Geographical Indication (GI) and is grown in the rural up-land of Sarawak. The variety of rice grown e.g., *indica*, *aromatica*, or *bashmati*, indicates the extent of soil variation

within the region.

The next most important annual food crop is corn (*Zea mays* L.), with some variations such as reddish-waxy corn (glutinous, rather than sweet corn) also cultivated to a lesser extent. Oil palm is mainly grown in medium to large plantations, with only a small percentage owned by smallholders. Soil nutrient dynamics (texture, acidity, nutrient availability, organic matter, salinity etc.) coupled with rainfall (2000-3500 mm year⁻¹) highly influence plantation management. Each plantation develops their respective management approach, making generalisations difficult. Oil palm is a tolerant crop to various soil conditions and can be grown either in mineral soil or peat. Mineral soil is preferred; however, land limitations often push cultivation into peat areas. Peat areas are known to be high in organic matter, and thus may provide readily available nutrients for uptake, however, peat areas often subside. The rate of subsidence depends on the stage of decomposition (between *hemist*, *fibrist*, and *saprist*). Under such condition, the management of oil palm, or any other crop, requires site specific management. These practices, while often costly at the early stage, are sustainable in nature.



Figure 14.1 Map of Southeast Asia.

Despite the variation in the natural geological processes of soil formation, the countries of the SEA region share similar soil constraints to crop production. The soil constraints that particularly affect the region include erosion, acidification, salinity, waterlogging, nutrient and organic matter depletion, and anthropogenic (man-made) pollution. Managing these issues is required for the sustainable development of agriculture.

Soil Erosion

The tropical areas of SEA are characterised by high rainfall (2000 – 3500 mm year⁻¹). These high rainfall rates make soil vulnerable to erosion, particularly in the steeplands (regions with slopes >20°) (Table 14.1, Figure 14.2). Steeplands have been utilised for agriculture despite the constraints of soil erosion (Table 14.1; Table 14.2). A way forward is to manage the steeplands using terraces (i.e., orchard terrace, riser, and bench terrace) and mulching, and with drainage and irrigation systems. Similar practices have been adopted successfully in Indonesia (Bogor), the Philippines, Laos, Thailand (Chiang Mai), and other regions of SEA as well.

Table 14.1 Land areas in Malaysia with a steep slope (> 20°)

Location	Total size (million ha)	Steep slope area (million ha)	Percentage land with a steep slope (%)
Peninsular Malaysia	13.2	5.04	38.19
Sabah	7.4	4.5	60.80
Sarawak	12.4	8.3	66.90

Source: Department of Agriculture, Malaysia (2020)

Soil water erosion can be categorised as splash erosion, sheet erosion, rill erosion, gully erosion, and riverine erosion. Of these, splash, sheet, and riverine erosion are common within SEA. Erosion often accompanies deforestation and the opening of land for agriculture, particularly in the steeplands. Many factors contribute to the soil erosion rates, with slope (categorised into terrain class), soil texture, and effective soil depth being among the most important Table 14.2 outlines the fundamental terrain class adopted in Malaysia to manage soil erosion problems.

Table 14.2 Terrain classes in Malaysia

Terrain Class	Slope	Percentage (%)	Terrain type
	Degree (°)		
C1	0-2	0-4	Flat
C2	2-6	4-11	Undulating
C3	6-12	11-21	Rolling
C4	12-20	21-36	Hilly
C5	20-25	36-47	Very hilly
C6	25-30	47-58	Steep
C7	>30	>58	Very steep

Source: Department of Agriculture, Malaysia (2020)

Studies conducted in Cameron Highland (Malaysia) estimated erosion was occurring at a rate of $24 \text{ t ha}^{-1} \text{ year}^{-1}$ under broad platform and cut-slope terraces (Midmore et al. 1996) and at some localities was as high as $115 \text{ t ha}^{-1} \text{ year}^{-1}$. Soil texture varied from sandy clay loam to clay loam, with terrain classes between C3 to C5. These highlands are cultivated with vegetables, flowers, and fruits for local consumption and export. Terraces are resurfaced periodically due to impoverished soil and reduced yields. In Thailand, Putthacharoen et al. (1998) recorded soil losses of $21\text{--}142 \text{ t ha}^{-1} \text{ year}^{-1}$. Cassava cultivation practices had the highest soil loss, followed by mungbean, sorghum, maize, and pineapple. Pineapple cultivation gave the lowest soil loss in its second cycle at $21 \text{ t ha}^{-1} \text{ year}^{-1}$. The soil texture was sandy loam and terrain classes were between C2 to C3. The author added that although pineapple requires thorough land preparation before planting and can cause serious erosion before and shortly after planting, its long growth cycle, especially if left as a ratoon crop, results in total soil losses being lower than those of most other crops. Meanwhile, in Indonesia Trigunasih et al. (2018) noted higher soil losses of $15 \text{ t ha}^{-1} \text{ year}^{-1}$ and higher in plantations cultivated with oil palm. In Central Vietnam, Pham, Degener and Kappas et al. (2018) recorded soil losses of between $1.45\text{--}3.7 \text{ t ha}^{-1} \text{ year}^{-1}$ under agricultural production.

In the Philippines, data based on a Food and Agriculture (FAO) study in the 1980's recorded soil losses of $\sim 88 \text{ t ha}^{-1} \text{ year}^{-1}$ from agricultural areas with a slope more than 15% (Class C3, Rolling) that required terracing. However, the use of terracing was limited among smallholder farmers. Thus, a case study in the uplands of the Philippines recorded yield improvement and decreases in soil erosion when farmers adopted soil conservation technologies such as terracing and reduced their reliance on subsistence rain-fed farming (Cramb et al. 1998). Fundamentally, soil erosion in the Philippines occurs due to a combination of factors, among them topography (flat, undulating, slope, hilly etc.), soil texture, cover crop availability, farmers' practice (tillage, slash and burn), type of crop, vegetation canopy, and climatic condition (Olabisi 2012). Continuous monitoring and assessment of soil losses is therefore important.

Even though steepplands ($>25^\circ$ slope) are Class C6 and C7 (Table 14.2), they have the potential to be used for crop production. Terracing, mulching, drainage, and irrigation systems, when well employed, can make the cost-benefit ratio acceptable. The former approach is often practised in Malaysia for the development of oil palm plantations, whereas the latter is used for rice cultivation. Bario rice (Sarawak) is one example of good and natural adaptation of drainage and irrigation systems in the highlands of Borneo to sustain rice production. Despite the constraints, Bario rice cultivation

receives geographical indication (GI) recognition for the success of rice cultivation in the highlands. A GI is a sign used on products that have a specific geographical origin and possess qualities, or a reputation, that are due to that origin. However, what is acceptable for the economy are sometimes not for the environment, hence the debates lives-on.

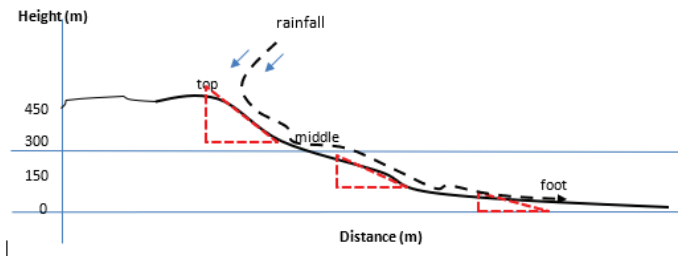


Figure 14.2 Soil erosion based on slope condition.

Soil Acidification

Soil acidity in Ultisols and Oxisols

Ultisols and Oxisols are highly weathered, and their mineralogy is dominated by kaolinite and sesquioxides. In the subsoil, Oxisols contain greater amounts of positive charge than in the topsoil. This has important implication for soil management for sustainable crop cultivation (Tessens and Zauyah 1982; Shamshuddin and Ismail 1995). The CEC of the soil is low, while basic cations are insufficient for crop requirements. Soil pH is <5 and exchangeable Al is above the critical level for the healthy growth of Al-sensitive plant species.

Ultisols and Oxisols are widespread in SEA, especially Malaysia, Indonesia (Kalimantan and Sumatra), and even Thailand. In Malaysia, the estimated land area covered by highly weathered soils is about 70% (Shamshuddin and Fauziah 2010). This is where oil palm and rubber are productively cultivated because the crops are acid tolerant. Ammonium sulphate is regularly applied onto cultivated Ultisols and Oxisols to up-scale rubber and oil palm production. Oxidation of the ammonium ions released by the fertiliser produces acidity that lowers soil pH slightly (Shamshuddin and Fauziah 2010).

However, to grow corn sustainably on weathered soils, soil acidity and/or Al^{3+} toxicity must be alleviated, either via ground magnesium (Mg)

limestone (GML) application (Shamshuddin, Fauziah, and Sharifuddin 1991) or possibly with Mg-rich synthetic gypsum (Ayanda et al. 2020). The use of GML is a common practice for rice farmers with acid sulphate soils. Some oil palm plantations also have adopted this measure to improve soil acidity. For rice farming, GML is applied by spreading thoroughly onto the land, whereas for plantation systems, pocket-hole application is more suitable.

Acid sulfate soils

In SEA, acid sulfate soils occur mainly in Indonesia, Vietnam, Cambodia, and Malaysia. These soils are characterised by the presence of a near-surface pyrite (FeS_2) layer. When pyrite is exposed to air, as occurs following the drainage of formerly inundated land, the sulfides are oxidised to Fe (III) sulfates, whereby sulfuric acid is generated. This process creates soil acidification, rendering these soils only marginally suitable for agriculture: the concentrations of Al, Fe, and Mn can reach levels toxic to crops. Often soil pH is as low as 3.3 to 4.0. To alleviate soil acidity, as stated earlier, application of lime prior to cultivation is a common practice. Recent research has also examined the effectiveness of biochar and organic manure; however, results have been mixed and these amendments are not widely used by the farming community.

Acid sulfate soils are often viewed as being very difficult to manage for sustainable crop production because of low pH stress and high Al^{3+} and/or Fe^{2+} toxicity. Decreased P availability, nutrient deficiencies, and hydrogen sulfide toxicity also inhibit plant growth (Dent 1986), and they are usually associated with a high water table. It has been established that proper water and soil/agronomic management is crucial to sustaining the production of rice and oil palm on these soils (Toh and Poon 1982; Janssen et al. 1992; Anda, Siswanto, and Subandiono 2009; Shamshuddin et al. 2014; Shazana et al. 2014; Elisa et al. 2014).

Rice cultivation

Some acid sulfate soils in Malaysia have been utilised for rice cultivation, but yield is below the national average of $4 \text{ t}^{-1} \text{ ha}$ (yield range between $2\text{--}3 \text{ t ha}^{-1}$). This is due to the effects of low pH stress or Al^{3+} and Fe^{2+} toxicity (Shamshuddin et al. 2014; Panhwar et al. 2014; Elisa et al. 2016), although, rice plants can partially overcome the phytotoxic effects of Al^{3+} or Fe^{2+} by secreting organic acids that chelate the metals, rendering them unavailable for root uptake (Panhwar et al. 2014). The critical water pH for rice cultivation is 6 (Alia et al. 2015), while the pH in acid sulfate

soil areas without undergoing mitigation is mostly <4 , which can have dramatic effects on rice root growth.

Two methods to ameliorate acidity in acid sulfate soils are the addition of GML (Elisa et al. 2014) and/or ground basalt (Shazana et al. 2013; Shazana et al. 2014). These additions increase the soil pH, and when values >5 are achieved, Al^{3+} and Fe^{2+} will be precipitated as inert Al- or Fe-hydroxides (Shamshuddin, Fauziah, and Sharifuddin 1991; Shamshuddin and Ismail 1995). GML can also supply Ca and Mg, which are required by rice in large amounts, and Ca can reduce Al^{3+} toxicity (Alva et al. 1986).

A study by Panhwar et al. (2016) found that applying GML and/or basalt in-combination with bio-fertiliser (4 t ha^{-1} each; biofertiliser fixed at 1×10^8 cfu) promoted rice growth on acid sulphate soils. Phosphate-solubilizing bacteria (PSB) present in the bio-fertiliser not only increased P availability, but also helped release organic acids that inactivated Al^{3+} and/or Fe^{2+} via chelation (Panhwar et al. 2014). According to Panhwar et al. (2016), the ameliorative effects of applying basalt + bio-fertiliser would last for up to 3 cycles of rice cultivation. The longer residual effect of basalt was postulated to be because basalt is an ingenious rock that takes longer to dissolve than limestone, even in an acidic environment. Therefore, basalt (ground finely) can be a slow-release (nutrient) soil amendment.

Oil palm cultivation

Oil palm is severely affected by the presence of high amounts of Al (Shamshuddin et al. 2014). Based on the study of Auxtero and Shamshuddin (1991), the critical pH for oil palm growth/production is 4.3, while the critical Al^{3+} threshold is $100 \mu\text{M}$. Flooding enhances oil palm growth considerably (Auxtero and Shamshuddin 1991) and has been observed to increase yields from 5.7 to $17.5 \text{ t fresh fruit bunch weight (ffb)}^{-1} \text{ ha}^{-1} \text{ year}$ (Poon 1977). Hence, it is advisable not to completely drain acid sulfate soils during land preparation for oil palm cultivation. Drains are only dug to remove the excess water present in the area, as this will prevent the oxidation of pyrite minerals and the generation of acidity. This method has proven to be a viable agronomic practice in the long run. According to Poon (1977) and Toh and Poon (1982), this approach can produce a yield of $25 \text{ t ffb}^{-1} \text{ ha}^{-1} \text{ year}$.

Soil salinity

Global climate change has the potential to cause sea level rise and changes in precipitation. This will have adverse effects on countries with vast coastal areas like Malaysia, Indonesia, Thailand, parts of Vietnam,

Myanmar, and other Southeast Asian regions. Sea level rise and associated seawater inundation and inland subsurface seepage can increase soil salinisation, especially for areas near the coast. This has been observed in the paddy granary around the East Coast of Malaysia and Sabah coastline, which has an elevation of only 1-2 m above sea level. High soil salt contents can affect plant growth by decreasing the rate of water uptake due to osmotic effects, causing ion toxicity (e.g., Na, Cl) and inducing nutritional imbalances due to the interference of high salt contents with the uptake and transport of essential nutrients.

Generally, as soil salinity increases, crops will begin to experience yield decline, with the point at which yield is affected dependant on the salinity tolerance of the crop. Soil salinity $>4 \text{ dS m}^{-1}$ can reduce yield by 50% in sensitive crops. Among crops, corn has been classified as one of the most sensitive and many studies have noted that corn growth (and yield) decrease by about 30% when subjected to 2 dS m^{-1} (EC) of salinity (Pesqueira, Garcia, and Molina 2003; Radic, Beatovic and Mrda 2007; Blanco et al. 2008; Shtereva et al. 2015; Shazia et al. 2019). In Malaysia, kenaf (*Hibiscus cannabifolius* L.) is cultivated in coastal soils with higher salinity levels ($2\text{-}6 \text{ dS m}^{-1}$) with mixed success. Rice, on the other hand, is a moderately salt sensitive crop with tolerance levels up to 4 dS m^{-1} . Although, it should be noted that salt application method, experimental set-up, and result interpretation can influence the critical tolerance limits identified (Shavrukov 2013; Woodrow et al. 2017). Interestingly, a study from China by Tang et al. (2019) stated that, phosphorus (P) deficiency improved salt tolerance of corn plants, which was associated with the increase of tissue mass density, accumulation of osmolytes, reduction of sodium (Na) accumulation, and selective absorption of potassium (K) over Na.

Study on the response of crops to salinity stress can assist in breeding salt tolerant cultivars by identifying the physiological features with the potential to increase salinity tolerance. In a study by Mondal et al. (2013) in Malaysia, for example, salt tolerant rice cultivars had lower Na and higher K content. In Malaysia, MR219 variety F1 (First generation) has been isolated and tested positive for salt tolerance up to 10 mM NaCl (Kalhori 2017). MR219 is preferred by farmers over MR269, even though they are equally salt tolerant, as farmers have sufficient knowledge of the management of the former variety.

Most rice fields in Thailand are located in central and northeast regions and affected by underground rock salt accumulation. Salinity levels in the order of $6\text{-}15 \text{ dS m}^{-1}$ have been recorded in the region. Hom jan rice was identified as salt tolerant, and further research led to the development of Lhao Dawk Mali 105 (KDML 105) and Pathumthani (PT1), which have a

unique scent, long grains, high amylose content, and a soft texture (Rha et al. 2016). It is paramount for Thailand to develop such tolerant varieties because rice farming is mainly conducted under rainfed conditions. Irrigation can reduce the salt stress; however, irrigation systems are costly to set-up, particularly for smallholder subsistence farmers. In addition, future reduced precipitation and sea level rises are predicted to reduce the water available for irrigation (surface and subsurface) (Teh and Koh 2016).

Waterlogging

Waterlogging affects approximately 15% of SEA's agricultural land and significantly reduces crop yield. In agricultural soils, the primary cause of waterlogging is heavy rainfall, inadequate soil drainage, and high clay content. The high rainfall (up to 3500 mm year⁻¹) experienced by SEA countries increases their susceptibility to waterlogging. Climate change (especially increases in temperature) will also increase waterlogging risk due to the expected increase in intense rainfall events (IPCC Report 2018, SR 1.5).

An example of the impact of waterlogging on crop production in SEA can be seen in chilli pepper production. The waterlogging tolerance of chilli pepper was studied by Higashi et al. (2017) in Indonesia using local genotypes (*Ferosa*, *Laris*, *Romario*). The author stated that because of heavy rain, land was often covered by excess water, which resulted in negative crop growth. This water stress limits root respiration and triggers root failure due to the development of anaerobic conditions. In Sarawak (Malaysia), chilli pepper is grown in drained peat areas. Water is drained from peat and maintained at lower than the initial water level to reduce the risk of excessive peat subsidence.

However, waterlogging can also be supportive of crop production when managed properly. Rice production in paddy fields is an example of where waterlogging is actually essential for good growth. Here flooding helps reduce weeds and pests and improves nutrient uptake. To generalise the effect of waterlogging on crop production is thus quite difficult because topography and rain distribution also play an important role. Model simulation is often adopted with various modelling tools (e.g., SWAT, Kinetic, Hydrus) used to assess water resources in selected areas.

Depletion of plant nutrients

According to the Food and Agriculture Organisation (FAO) (2003), the depletion of soil nutrients can be defined as a "soil degradation phenomenon

that occurs as a consequence of soil erosion (it is the topsoil in which generally most plant nutrients are present and which erodes fastest) but also because of poor management practices, such as slash and burn and other subsistence agricultural practices that do not replenish the nutrients taken out of the soil by the crops.” In SEA, plant nutrient depletion can be due to many factors, and poor management is a major reason. For example, oil palm is a perennial plant that grows well in tropical climates and is highly productive. However, to support its growth and yield, it requires large inputs of nutrients such as nitrogen (N), P, K, and Mg. Many of the soils used for oil palm production across SEA are inherently infertile (e.g., Ultisols and Oxisols). Repeated cultivation of the same land parcel further aggravates the depletion of plant nutrients. Thus, macro- (N, P, K, Ca, Mg) and micro- [e.g., boron (B), zinc (Zn), copper (Cu)] nutrient input to meet crop requirements is paramount. In the region, farmers are guided on fertiliser type and application rates, with basic fertiliser requirement based on the crop. For example, 120 kg ha⁻¹ of N is recommended for paddy production. However, with increases in fertiliser prices, SEA farmers tend to reduce their fertiliser usage in favour of alternatives, such as palm oil mill effluent (POME), biochar, food waste (bokashi), wood vinegar, compost, sawdust, and other locally available organic sources.

Another common approach to mitigate nutrient depletion is the application of ground dolomitic limestone as GML. Dolomitic limestone is a sedimentary carbonate rock that contains a high percentage of the mineral dolomite, CaMg(CO₃)₂. For paddy cultivation, surface application of GML supplies Ca and reduces soil acidity, hence improving nutrient availability.

Silicon (Si) is also an essential nutrient that is applied for plant growth. In paddy, it improves yield (Datnoff and Rodrigues 2005), and in oil palm, Najihah et al. (2015) recorded that Si concentration was significantly associated with the transpiration rates of oil palm seedlings that were not infected with *G. boninense*. A concentration of 1200 mg Si L⁻¹ of silicon oxide increased seedling transpiration rates by about 56% and reduced basal stem rot disease. Silicon deficiency in paddy affects the development of strong leaves, stems, and roots (Elisa et al. 2016). It also affects the formation of a thick silicated epidermal cell layer (Datnoff and Rodrigues 2005), and deficiency makes rice plants susceptible to fungal and bacterial diseases, and insect and mite pests. Rice hulls are a common source of silica for paddy farmers. Silica uptake is governed by the root system, and when nutrient depletion occurs, the plant will respond to maximise nutrient acquisition using a complex set of mechanisms within the root system that maximise nutrient acquisition while protecting against the accumulation of excess nutrients, which can be toxic to the plant.

The soil organic matter (SOM) pool is also an essential source of plant nutrients, with each carbon source having specific functions to improve soil fertility. Crop production can deplete SOM by decreasing organic matter inputs and increasing decomposition rates. Among the factors involved in the disruption of organic matter supply to the soil is the removal of crop residues to feed livestock and/or to make compost. This removal impoverishes the soil as the nutrients in plant residues are removed off-site rather than being recycled. Soil tillage is also well known to decrease SOM concentrations due to the soil aeration and stimulation of organic matter decomposition that it induces. Tillage is a common practice for paddy cultivation in many SEA countries and depletes SOM to some extent. Reduced tillage is a good approach; however, farmer adoption remains limited. Groundnut and sweet potato are two crops that can benefit from reduced tillage.

In peat (organic soil) areas, which are often poorly aerated, the decomposition/depletion of organic matter is slower due to reduced oxygen supply, which limits aerobic decomposition (Brady and Weil 2008). Peat soils are cultivated with oil palm and pepper in Malaysia, and with oil palm in Indonesia. Pineapple cultivation in Malaysian peat initially results in good yield, however, with intensive and repeated cultivation, yield declines of about 10-15% per hectare can be noted after 3-5 years. After harvest, 13 t ha⁻¹ of pineapple residues are generated. A study by Choo et al. (2021) found that pineapple residue ash was able to maintain higher soil pH during pineapple growth and contributed to lower CO₂ and N₂O emissions. Increases in peat soil pH also facilitates organic nitrogen mineralisation and nitrification.

To maintain supplies of plant nutrients, organic matter must be regularly added to the soil to replace that lost via decomposition or erosion. The quality of the organic material added will affect its ability to supply crops with plant nutrients. A common measure used to quickly assess organic material quality is its C:N ratio (known as carbon to nitrogen ratio), as this indicates the type of organic material (Table 14.3) and its ease of decomposition (Table 14.4). A high C:N ratio indicates hard woody materials more resistant to decomposition compared to low C:N ratio materials, which are often associated with soft leafy materials. The C:N ratio of soil is dynamic and variable depending on various factors involved in the increase and depletion of soil organic matter.

Table 14.3 C:N ratio (range) of selected organic materials

Organic materials	C:N ratio (range)
Sawdust (softwood/hardwood)	150-300
Leaf litter (plant residue)	34-85
Straw (paddy harvest residue)	50-70
Vegetable scrap	20-50
Sewerage sludge	6-20
Manure	5-25

Table 14.4 C:N ratio effect on decomposition

C:N ratio	Description
<10	rapid decomposition
15-25	slower decomposition
>20	nitrogen may be immobilised, reducing crop availability
>25	shows that organic matter is at stage “raw”

As added organic material decomposes it produces humus (humification). Humus in the soil:

- i. improves fertiliser efficiency.
- ii. stores N.
- iii. improves nutrient uptake, particularly P and Ca.
- iv. acts as a food source for soil flora and fauna.
- v. helps reduce the impacts of salinity as it buffers plants from excess Na.

Compost and biochar application to soil are known to improve SOM and plant nutrients status in the region. However, application is labour intensive, and the quality varies from batch to batch produced.

Degradation of soil structure

Soil physical fertility is essential for optimum plant growth. As such, particle distribution, texture, water-holding capacity, hydraulic conductivity, bulk density, and porosity all contribute to crop productivity and play a vital role in soil fertility management as well as soil chemical fertility. For example, soil strength is measured via penetration resistance (MPa). A soil

with a strength below 1.25 MPa will have a loose to medium degree of soil consolidation, meaning that plant roots can grow well. Soils with strengths between 1.25-2 MPa are denser and may hinder root penetration to the subsoil. At 2-3 MPa, very few plant roots are able to penetrate the soil, and above 3 MPa, plant root growth is physically impossible. However, one caveat is that moisture content can reduce soil strength and improve root penetration. In addition, cracks, fissures, and pre-existing biopores may assist in root penetration, and the presence of these structures are not reflected in a penetrometer measurement. Therefore, plants sometimes thrive, even in dense soils.

South-East Asian countries have abundant rainfall, which means the moisture content in the soil often assists plant root growth. However, in Kedah in the northern part of Malaysia, soil cracking can occur. After the rainy season, weather is hot and humid for an average of 3 months. Under these condition, high clay content soils can dry and experience soil cracking. In the past, tunnel erosion has also been noted. Irrigation during rice production can close soil cracks, although this requires large volume of water as these soils have a water holding capacity of up to 70%. Excessive water may lead to over-saturation of parts of the soil profile, which can damage the root system.

Tunnel erosion occurs when surface water percolates into and through dispersive subsoil layers. Dispersive soils are poorly structured and disperse and erode easily when wet. Tunnel erosion occurs when surface water moves into and along cracks. This problem can be managed by irrigating the soil cracks and applying lime/gypsum. Calcium ions from the lime/gypsum can displace Na ions in sodic layers and provide electrolyte balance to help flocculate clays. However, both irrigation and lime/gypsum application often require governmental assistance because farmers have limited resources and lack the technical knowledge to manage this problem.

In Southern Thailand and north-eastern Malaysia, sandy soils (Entisols) are dominant and can comprise 80% of soils (Roslan et al. 2010). Entisols are soils with no profile differentiation and/or with very minimal differentiation within 100 cm depth. Despite this limitation, tobacco, chilli, and kenaf (Roslan et al. 2011; Basri et al. 2014) thrive well in these areas, locally known as Beach Ridges Interspersed with Swales (BRIS) soils. In Southern Thailand, leaf litter decomposition is noted to improve overall soil physical and chemical properties for rubber-based agroforestry systems compared to monoculture rubber systems (Uraiwat et al. 2018). *In-situ* leaf litter (leaves, twigs, and fruits) was compared with no external leaf litter application. Under agroforestry systems, oak plant plots had higher organic C and N

content compared to maize plots. Nutrient content was higher in the topsoil compared to subsoils, and it improved soil-physio-chemical characteristics such as soil pH, EC, organic matter, and nutrient content.

Decline in soil structure can be noted visually via several indicators, as stated below:

- i) Hard surface – a very hard surface can be noted when the soil dries out, with continued drying leading to soil cracking.
- ii) Ponding – water ponding after relatively light rain.
- iii) Surface crust – the crust can often reduce infiltration and restrict seed emergence.
- iv) Erosion (rill and sheet) – leads to increase runoff and low/weak resistance to erosion.
- v) “Soft soil” – this phenomenon is often associated with the pocket (crack) in the subsoil hardpan layer and often noted in granary areas in Peninsular Malaysia.

Soil pollution from an agriculture perspective

Soil pollution can be defined as the accumulation of persistent toxic substances (compounds, chemicals, salts, radioactive materials, and other anthropogenic materials) in the soil, which can induce toxicity and adversely affect plant development and animal health. Pathways of pollution can include:

- i. Excessive use of herbicides, pesticides, and fungicides.
- ii. Excessive use of low-quality fertiliser (mixtures of unknown chemical compounds).
- iii. Seepage (effluent) from landfill.
- iv. Irrigation with polluted water (from industrial discharge/waste).
- v. Application of solid municipal waste.

In terms of agriculture, the major causes of soil pollution are often related to (i) and (ii) above, with secondary inputs from landfill-municipal-industrial discharge and seepage to nearby agriculture land.

As the demand increases for plant-based sustainable oil, producers of palm oil are placed under pressure to produce more oil at a lower cost. Indirectly, this may lead to pollution due to the discharge of palm-oil mill effluent (better known as POME) to river systems. Many plantation companies are recycling POME back to the palms; however, small factories may not adhere to environmental acts and policies. Enforcements are

constantly conducted, however, as in any other plantation and industry, lack of compliance remains as an issue. POME is not all bad, as researchers have found various methods to reuse and recycle POME (Tan, Bello, and Raman 2021; Lok, Chan, and Foo 2020). Olive oil mills face similar issues elsewhere, and studies conducted by Lee et al. (2019) and Vuppala et al. (2019) have shown remediation of wastewater from factories is a sustainable practice. Malaysia has also introduced proactive measures to sustainably manage POME, while Indonesia has focused on the use of POME for phytoremediation and the outcome from a study by Ilham Zulhami et al. (2021) is positive.

In SEA rice production systems, soil pollution can occur due to the improper storage and/or excessive use of herbicides and pesticides. These chemicals are used to ward off pests/rodents/snails, etc. that may reduce paddy yield. Improper storage or overuse of these chemicals may lead to seepage into the land and leaching into nearby water systems. These chemicals are often highly diluted and used in small doses to minimise human hazard and their excessive accumulation in the environment. Besides that, weeds and pests equally have a role in the ecosystem dynamics. The main purpose of herbicides and pesticides is not to totally eliminate weeds and/or pests, but to inhibit/ward off their presence during certain growth phases. Total disappearance of weeds and/or pest often leads to ecosystem imbalance. For example, in oil palm plantations, certain bees are pollinators while others are not. Improper use of pesticides will ward off/eliminate all the bee colonies without differentiation between helpful/harmful populations, and this is not favourable as it can lead to reduced pollination and yield.

A small portion of hazardous chemicals can have instantaneous effects on aquatic life. However, *Pila ampullacea*, better known as snails, are still able to thrive in paddy fields. In Malaysia, one species of snail, locally known as Siput Gondang Emas (SGE), is an issue in granary areas along the east coast of Peninsular Malaysia. A similar issue arises in Indonesia with *Pomacea canaliculate* (Lamarck), an invasive snail species (Wagiman and Bunga 2021). Whereas in Thailand and Vietnam, Ichinose, Chanyapate, and Hung (2001) studied control of invasive apple snail in paddy fields and suggested that chemical control was able to reduce pest attack, however, monitoring chemical use is equally important.

Heavy metal release from the excessive use of fertiliser is another ongoing issue. Arsenic (As), mercury (Hg), cadmium (Cd), and other heavy metals occur in trace quantities in fertiliser (Brady and Weil 2008) and can contaminate the soil environment (Zhou et al. 2015) and pose a threat to human health. Fundamentally, it raises one question - what are the threshold heavy metal values for soil and plants that are tolerable for humans? To

answer this for SEA countries, a coordinated effort between all stakeholders is required. In a reconnaissance geochemical soil and plant survey in Malaysia, Zarcinas et al. (2004) proposed a 95th percentile value for heavy metals in Peninsular Malaysian soil. The authors stated that heavy metals such as As, Cd, and Cu were strongly associated with organic matter, and available and aqua-regia soluble soil P, which was attributed to inputs from agricultural fertilisers and soil organic amendments (e.g., manures, composts).

In general, while the use of fertilisers and chemicals to control pests are necessary in agriculture production, they may also lead to soil pollution. Thus, monitoring, regulation, and the ethical use of such chemicals and fertilisers should be put in place for agriculture production to be sustainable throughout SEA.

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CHAPTER 15

CENTRAL ASIA

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Abstract

Soil is the basic medium for plant growth and a healthy soil is essential for sustainable crop productivity. Different soil constraints, such as soil salinity, waterlogging, and soil organic carbon (SOC) depletion, are major hindrances to sustainable crop production and severely affect the food security of many Central Asian countries. To resolve all soil related constraints and propose potential remediation approaches to maintain crop productivity, we reviewed recently published articles to identify the main cause of soil constraints and possible reclamation options in this region. This systematic study showed that soils in the region are mostly low in SOC, have limited soil nutrient reserves, and suffer from secondary soil salinisation and waterlogging due to improper water irrigation approaches. In addition, conventional cropping systems (including wheat, maize, and cotton) exacerbate soil nutrient depletion. Climate change, specifically the rising temperature in last few decades, is also likely to have decreased SOC and soil microbial diversity, thereby affecting soil productivity. The best possible reclamation options for soil salinisation, water logging, soil erosion, and nutrient depletion are soil conservation approaches i.e., no-tillage, residue retention, and crop rotation. These practices will mitigate soil salinisation and soil erosion and enhance soil fertility by improving SOC, soil structure, water storage in the root zone, and water infiltration. Increased water infiltration will carry soil salts (sodium, chloride) into deeper soil layers and improve growing conditions for plant roots. However,

the research progress in this region is very limited and further work is needed to improve soil and crop sustainability and eliminate expected food insecurity.

Keywords: Conservation agriculture; reclamation options; soil salinisation; sustainability; water logging.

Introduction

Central Asia is located in the centre of the Eurasian continent and consists of five major countries: Kyrgyzstan, Kazakhstan, Tajikistan, Turkmenistan, and Uzbekistan (Figure 15.1). The region has a diversified continental climate, with cold winters and dry hot summers. This region is classified as one of the largest semi-arid regions in the world, with a high variability of rainfall pattern. The region has large mountains and dry areas, and high degree of soil erosion due to intensified agricultural systems (Kienzler et al. 2012). Water shortage for agricultural production is the major reason for lower crop productivity.

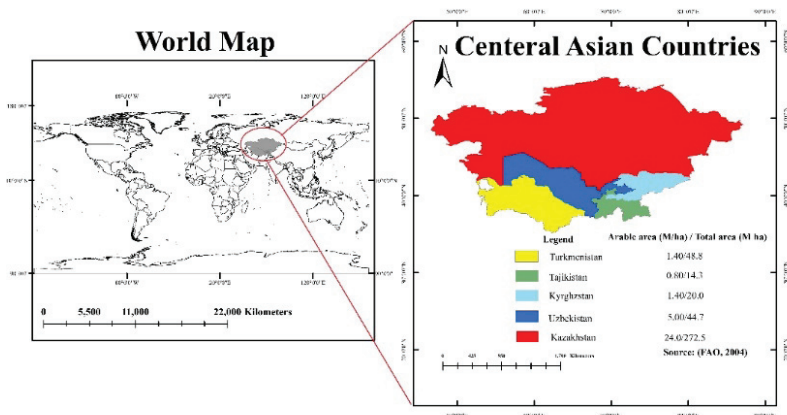


Figure 15.1 Geographical location of Central Asian countries.

Central Asian soils suffer from a variety of soil constraints, including soil erosion, nutrient depletion, loss of organic matter (OM), waterlogging, and soil salinisation (Lal 2004). The majority of soils in the region have low or poor fertility due to intensive agricultural practices and the production of exhaustive crops. Soil quality is also degrading over time due to increasing soil salinisation and desertification (Saparov 2014). Some reports suggest

that 40 – 60% of the irrigated croplands in Central Asia are salt-affected and the irrigated area affected by soil salinisation in Turkmenistan and Uzbekistan is almost 95% and 51% respectively (Kushiev et al. 2005). This situation is alarming for the future of sustainable agriculture and food security and will severely affect the productivity of the major fruit, vegetable, and broad acre crops that the region relies on (Table 15.1) (Hamidov, Helming, and Balla 2016). Excessive use of river (Amudarya, Syrdarya) water for irrigation is also elevating the water table and leading to secondary salinisation (Funakawa and Kosaki 2007). As a result, waterlogging and rapid soil salinisation is adversely affecting cotton and wheat (two major crops) growth (Djanibekov et al. 2016).

Collectively, Central Asia has a land area of almost 400 M ha, out of which only 32.6 M ha is used for arable crops (FAO 2017) (Figure 15.2). In Kyrgyzstan, half of the land (49%) is under meadows and pastures, 3% is with cultivated forests, and only 7% is used for growing crops. A large portion of meadows and pastures have been degraded due to over grazing and poor management practices. Of the arable land, almost 21% is used for wheat, 11% for barley, 7% for maize, 5% for potatoes, and 4% for beans (Table 15.1). Among the five countries of the region, Tajikistan has the lowest crop productivity due to degraded soil and poor irrigation management. In this country, one-fifth of the land is used for broad acre crop cultivation, with 36% and 24% of the cultivated area planted to wheat and cotton respectively, while the remaining area is used for minor crops, vegetables, and fruits (Table 15.1). In addition, out of the total land area of Turkmenistan, 72% of land is used for agriculture and livestock rearing and grazing. Of the area used for agriculture, only 4% is used for arable farming, with cotton-wheat being the dominant cropping sequence. In Uzbekistan, almost 40% and 60% of the agricultural land area is used for animal rearing and arable farming, respectively. The arable land in Uzbekistan is dominated by wheat, barely, maize, and rice. This specific zone in Central Asia has higher SOM and less soil salinisation compared to adjacent countries. In Kazakhstan 61% of agricultural land is used for pastures and meadows, while 32% is used for cropping. In the arable land, wheat, barley, cotton, and rice dominate, with some other minor crops like, carrot, cabbage, and melons also planted.

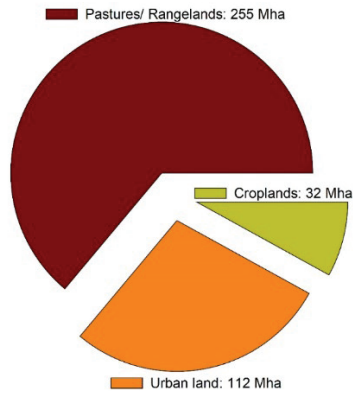


Figure 15.2 Land use distribution pattern of Central Asia (Data source: FAO 2017).

Table 15.1 An overview of the fertility status and major growing crops in Central Asia

Country	Soil fertility	Major crops	Reference
Kyrgyzstan	Low fertility	wheat, barley, maize, potatoes, oilseeds, sugar beet, cotton, tobacco, vegetables	Gupta et al. (2009)
Tajikistan	Higher altitude; fertile Lower altitude; low fertility	wheat, cotton, potatoes, vegetables,	Gupta et al. (2009)
Turkmenistan	Low fertility	cotton, wheat, fruit, and vegetables	Gupta et al. (2009); Berkum (2015)
Uzbekistan	Low fertility	wheat, barley, maize, grapes, apples, tomatoes, potatoes, rice	Gupta et al. (2009)
Kazakhstan	Low to medium fertility	wheat, barley, cotton, sugar beet, sunflower, flax, rice.	Gupta et al. (2009); Berkum (2015)

Across the globe, almost 30% of the land area has been affected by different soil constraints, resulting in economic and agricultural productivity loss. Likewise, Central Asian soils are experiencing land degradation, the causes of which are numerous, complicated, and interrelated (Simonett 2010; Nurbekov et al. 2016). Thus, proper identification of the drivers of land degradation and their solutions is required to mitigate soil quality depletion, improve agricultural sustainability, and mitigate climate change (Arshad et al. 2021). Mainly, overuse of irrigation water is converting Central Asian soil into a secondary saline area (FAO 1994). Some other problems such as heavy use of agro-chemicals and repeated cultivation patterns are also diminishing soil resources and soil nutrients status (Simonett 2010). Therefore, this chapter will identify the major causes of soil degradation as well as their possible solutions to improve soil fertility, sustainability, and crop productivity. The basic aim is to conserve soil and climate resources to improve productivity and eliminate food insecurity from this future climate threatened area.

Identified soil constraints in Central Asian soils

Crop productivity depends on several ecological factors, such as soil fertility, water availability, and growing environment. The best possible combination of all these factors will benefit soil fertility and yield establishment (Martius et al. 2012). Several studies have reported that identifying the important drivers of soil constraints is crucial. Central Asian soils are vulnerable due to following causes.

Soil nutrient depletion

Soil nutrients are a primary factor that can affect crop productivity and plant growth. Most soils have inherent nutrient concentrations and supply capacity during the crop growing season (Passioura 1991). In Central Asia, most soils are sandy in nature. Some soils are formed after rock weathering and have inherently low nutrient holding and storage capacity (Lioubimtseva et al. 2005). It is also well understood that sandy soils in Central Asia have been depleted due to soil erosion (Figure 15.3) (Juliev et al. 2019). Moreover, improper use of agricultural farm machinery, such as tillage equipment, has further depleted soil nutrients. As Lal (2004) explained, soil organic carbon (SOC) is an important soil indicator that can be decreased after land use change i.e., deforestation, removal of vegetation, and intensive tillage practices. These practices decrease the SOC pool in the soil and increase carbon (C) emissions due to organic matter mineralisation/oxidation

(Hamidov, Helming, and Balla 2016; Kienzler et al. 2012; Kan et al. 2020).

In addition, wheat and rice have been the main cereals crops grown in Central Asia for many years. These crops need huge amounts of nutrients due to their extensive root systems and huge biomass production and are also known as exhaustive crops (Nurbekov et al. 2016). Despite this, the study of Saparov (2014) highlighted that the use of mineral fertiliser has decreased from 1039 to 30.9 kilotonnes, and organic fertiliser application has also constantly decreased in Kazakhstan since 1986. There are several reasons for the low nutrient inputs of the farming community in Central Asia, including, low economic status, less availability of fertiliser, and the high price of inorganic fertiliser (Kienzler et al. 2012). This reduced inorganic fertiliser use could be creating a hidden hunger for plants on marginal lands and might be the reason for lower yield.

Irrigation water contamination, soil salinisation, and microbial diversity

Water resources on earth contribute significantly to global agricultural production. However, due to a rapid increase in industrialisation, many water supplies used for irrigation have become contaminated. Some studies have reported that almost 7.8 Mha land is dependent on irrigation water in Central Asia (Oberkircher et al. 2011), although fresh water resources have been decreasing due to rapid glacier melt (Sutton et al. 2013). Funakawa and Kosaki (2007) has reported the partial secondary soil salinisation in rice and cotton-based cropping in this region.

Specifically, water quality in Kazakhstan is poor due to heavy chemical usage, the oil industry, mineral extraction, and urban contamination. In addition, some water is contaminated with nitrogen (N) pollutants during storage in dams or reservoirs (Hamidov, Helming, and Balla 2016). In Kyrgyzstan, water quality is satisfactory as compared to Kazakhstan. However, some underground water is contaminated with lower levels of nitrates and heavy metals. The basic causes of water pollution in Kyrgyzstan are improper use of fertiliser, chemicals, industrial waste, and poor sewerage treatment (Karthé et al. 2017). Similarly, in Tajikistan ground and lake water resources also contain low to medium levels of salinity (Abdullaev and Rakhmatullaev 2015). The water sources of Turkmenistan are also heavily polluted with salinity (Table 15.2). The major identified polluting agents in this region are pesticides, salts, and industrial drainage. In Uzbekistan, the salinity levels of fresh water resources are medium, with some contamination by other organic pollutants (Karthé et al. 2017). However at certain times, the salinity level of water can rise exceptionally

due to the discharge of sewage and municipal waste water (Karthé, Chalov, and Borchardt 2015).

Table 15.2 An overview of the water salinity in Central Asian countries

Country	Water status	Salinity
Kazakhstan	Poor	0.12-2.7 g/l
Kyrgyzstan	Satisfactory	-
Tajikistan	Medium	0.05-0.40 g/l
Turkmenistan	High	1.5-2.00 g/l
Uzbekistan	High	1.1-2.00 g/l

Data source : Aquastat 2012

Soil salinisation, or salt accumulation in soil, is one of the major reasons for lower crop productivity in Central Asia. Lower productivity is mainly due to the imbalance of osmotic pressure in plant tissues (Funakawa and Kosaki 2007). Because this is a semi-arid region, where precipitation is less than evapotranspiration, salts are not washed downwards and accumulate in the rooting zone (Kushiev et al. 2005). The highest proportion of arable land affected by salinity is in Turkmenistan (Figure 15.3). Most of the arable area of Kazakhstan is also affected by sodicity, with some smaller areas also affected in Uzbekistan, Turkmenistan, Tajikistan, and Kyrgyzstan (Figure 15.3). Soil salinity and sodicity levels have increased in last few years due

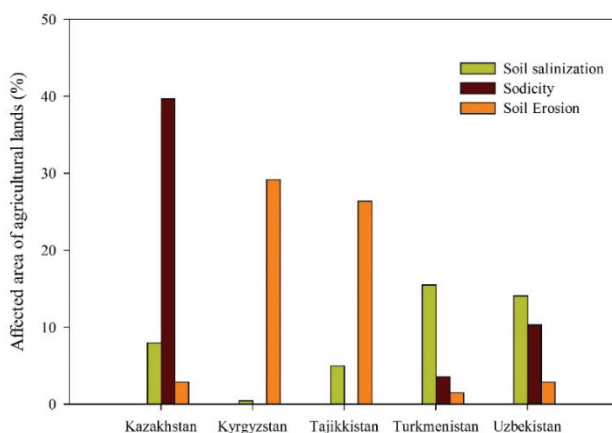


Figure 15.3 Area affected by different soil constraints in Central Asian countries (Data sources : FAO 2017; ISRIC 2006; Qushimov et al. 2007).

to intensive and improper agricultural practices (Simonett 2010). A higher rate of secondary salinisation due to the application of contaminated water is also an emerging threat to soils of this region (Lal 2004). In addition, almost 43 MT of salts and other contaminants are blown out of the Aral Sea basin and spread over 1-2 million square kilometres by the wind, also adding salt to the soil (Létolle and Chesterikoff 1999). Due to the semi-arid characteristics of the region and continuing deforestation, this region is prone to soil salinity and turning into non-productive land (Simonett 2010). Therefore, it is critical to overcome this issue with economic options.

Soil microbes play a significant role in soil ecosystem functions and nutrient transformations. They provide soil nutrients to plants and affect nutrient transformations between different soil layers (Legay et al. 2014). The microbial community is also a sensitive soil health indicator that changes with soil chemical and physical properties (Oren 2002; Mamilov et al. 2004).

Salinity significantly affects global bacterial diversity more than other soil chemical characteristics (pH, nutrient concentrations) (Lozupone and Knight 2007). While the salinisation of Central Asian soils is well recognised, little information is documented about the effects of salinity on microbial community distribution and abundance (Funakawa and Kosaki 2007). However, it is expected that the behaviour of microbes around the world would also be similar in Central Asia. A study of Zhang et al. (2019) showed that increasing soil salinity decreased root colonisation and microbial biomass carbon and nitrogen. In addition, several studies have demonstrated that soil salt stress decreased SOC mineralisation, soil enzymatic activities, and microbial biomass (Mamilov et al. 2004). Most soils in Central Asia are sandy and receive low levels of precipitation, which could cause osmotic stress in microbes and result in drying and lysis of microbial cells (Oren 2008).

Waterlogging

In the Aral Sea basin (45°N 60°E) and surrounding area, waterlogging problems have increased due to the excessive application of irrigation water irrespective of crop requirements and excessive water application for the leaching of salts (Létolle and Chesterikoff 1999). Among all five Central Asian countries, Uzbekistan has the highest (1 Mha) well drained area. However, in general, Central Asian countries do not have well managed drainage systems for waterlogged soils. In addition, improper and over irrigation management practices have increased waterlogging (Haleemzai and Sediqi 2018). Excessive irrigation and poor drainage cause waterlogging

and produce anoxic condition that can affect growing plant roots, resulting in abnormal plant growth and reduced productivity (Ashraf 2012). For example, the mishandling of irrigation operations in Kyrgyzstan, Tajikistan, and Turkmenistan has produced a waterlogged area of 35,399 ha, 25,742 ha, and 756,500 ha, respectively (Shi, Wang, and Guo 2014).

Soil erosion

Strong winds and water from runoff in arid and semi-arid regions cause the removal of fertile surface soil layers and deplete soil fertility (Qushimov et al. 2007). Several factors have been recognised as favouring wind and water erosion, including excessive tillage practices, deforestation, and over grazing (Lal 2004). Most soils in this region also have low SOM concentrations. The presence of SOM not only binds soil particles but protects against soil erosion by improving soil structure (Lal 2004; Virk et al. 2021). Other factors, for example steep slopes and the removal of vegetation cover, also contribute to increased soil erosion risk (Kienzler et al. 2012). Furthermore, due to the impact of climate change, changes in precipitation patterns are expected in the region, along with glacier shrinkage and changes to flooding patterns (IPCC 2007). This is likely to increase the intensity of flooding and increase water erosion. However, not many studies have quantified the extent of the land area affected by wind and water erosion in the region.

Climate change impacts on soil biology and agricultural productivity

In the past few decades, climate change has severely affected most world regions (Kessel et al. 2013). Similarly, Central Asia has also been affected by elevated temperature and carbon dioxide levels (Figure 15.4). Indeed, an obvious increase in the average temperature of all countries has been observed as compared to previous decades (Lipper et al. 2014). Of all the Central Asian countries, variability in the average temperature of Kazakhstan is the highest, followed by Turkmenistan and Uzbekistan (Figure 15.4). Rising temperatures are creating significant hurdles for sustainable crop production in Central Asia (FAO 2020). The early consequences of climate change have been examined mostly in regard to the agricultural sector and farmers' livelihoods (Arshad et al. 2021).

This region is characterised by low rainfall and high temperature during the summer (Lioubimtseva et al. 2005). Consequently, agricultural production often depends on irrigation, with most water supplies coming from glacier

melting. However, the shrinkage of glaciers in recent years due to temperature increase has caused water shortages and poses a serious threat to agricultural productivity in some regions (Qushimov et al. 2007; Pradhan, Idol, and Roul 2016). In addition, the dryness caused by water shortages has resulted in increased soil salinity due to reduced profile leaching and the increased accumulation of salts (Funakawa and Kosaki 2007). Increasing salts in the soil could deplete the soil microbial community, SOC, and nutrients. Some countries are trying to adapt to the arising climate problems, but their limited resources to implement adaptive strategies is a major issue in the region.

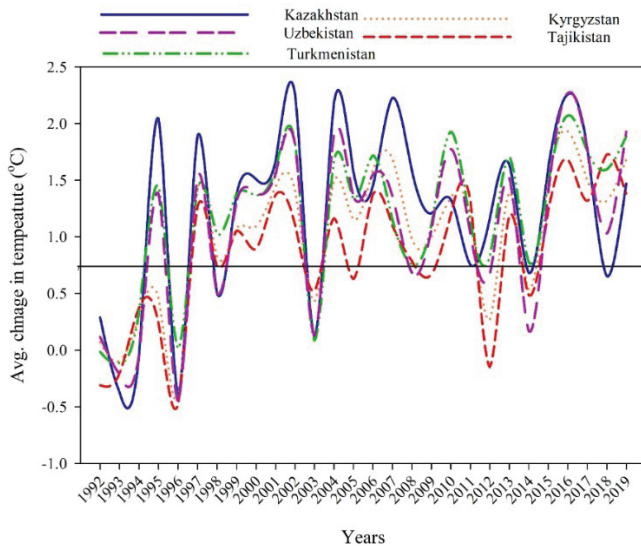


Figure 15.4 Changes in average annual temperature in five Central Asian countries (Data source: FAO 2020).

Possible solutions for sustainable crop production

Sustainable crop production in Central Asia heavily depends on soil fertility and water availability. As such, integrated nutrient management practices, the maintenance of organic matter and microbial biodiversity, and a reduction in intensive agricultural practices to improve plant nutrient availability will all be important to maintain productivity. This section highlights some of the most promising approaches to increase/maintain soil fertility and crop productivity by mitigating soil constraints in Central Asia.

Crop rotation

Crop rotation is a cost-effective method to improve soil nutrient status and crop yield. Mono-cropping of cereals can deteriorate soil structure, fertility, and microbial diversity (Li et al. 2019). Specifically, mono-cropping in the salt affected soils of Central Asia imposes negative impacts by depleting soil nutrient reserves and affecting the soil microbial community (due to nutrient depletion). Some halophytic crops could be useful to mitigate salinisation by taking up accumulated salts (Farooq et al. 2017). The strategic inclusion of legumes in the cropping system can also help to improve soil nitrogen concentrations, SOM, and microbial abundance (Virk et al. 2021), and sustain crop productivity. Although legumes are sensitive to saline conditions, and their inclusion in rotations in saline areas can be challenging, alternative practices, such as hydropriming (the soaking of seeds in water before sowing), can be used to mitigate salinity impacts on the early growth of legumes (Farooq et al. 2017; Srivastava et al. 2017). Suleimenov et al. (2005) suggested that besides cotton, maize, and wheat, other crops such as mungbean, common bean, buck wheat, chickpea, field pea, and sugar beet could be alternative options in this region to improve soil productivity. This has been supported by studies where the growing of legumes has positively affected soil chemical properties like SOM, N, and soil bulk density (Lal 2004; Suleimenov et al. 2005). However, some authors have emphasised that crop rotations must be adapted to the socioeconomic and climatic conditions specific to individual regions, and it is not possible to recommend species that will be suitable across the region as a whole (Chen et al. 2018; Yu, Xue, and Yang 2014).

Conservation tillage

The use of conservation tillage is increasing worldwide. The main purpose of tillage is to prepare land to facilitate seed germination, increase nutrient availability, and improve soil structure (Lal 1989). However, conventional tillage also inverts the subsoil and exposes deep layer accumulated SOM to field macro environmental conditions, resulting in increasing SOM oxidation and nutrient depletion (Qi et al. 2020). Repeated tillage operations can also be harmful under some field conditions, such as where salts (sodium, chloride) accumulated in deeper layers are brought to the surface.

In Central Asia, research focus has been on testing the impact of conventional tillage vs minimal tillage (Kienzler et al. 2012). Minimal tillage has been demonstrated as a useful practice to conserve soil nutrients,

water, and SOM. However, deep tillage practices, such as chisel plough or mould board plough, can be useful to increase water infiltration and salt movement to deeper soil layers. A study by Suleimenov et al. (2005) showed that weeding problems in fallow lands should be managed with herbicides instead of surface hoeing operations, because soil manipulation increased soil evapotranspiration and salt accumulation. Additionally, Nkonya, Mirzabaev, and von Braun (2015) suggested that direct seeding or minimal tillage is a better strategy for rainfed wheat and barley for southern part of Kazakhstan. This practice not only reduces farming expenses but also conserves rainwater efficiently. Similarly, Kienzler et al. (2012) found that the grain yield of wheat sown in a no-tillage system in a rainfed area was about 0.5 t ha⁻¹ higher as compared to conventional tillage practice. However, in the irrigated regions, Lioubimtseva et al. (2005) concluded that crop sowing under no-tillage was not as successful as conventional tillage due to various factors including, crop type, land management, and nutrient application approaches. However, overall, conservation tillage is considered a useful approach throughout the region to improve soil nutrients, water storage, and water infiltration in different soil layers.

Management practices for waterlogging

Excessive water in the root zone produces anoxic conditions and affects plant growth. Some studies around the world have demonstrated that pumping groundwater could be a useful option to mitigate waterlogging (Saparov 2014; Pradhan, Idol, and Roul 2016). Similarly, useful and cost-effective management practices such as surface drains and raised seedbeds could be used in this region to mitigate waterlogging stress and improve plant growth. Surface drains are currently used in many areas to carry away extra water received after precipitation, although digging these drains is laborious and time consuming. Raised seedbeds are also an alternative, and less labour-intensive approach, as they only require planting beds to be raised about 0.2-0.5 m above the mean surface level. These raised beds provide an adequate amount of water to growing plants, while allowing excess water to flow into drains in between the planting rows. However, this region requires further innovative and artificial drainage control systems to reduce waterlogging in irrigated areas (Benduhn and Renard 2004; Loodin 2020).

Residue retention

Crop residues left after harvest carry nutrients e.g., N, phosphorus, potassium, and a large portion of the crop assimilated C (Redin et al. 2018).

Carbon sequestration in agriculture land through residue retention can enhance food production by increasing soil fertility and decreasing global warming risk through C increment (Yadav et al. 2019). In contrast, the burning of crop residues can lead to declines in SOM and fertility and increase the risk of wind and water erosion. However, these problems can be addressed with proper residue retention. Residue retention improves soil structure (particle binding), reduces wind and water speeds at the soil surface and thus reduces wind and water erosion, and also improves soil fertility (Griffin, Matt, and Jemison 2000).

To mitigate secondary salinisation problems in Central Asia, Vlek et al. (2012) also found that tillage and residue retention in conjunction with drainage could decrease secondary salinisation by increasing infiltration rates and helping to wash salts below the root zone, providing water tables were well below the soil surface. But the impacts of washing salts into the water table have not been reported. However, it should be noted that residues of wheat and maize may need to be applied with additional N fertiliser to avoid N immobilisation and yield decrease (Douglas et al. 1980; Urquiaga et al. 1998). Alternatively, legume cultivation and the retention of residues with lower C/N ratios can favour N mineralisation (Virk et al. 2021). Green manures with legumes increase soil N and decrease the gaseous loss of N as compared to N fertiliser (Sajjad et al. 2019). Moreover, green manuring also improves the chemical, physical, and biological characteristics of soil.

Integrated nutrient management

Various regions of Central Asia are characterised by poor fertility and low agricultural productivity. The major causes of soil fertility decline in croplands are improper crop rotations and the imbalanced use of chemical fertilisers (Qushimov et al. 2007). For example, the improper timing of fertiliser applications decreased nutrient use efficiency (NUE) in cotton production by almost 22% in Uzbekistan (Kienzler, Djanibekov, and Lamers 2011). Decreased SOM and nutrients also directly affect soil microbial life, and result in low crop productivity (Lal 2018). Therefore, the proper management of nutrients should be used to enhance and maintain soil fertility. Integrating different nutrient use techniques can sustain crop productivity on a long-term basis.

Several studies have found that only chemical fertiliser application or only organic sources are not sufficient to improve soil fertility and crop yield (Blair and Crocker 2000; Zhou et al. 2019). Even the balanced use of macro nutrients cannot supply sufficient micronutrients. Thus, the combined

applications of organic and inorganic of fertilisers may be required to stabilise soil fertility and crop nutrient requirements. Application of proper fertiliser sources at the proper time can boost crop yield and soil fertility status (Zhou et al. 2020). Intensive cropping systems, like cotton-wheat in Central Asia, deplete soil nutrients quickly as compared to other cropping systems (Huang, Birch, and George 2006). Therefore, these cropping systems should be managed with integrated nutrient management approaches instead of the overuse of chemical fertilisers. Qushimov et al. (2007) observed that the use of inorganic fertilisers has reduced in all Central Asian countries by 50-80%, but the single use of N fertiliser has increased since 2002. This suggests that the overuse of N fertiliser could increase water pollution (eutrophication) and greenhouse gas emissions (Wang et al. 2017). There must be strategic policy for efficient inorganic fertiliser use in combination with other organic sources (bio-fertilisers, farmyard manures, etc.) to maintain yield and environmental sustainability.

Future research needs

The research framework in Central Asia is weak due to limited resources and economic issues. But there must be further developments in research planning and the implementation of research results. The following are some key research priorities to develop agriculture sustainability.

Land rehabilitation

Central Asian soils are experiencing many soil problems i.e., soil salinity, waterlogging and rapid nutrient depletion. Among all the problems, soil salinity is one of the major causes for land degradation and lower crop productivity (Lerman and Sedik 2008). Soil affected by salinity needs some special efforts to make it cultivable. Some agronomic practices, such as residue retention, have been identified as mitigation approaches for soil salinisation (Kienzler et al. 2012; Martius et al. 2012). However, the combined effects of all other practices on soil salinity have not been previously investigated. The findings of these types of studies would be helpful to resolve soil salinity problems and identify further research pathways.

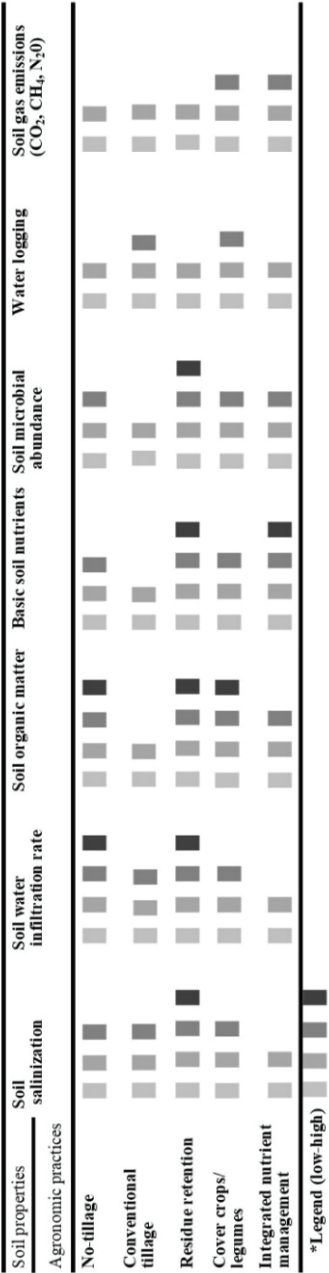
In addition, some studies in the area have reported on the optimal use of N in irrigated zones and specific cropping systems (Kienzler, Djanibekov, and Lamers 2011). But not much research has been done related to the optimal use of different combinations of chemical fertilisers under diversified cropping systems. This type of investigation will not only identify the best

chemical fertiliser application combinations, but also will be helpful to propose different types of fertilisers in different cropping systems.

Currently, very limited options are available to the farming community to remediate soil problems like soil salinisation, waterlogging, and nutrient depletion (Djanibekov et al. 2016). Indeed, given that the success of conservation agriculture is often dependant on how well it is adapted to the local physical and socioeconomic environment, it is essential to identify production systems that have been optimised for local conditions (Qushimov et al. 2007). The rate of research into conservation agriculture and fertiliser use techniques is very slow compared to the expanding areas of salinity, waterlogging, and other problems (Lerman and Sedik 2008). At the same time, the population is increasing, and food insecurity issues need to be resolved urgently. One of the case studies above suggested that SOM increment could be useful to rehabilitate degraded land in Central Asia. Moreover, this study suggested that adoption of conservation agricultural approaches such as no-tillage, crop rotation, and improved washing of salts from the root zone could be helpful to increase crop productivity (Figure 15.5). However, all these practices need to be verified by experimentation on a regional scale.

Climate change adaptation strategies for soil health

Temperatures are increasing in most of the regions of Central Asia and are expected to increase by a further 3–4°C up to 2099 (IPCC 2007). Hagg et al. (2007) used five different models to simulate climate change scenarios in coming years, with a temperature and rainfall increase of 4.2°C and 17%, respectively. An increase in surface temperature will impact the soil hydrological cycle and soil fertility by altering evapotranspiration rates and the oxidation of SOM, respectively. Carbon sequestration in the soil is intricately linked to soil and crop sustainability. If the temperature rises each year, it will increase CO₂ concentrations by oxidizing SOM. Some reports have suggested that the Amu Darya and Syr Darya basins will lose 40% and 28% their runoff due to elevated CO₂ levels (Hagg et al. 2007; Haleemzai and Sediqi 2018). Therefore, these issues need to be mitigated with SOM accumulation strategies, such as residue retention and green manuring with legumes. The basic purpose of increasing SOM in these degraded lands is to increase available water storage in the root zone by increasing water infiltration and reducing soil temperature. In this respect, farming techniques to increase SOM should be tested in this region to mitigate climate change challenges. Other adaptation options for successful crop production and the elimination of food insecurity should also be sought.



Conclusions

This study highlights the major soil constraints in Central Asia and possible solutions to mitigate these for sustainable crop production. This systematic literature review has identified that most of the soils in this region are suffering from secondary soil salinisation, nutrient depletion (due to conventional and extractive cropping systems), erosion, waterlogging, and loss of SOC and soil biodiversity. Furthermore, this study proposed some remediation approaches based on previously published research, as follows.

- The implementation of no-tillage with residue retention could increase SOC (and soil microbial activities), root zone water storage, and water infiltration rates to help economically reclaim areas affected by secondary soil salinisation.
- Soil structure and surface cover can be improved with conservation agriculture approaches, and these would be helpful to reduce soil erosion in rainfed areas.
- Diversified cropping systems (including legumes) instead of conventional/ extractive cropping patterns would be helpful to improve soil fertility.
- Innovative management practices, such as surface drains and raised beds for crop planting, are required to help improve production in waterlogged areas.

Research into appropriate responses to soil constraint issues is very limited in all parts of Central Asia, even though information specific for local farming communities is required. More research work is essential for the proper implementation of appropriate land use management strategies.

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CHAPTER 16

SOIL CONSTRAINTS AND THEIR MANAGEMENT FOR CROP PRODUCTION IN SOUTHERN ASIA

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Abstract

South Asia (SA) is known for its contrasting climate and geographical settings/landscapes. Rice, maize, and wheat are the most important cereals, contributing to the food and nutritional security of the region. However, rapid urbanisation and changes to socio-economic conditions have led to adverse impacts on agro-ecological environments and soil resources. Soils in SA are degrading rapidly due to water- and wind- erosion, and chemical and physical deterioration. Soil chemical constraints, such as loss of soil organic carbon (SOC), acidification, salinisation, sodification, acid sulphate soils, and heavy metal contamination are prevalent, and need appropriate management for sustainable crop productivity and soil health. Climate change also poses a considerable threat to food security in the region. It directly impacts food production through crop yield changes, typically due to temperature rise and increasing rainfall variability. This chapter addresses the important issue of soil constraints and their management for crop production in the SA region. Management options to address soil constraints and minimise their impact on crop productivity are discussed.

Many of these, such as carbon sequestration in soil through brown manuring, conservation tillage/agriculture, mulching, and soil biodiversity improvement, can also play an important role in mitigating climate change and sustaining soil health.

Keywords: Soil constraints, crop productivity, nutrient imbalance, carbon sequestration, soil health, greenhouse gas emissions, climate change.

Introduction

South Asia (SA) is comprised of a total of 8 countries, namely India, Sri Lanka, Bangladesh, Afghanistan, Nepal, Bhutan, Pakistan, and the Maldives. The SA region has a total population of 1.7 billion people and has been identified as the most populated region in the world (Lal 2006). Out of these 8 countries, India has the largest geographical area as well as coastline (~2/3rd area). SA is characterised by diverse climates, soil features, and water resources that affect the form and type of agriculture (Somasundaram et al. 2020). SA agriculture is not only the major contributor to the region's economy, but also ensures food and nutritional security/needs. Therefore, understanding and conservation of land resources is the key for the development and prosperity of these countries.

The diverse climatic conditions across the region, coupled with extreme landforms, can lead to severe land degradation, particularly in areas where the impact of steep topography is accelerated by anthropogenic activities. Intensive farming practices that exploit the soil further exacerbate the condition of SA land. According to recent research, land degradation will continue as a consequence of a predicted three to six-fold increase in global GDP by 2050, even while global populations are expected to level off (UNDP 2005; Dumanski et al. 2013). Therefore, sustainable soil management (SSM) is essential. This can be achieved through efficient nutrient management and appropriate soil conservation measures/practices (Acosta-Martinez and Cotton 2017). Therefore, this chapter focuses on the impact of soil constraints on crop productivity in SA and their management to improve the sustainable use of soil resources.

The climate and agroecology of South Asia

Climate

South Asia has contrasting climate and geographical settings/landscapes. This chapter mainly considers India, Pakistan, Bangladesh, Sri Lanka,

Nepal, and Bhutan, which are located between 5° and 37° N, in tropical and equatorial latitudes. The principal boundaries of SA are the Indian Ocean in the south, the Great Himalayas to the north, the meridional chain of mountains in the east and the Baluchistan highlands in the west. These provide distinct climatic and geographical features. SA is divided into four climate zones, namely i) a dry continental and subtropical climate along the northern Indian border and the upland and mountainous areas of Pakistan; ii) an equatorial climate in the far south of India and southwest Sri Lanka; iii) a tropical climate in northwest India and Bangladesh; and iv) a semiarid climate in the centre-west of the Indian peninsula (Rao 1981; Martyn 1992).

South Asia's precipitation is highly variable throughout the region. In the driest areas of northwest India and Pakistan, annual precipitation is less than 250 mm. In contrast, annual precipitation in southwest Sri Lanka exceeds 2,200 mm per year. Due to the movement of the inter-tropical convergence zone (ITCZ), there are two rainy seasons in southern India and southwest Sri Lanka. The first one is from May to July (South-west monsoon), and the second one is from October to November (North-east monsoon).

Agroecology

Agriculture is the primary source of income in SA and the majority of farmers are small and marginal. In India and Pakistan, 80% and 58% of farmers, respectively, have land holdings of less than 2 ha (Gumma et al. 2020). Over 90% of farmers in Bangladesh and Nepal have less than 1 ha of land. The average farm size in Bhutan is slightly over 1 ha, and women own roughly 70% of all agricultural land. In terms of both net and gross area, SA has one of the largest cropland areas of the world (541 M ha) (Thenkabail and Wu 2012; Teluguntla et al. 2015). Both irrigated and rainfed agriculture are prevalent, accounting for 35% and 65% of total cropland area, respectively.

Rice, maize, and wheat are the most important cereals, contributing to both the food and nutritional security of the region. These are also exported to other countries (Timsina, Jat, and Majumdar 2010). Crops are typically grown as monocultures or in a crop rotation system. Rice-rice (R-R), rice-wheat (R-W), and rice-maize (R-M) are the most common cropping systems in irrigated and rainfed lowland areas (Jat, Saharawat, and Gupta 2011). The major cropping systems of SA are presented in Table 16.1. Summer (*kharif*) from June/July to September/October, winter (*rabi*) from October/November to February/March, and spring (pre-monsoon) from March/April to May/June are the three major cropping seasons (Jat,

Saharawat, and Gupta 2011). In the *kharif* season, rice is the main crop while wheat, maize, chickpea, lentil, field peas, potatoes, and mustard are grown in the *rabi* season (Jat, Saharawat, and Gupta 2011). However, rice is grown as a winter crop in Bangladesh (also known as Boro rice), eastern India, and eastern Nepal.

Table 16.1 Major cropping systems in South Asian countries

Cropping system	Area (Mha)				Total
	Bangladesh	India	Nepal	Pakistan	
Rice–rice	4.5	4.7	0.3	-	9.5
Rice–rice–rice	0.3	0.04	-	-	0.04
Rice–wheat	0.4	10.5	0.57	2.2	13.67
Rice–maize	0.35	0.53	0.43	NA	14.98
Maize–wheat	-	1.8	0.04	1	2.84
Rice–pulses	-	3.5	-	-	3.5
Rice–vegetables	-	1.4	-	-	1.4
Millet–wheat	-	2.44	-	-	2.44
Rice–potato	0.3	NA	-	-	0.3
Cotton–wheat	-	2.6	-	3.1	5.7

Source: Timsina, Jat, and Majumdar (2010) and Jat, Saharawat, and Gupta (2011)

Impact of climate variability on SA agriculture

Climate change poses a considerable threat to SA food security. It directly impacts food production by adversely affecting crop yields, typically due to temperature rise and rainfall variability (Mahato 2014). Climate change also adversely affects soil health by reducing soil organic matter content (Mohanty et al. 2020), decreasing soil structure, and increasing vulnerability to erosion and other degradation processes (Brevik 2013). It also reduces the nutritional quality of agricultural produce. Recent research has shown that under highly elevated atmospheric carbon dioxide (CO₂) concentrations, iron and zinc contents in plants could fall by as much as 17%, accompanied by an increase in starches and sugar production (Myers et al. 2015).

Climate change and rainfall variability accounts for nearly 60% of yield variability in SA, making it a critical factor in food production and farmers' income (Ray et al. 2015; Matiu, Ankerst, and Menzel 2017). The average

annual maximum temperature in SA is expected to rise by 1.4–1.8 °C by 2030 and 2.1–2.6 °C by 2050, resulting in a 12 % increase in heat-stressed areas in 2030 and a 21% increase in 2050 (Tesfaye et al. 2017). According to climate projections, nearly half of the Indo-Gangetic Plains (IGP) will become unsuitable for wheat production by 2050 due to heat stress (Ortiz et al. 2008). As a general rule of thumb, in the equatorial tropics every 1 °C rise in mean temperature is associated with a 10% drop in crop yield (Brown 2008). Water availability is also expected to decline due to climate change, while agricultural water consumption is predicted to increase by 19% by 2050 (WWAP 2012). Increases in the prevalence of droughts and floods are also expected to affect economic productivity and threaten overall food security across the region. Besides its impact on crop production, climate change also significantly affects the sustainable use of natural resources.

Soil constraints and their management

The large increases in food production in south Asian countries during the last 50 years enabled the region to support large populations from limited arable land. But rapid urbanisation and changes in socio-economic conditions have negatively impacted the agro-ecological environments of the region (Masters et al. 2013). This, coupled with climate change, is threatening soil resources. The interaction between the causes, factors, and processes of soil degradation exacerbates the process of soil degradation (Lal 2015). The causes of soil degradation include intensive tillage, faulty irrigation and nutrient management practices, deforestation, land use and land cover adoption, and inappropriate livestock management practices. Factors of soil degradation include climate, physiography, landforms, and socio-economic factors. The principal mechanisms or processes of soil degradation include compaction, erosion, acidification, salinisation, sodification, heavy metal contamination, and nutrient depletion. These soil degradation processes feed on themselves in an ever-increasing downward framework (Figure 16.1). To restore soil quality and mitigate soil degradation, it's crucial to understand the process-factor-cause nexus.

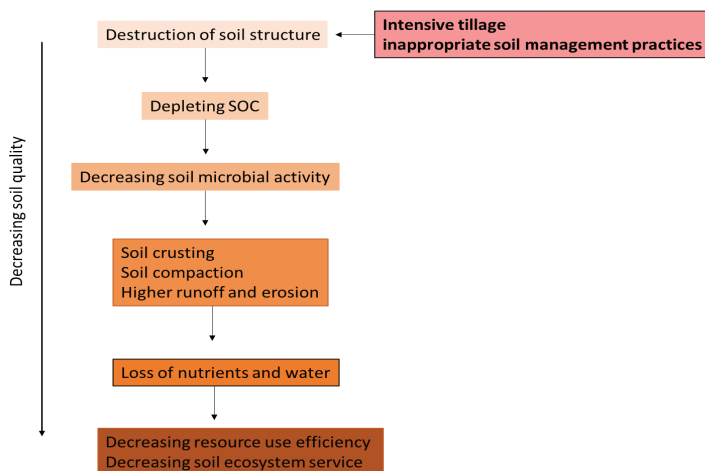


Figure 16.1 Decreasing soil quality due to accelerated soil degradation (Source: modified from Lal 2015).

Soil chemical constraints in SA

Soil chemical degradation is characterised by declining organic carbon, acidification, salinisation, solonisation, nutrient imbalance, and heavy metal pollution. The following sections address the causes, impacts, and management of the major chemical constraints in SA.

Depletion of soil organic carbon (SOC)

The SOC content is the defining constituent of soil, both in terms of quantity and quality (Manlay, Feller, and Swift 2007; Krupenikov et al. 2011). The SOC content is the most reliable indicator of soil degradation (Rajan et al. 2010). Plant available nutrients, including primary, secondary, and micronutrients, are depleted as a result of soil degradation. Developing strategies to ensure that the SOC pool rises above the threshold, or critical level of 10 to 15 g kg⁻¹, is essential for minimizing soil degradation risks (Lal 2015).

The data pertaining to SOC change in SA is not adequate, as SOC has not been monitored consistently over time. However, it is known that SOC is declining (FAO and ITPS 2015). This is due to the removal of crop residues for fuel and fodder, and the fact that they are not returned to the soil. Soil organic carbon levels in most soils of SA are extremely low, ranging from 8 to 10 g kg⁻¹ (Lal 2004). Fertility exploiting practices and

land degradation processes such as intensive tillage, increased deforestation, desertification, soil burning, and nutrient mining contribute to the depletion of the SOC pool. Long-term research sites across SA have seen a significant reduction in crop yield, indicating loss of SOC in this region (Ladha et al. 2003; Hobbs et al. 2008). Reductions in stable soil aggregates aid in the development of compact layers beneath the tillage depth, and severe soil cracking in highly puddled rice soils indicates low SOC content in the soil profiles of SA (Hobbs et al. 2008). Depletion of SOC causes a decline in soil structure, poor soil aggregation, vulnerability to crusting and compaction, and poor soil fertility. The effects of depleting SOC and the associated soil constraints to crop productivity are presented in Figure 16.2.

Restoration of degraded soils and ecosystems, agroforestry and afforestation, grazing management, and adoption of best management practices (BMPs) on cropland are some of the sustainable strategies for increasing SOC (Lal 2003). Some recommended BMPs are conservation tillage, application of organic manure, integrated nutrient management, green/brown manuring, and inclusion of legumes in crop rotations. In addition to these practices, soil water conservation, and water harvesting and recycling, are effective strategies for increasing plant biomass production and SOC (Ghimire et al. 2016).

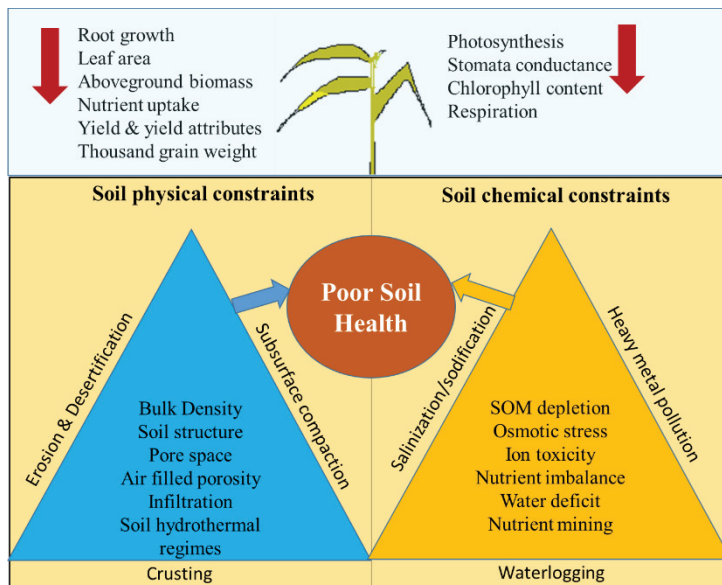


Figure 16.2 The effect of soil constraints on crop productivity in South Asia

Heavy metal contamination

Heavy metal contamination in SA agricultural soils can be naturally occurring, or due to anthropogenic activities including mining (grinding, concentrating ores, tailings disposal, smelting, fly ash production), and the application of agrochemicals, fertilisers, wastewater irrigation, sewage-sludge application, and livestock feed additives. Heavy metal contamination can be a significant issue for both crop production and human health in SA. Norra et al. (2005) reported the concentration of arsenic (As) in winter wheat grain grown in agricultural areas of the west Bengal Delta Plain that had been irrigated with As rich groundwater could reach 0.7 mg As kg⁻¹. By interfering with photosynthesis, water and mineral uptake, and N metabolism, excessive heavy metal uptake by plants can reduce crop productivity (Austruy et al. 2014, Shahid et al. 2014, Hasanuzzaman et al. 2014).

Heavy metal pollutants can be remediated by conventional processes such as land filling, fixation of contaminants into immobilised forms, and leaching. Recently, due to the advancement of technologies, biological, physical/chemical remediation, and integrated remediation measures have also been employed for the systematic disposal of heavy metals. The SA countries must act cooperatively towards soil environmental protection and enhance capacities for the management and technological innovation that will allow the management of heavy metal contamination across the region.

Soil acidification

Acidification is a natural process that occurs in high-rainfall areas as a result of the leaching of basic cations. Substantial increase in rainfall can cause increased leaching of basic cations and increased acidification, while rainfall decreases should reduce the intensity and extent of acidification (Jat et al. 2018). Under projected climate change scenarios, warm temperatures and reduced total rainfall in wheat-growing regions of SA do not pose a serious threat to soil acidification in arid and semi-arid areas. However, using too much urea to supply nitrogen to rice and wheat, particularly in an unbalanced ratio with P and K, can cause soil acidification (Jat et al. 2018). The cost-effective technologies for reclaiming soil acidity problems include liming, organic matter application, balanced fertilisation, and the selection of acid tolerant crops. Moreover, soil acidifying farming practices, such as the application of acidifying fertilisers like ammonium sulphate, should be avoided (Kamprath 1984).

Soil salinisation and sodification: natural or anthropogenic in SA agriculture?

Soil salinisation is a major issue in SA due to the arid and semi-arid climate and because of its negative effect on agricultural production and sustainability. This problem can arise due to both natural and human-induced factors (Shahid, Zaman, and Heng 2018). Saline soils can naturally occur in arid and semi-arid regions where evapotranspiration exceeds rainfall and salt accumulates in the soil profile (Mustafa, Akhtar, and Abdullah 2018). In SA, the region affected by soil salinisation is estimated to be 33 Mha (Srinivasarao et al. 2019). India has the greatest area (7 Mha), followed by Pakistan (4.2 Mha), Afghanistan (3.1 Mha), and Bangladesh (3 Mha) (Lal 2004).

In irrigated areas, salts can also enter the soil profile with irrigation water and, if not leached out, accumulate in the soil profile, particularly under conditions of high evapotranspiration (secondary salinisation) (Jat et al. 2018). As a result, regions that depend heavily on irrigation for crop production are vulnerable to soil salinity (Brady and Weil 2008). These areas will be even more affected when temperatures rise and rainfall decreases under climate change. On irrigated lands of the Mahaweli scheme in Sri Lanka, small areas of light salinisation have appeared; however, the problem has not yet reached serious levels, although the area needs periodic monitoring and management. In parts of northwestern India (Punjab, Haryana, and Rajasthan), secondary salinisation is a widespread problem. It is caused by excessive canal-based irrigation, the use of brackish irrigation water, and a lack of drainage (Lal 2010).

Plant water and nutrient uptake decrease as salt accumulates in the soil solution. Excessive salt ion concentrations damage photosynthetically active leaves, causing chlorosis and early leaf senescence (Hanin et al. 2016). Crops cultivated in salt-affected soils experience osmotic stress, ion toxicity, nutrient imbalances, and physiological drought (Mittal, Kumari, and Sharma 2012; Ahanger and Agarwal 2017; Gengmao et al. 2015). Mallahi et al. (2018) reported that salinity levels ≥ 8 dS m⁻¹ had a significant adverse impact on plant biomass production, the number of plant leaves, and leaf area development.

Many options have been identified to manage soil sodicity including gypsum application, conservation agriculture practices such as zero tillage and reduced tillage with crop residue retention, acidifying fertilisers to help dissolve naturally occurring calcium carbonate, increasing soil organic matter, and growing salt tolerant crops (Page et al. 2018).

Nutrient imbalance

The continuous removal of nutrients from the soil by crops without replenishment, also known as "nutrient mining," is one of the most common types of land degradation in SA. Balanced application of nutrients is crucial for sustainable crop production, but excessive and imbalanced nutrient application may pose risks to soil health, human health, and the environment. Imbalanced nutrient application is the most common problem in SA. Subsidy-induced distortions of the type of fertiliser applied, as well as a lack of knowledge on soil nutrient requirements, are two of the key causes of this issue, which harms farmers, taxpayers, and the local and global climate (Kishore, Alvi, and Krupnik 2021). Micronutrient deficiencies are also widespread in India, according to the ICAR's All India Coordinated Research Project on Micronutrients (AICRP-M). Zinc (Zn) deficiency affects 36.5% of cultivated land, 24.2% of cultivated land is deficient in boron (B), and 12.8% of cultivated land is deficient in iron (Fe) (Kishore, Alvi, and Krupnik 2021). Therefore, an analysis of a nutrient balance sheet in crop production systems provides valuable insight into whether a soil's nutrient status is being maintained, built up, or depleted (Tandon 2007). Figure 16.3 shows a simplified representation of a nutrient balance sheet.



Figure 16.3 A simplified presentation of a nutrient balance sheet for agricultural soils (Source: Modified from Smaling 1993).

Each crop has a unique pattern of nutrient uptake per unit of produced yield. For example, rice requires 14.6 kg N, 2.7 kg P, and 15.9 kg K to produce 1,000 kg grain (Buresh, Rampolino, and Witt 2010), while wheat requires 22.8 kg N, 4.4 kg P, and 19.0 kg K to produce 1,000 kg grain (Chuan et al. 2013). Crops extract nutrients from the soil based on yield, and this is the most significant source of soil nutrient depletion. Apart from crop removal, there are other ways for nutrients to be lost from the soil, e.g., volatilisation, leaching, and soil erosion. Under certain crop-growing conditions, these losses could significantly contribute to the negative side of the nutrient balance equation (Sanyal, Majumdar, and Singh 2014).

The nutrient balance sheets in most Indian soils are negative (Figure 16.4). The estimated average net depletion per hectare from India's 142 million ha of net sown area is 16 kg ha⁻¹ of N, 11 kg ha⁻¹ of P, and 42 kg ha⁻¹ of K (Tandon 2007). In the IGP region, Zn deficiency has become common in rice-wheat cropping systems and it is now the third-highest yield-limiting factor after P and N (Nayyar 2003).

Many soil and crop management practices have been developed for nutrient management, such as site-specific nutrient management (SSNM), integrated nutrient management (INM), conservation agriculture, balanced application of plant nutrients, organic manures, green/brown manures, and the inclusion of nitrogen fixing legumes (Agegnehu and Amede 2017).

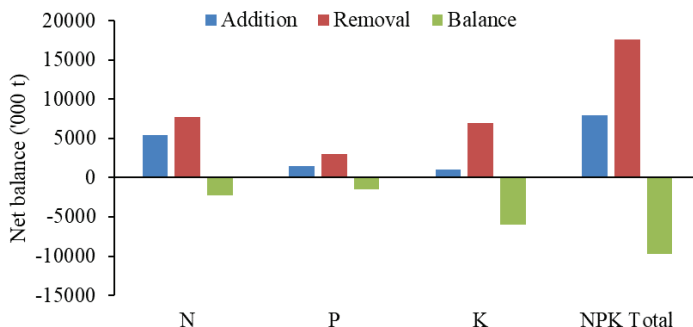


Figure 16.4 Soil nutrient balance sheet in India (Modified and adapted from Tandon 2004)

Soil physical constraints in SA

Soil Compaction

The mechanisation of agricultural operations in SA countries has increased tremendously over the last three decades. Frequent use of heavy machinery results in soil compaction and associated increases in bulk density (BD), and reductions in porosity and hydraulic conductivity. In particular, many studies have reported significant compaction in soils of the rice-wheat cropping areas of India, with increases in BD to as high as 1.80 g cm⁻³ observed in the subsoil due to the use of heavy machinery along with puddling (Sidhu et al. 2014). Compacted agricultural soils not only decrease plant growth and yield but generate more runoff during the rainy season, which causes erosion, flooding, and delivers contaminants, such as heavy

metals and persistent organic pollutants, into receiving water bodies.

In paddy fields, puddling helps to suppress weed growth and minimise water percolation, but it also decreases the growth of the subsequent wheat crop by increasing soil BD, decreasing porosity, and restricting root growth due to the formation of hardpans (Farooq and Nawaz 2014). Arora et al. (2006) found a decreasing trend in wheat yields ($0.2\text{--}0.3\text{ Mg ha}^{-1}$) with increasing puddling intensity in a three-year study. Crop yields, root length density, and root volume have also been observed to decrease as a result of soil compaction due to intensive puddling (Tomar et al. 2006). The sub-surface compaction restricts root growth of the subsequent wheat crop and limits water and nutrient uptake (Kukal and Aggarwal 2003). Ishaq et al. (2001) reported a reduction in N uptake of 12–35% in wheat due to subsoil compaction.

To minimise soil compaction the first step is to avoid cultivation when soil moisture is at, or near, field capacity. This is often done by farmers in the field by observing the stability of a soil ribbon or ball (Hanna, Al-Kaisi, and Tidman 2002). Additionally, controlled traffic lanes can be established so machinery isn't driven randomly through the field, and compaction is restricted to designated areas. Modification of tire size and pressure can also assist in reducing compaction. Larger tires allow better floatation and lower tire pressures reduce the load on the soil. Practices such as conservation tillage can also help minimise compaction as this reduces the number of passes made through the field and does not disrupt the soil structure as much. The addition/retention of crop residues also improves soil structure as well as soil organic carbon. Similarly, the application of manure in autumn on drier soils can help reduce compaction by stimulating soil microbiological activity, which promotes improved aggregation.

Soil Erosion

Soil erosion in the SA occurs due to both the natural characteristics of the region (high rainfall intensity, seasonal dry spells, naturally erodible and unstable soils) and poor management (particularly, cultivation on steep slopes and failure to maintain adequate levels of soil cover) (Lal 2017; Panagos and Katsoyiannis 2019). The total area affected by water and wind erosion in SA is presented in Table 16.2. The most severely affected countries are India and Pakistan. Significant areas in the Pothwar Plateau of the Punjab region of Pakistan (Nizami and Shafiq 1990) and hill areas of Sri Lanka (Stocking 1992) have been abandoned due to severe erosion. In parts of India, Nepal, and Sri Lanka, estimates of soil loss exceeding $100\text{ t ha}^{-1}\text{ y}^{-1}$ have been recorded (Das et al. 1977; Stocking 1992).

The impact of soil erosion is more severe in hilly areas and in semi-arid regions where both wind and water act as major agents of erosion (Iqbal and Khan 2014). The accelerated soil erosion is a serious threat to achieving food security in SA. Soil erosion removes not only the macronutrients and topsoil, but also soil micronutrients such as Zn, Fe, selenium (Se), and B. This can lead to nutrient deficiencies in food and thus malnutrition, especially in children (Lal 2009). Erosion also causes the siltation of reservoirs, which decreases the capacity of dams to store water and increases the risk of floods during the rainy seasons (Sabir et al. 2013; Ghosh and Mistri 2015). These floods affect cultivated crops and the quality of grain. Soil erosion decreases N, P, and organic carbon, reduces clay content of the soil, depletes available water capacity, and reduces soil aggregation, all of which adversely affect crop productivity (Lal et al. 2000).

Many practices are recommended for the control of soil erosion, including agronomic practices (conservation agriculture, intercropping, cover crops, and mulching), agrostological practices (grass cultivation), and mechanical practices (contour farming, contour terraces, contour bunds, micro basin tillage, and geo-textiles) (Figure 16.5) (Ahmad, Mustafa, and Gideon 2020).

Table 16.2 Total area affected by soil erosion in South Asia (M ha).

Country	Soil erosion (M ha)		Reference
	Water	Wind	
Afghanistan	11.2	2.1	Lal (2004); Srinivasarao et al. (2019)
Bangladesh	1.7	0	Karim and Iqbal (2000)
Bhutan	0.04	0	Lal (2004); Srinivasarao et al. (2019)
India	94	9.0	Bhattacharyya et al. (2015)
Nepal	1.6	0	Lal (2004); Srinivasarao et al. (2019)
Pakistan	13.8	4.4	Khan, Ahmad, and Shah Hashmi (2013)
Sri Lanka	1.1	0	Lal (2004); Srinivasarao et al. (2019)

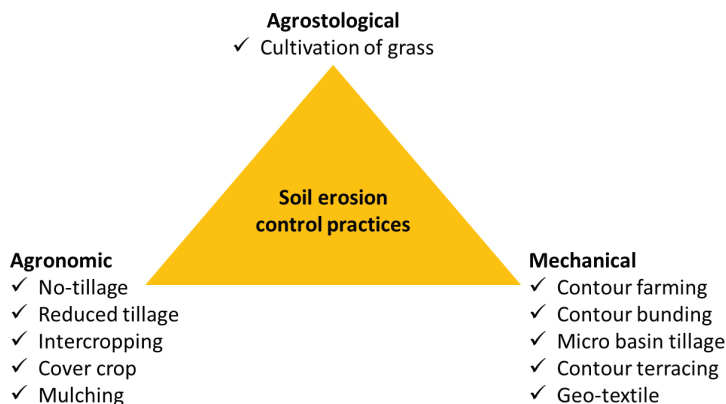


Figure 16.5 Recommended soil erosion control practices in South Asia.

Soil crusting

Soil crusting or surface sealing is a common and critical problem in SA, particularly in India (Indoria et al. 2017), Pakistan (Khan, Ahmad, and Shah Hashmi 2013) and Afghanistan (ICARDA 2002). Generally, soil crusts are formed in soils with high amounts of fine silt and sodium and low amounts of soil organic matter (Hillel 1998). In India, this is a common problem in rainfed regions of Andhra Pradesh, Haryana, Rajasthan, Uttar Pradesh, Bihar, and West Bengal (Indoria et al. 2017). However, this problem is also reported in arid regions of India with average annual rainfall of 140-180 mm (Srinivasarao et al. 2016).

Poor emergence of seedlings caused by soil crusts has a negative impact on crop stand/establishment. Even when other factors such as moisture, oxygen, soil temperature, and planting depth are not restrictive, soil crusts hamper the emergence of young seedlings (Awadhwai and Thierstein 1985). The soil crust decreases infiltration, increases runoff, inhibits seedling emergence, and reduces crop yield. Soil crusts have been blamed for limiting the availability of oxygen to germinating seeds due to their low porosity and the presence of strongly oriented soil particles. The red sandy loam soils, also known as *Chalkas*, that cover a huge area of Andhra Pradesh (India), become extremely hard after drying, affecting crop growth and development (Indoria et al. 2017). Nabi et al. (2001) also reported the hampered growth of cotton in Pakistan due to high crust strength.

To reduce the impact of soil crusts on crop production, a number of practices are recommended. These can include strategic tillage/the use of

mechanical crust breakers to disturb soil crusts, followed by the use practices that can help stabilise soil structure. Stabilising practices can include conservation agriculture, surface mulching, and the addition of soil amendments/conditioners or organic manures. Agronomic practices, such as sowing at appropriate depths and planting on ridges, can also help minimise the impact of crusting (Nizami and Akhtar 1990).

Waterlogging

A waterlogged soil is one that has become saturated with water and is unable to retain oxygen in its pore space (Singh 2015). Waterlogging is a major issue in the eastern IGP, Indus Basin of Pakistan (Chaudhary 2010), and southwest coastal region of Bangladesh (Awal 2014). In India, waterlogging is a major problem in Kerala, Karnataka, Andhra Pradesh, West Bengal, and Tamil Nadu (Bhattacharyya et al. 2015).

Waterlogging can be caused by several factors, including heavy rainfall, runoff accumulation, inadequate drainage, clayey soil, mismanagement of irrigation water, shallow water table depth, seepage from upstream reservoirs, flat to concave topography, and the discharge of canal water into channels with insufficient drainage capacity (Bilal, Ali, and Kim 2014; Pandey, Singh and Singh 2015).

Waterlogging has a direct impact on plant growth by reducing physiological activities and an indirect impact by reducing nutrient availability due to poor soil aeration. Plants become shallow rooted (Barrett-Lennard 1986), resulting in salinisation as salts rise due to capillary action (Hillel 1998). Waterlogging and salinisation have the greatest impact in heavy soils, which have a longer effective period of waterlogging than light soils (Ponnamperuma 1984). Excessive water content causes hypoxic or anoxic conditions, which inhibits the function of nitrifying microorganisms, resulting in a reduction in nitrogen availability and crop yield (Jaiswal and Srivastav 2018).

Waterlogging problems can be effectively controlled by engineering and crop management practices. These can include installing shallow interceptor drains (Hatton et al. 2002), installing a drainage pump to discharge congested water (Chandio et al. 2012), rainwater harvesting to reduce the accumulation of congested water (Li 2012), reducing ground water recharge to lower the water table (Singh 2011), subsurface drainage (Ritzema et al. 2008), and biodrainage/mole-drainage (Angrish, Toky, and Datta 2006). *Eucalyptus spp.*, *Prosopis juliflora*, *Tamarix aphylla*, and other tree species have been proven to be the greatest bio-drainers of waterlogged soils (Toky et al. 2011). In black soils, various tillage practices to produce ridges and furrows, as well as sunken and raised beds /broad beds and furrows, are

effective at preventing waterlogging during the rainy season (Painuli and Yadav 1998).

Conclusions

To feed the world's growing population, improve soil health and ensure long-term sustainability, the effective management of soil constraints is required. South Asia has a wide range of climatic zones, which are experiencing change due to a variety of climate change effects. This region is known for hot spots of population growth and has food baskets that supply both the region and other countries. Human pressures, in combination with changing hydrology and land resources, have a significant effect on food grain production and ecosystem resilience. Improving soil carbon/health through management practices such as conservation agriculture, site-specific nutrient application/-soil management zones, balanced/integrated nutrient management, soil amendments (e.g., lime for acidic soils, gypsum/organic matter for sodic soils), precision farming, and organic farming in niche areas, are some of the possible strategies to overcome soil constraints and increase crop productivity in the region. In addition, appropriate mitigation strategies for existing degraded land in the SA are essential. Therefore, increasing our expertise in these areas represents a significant opportunity to increase farmers' ability to profitably manage soil constraints across SA.

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CHAPTER 17

THE MIDDLE EAST

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Abstract

Food security and sustainability are major challenges faced by Middle Eastern countries, and particularly those reliant on food imports. The major constraints that prevent countries achieving self-sufficiency in food supply are often soil related issues such as soil salinity, desertification, low organic matter, erosion, mountainous soil profiles etc. Along with these soil constraints, high temperatures and lack of water supply are also major factors that affect food production. However, with the aid of appropriate management strategies, increases in food production can be achieved. For example, growing salt tolerant crops, managing fertilisation with integrated approaches and seed treatments, treating wastewater to allow its use in agriculture, focusing on agroforestry, and diversifying the cropping system are all strategies suitable for Middle Eastern regions. In conclusion, food production and security can be ensured in the Middle East by using appropriate management strategies.

Keywords: Salinity, organic matter, desertification, agroforestry, diversified cropping system.

Introduction

The Middle East is a large and diversified geographical area located in southwest Asia and northeast Africa. It is made up of 16 countries with a population of more than 440 million people, and in 2019 had an average annual population growth rate of ~1.9% (World Bank 2021). The region spreads over 2,000 miles from black sea in the north to the Arabian Sea in south and 1,000 miles from the Mediterranean Sea in the west to the mountains of Iran in the east. The largest country in land area is Saudi Arabia, while Bahrain is the smallest.

The Middle East has an arid to semi-arid climate; however, climate varies from region to region. Summers are hot and humid in coastal areas and hot and dry in the interior (FAO 2008). The region can be classified into four major zones, namely, mountainous areas, large gravel fans bounding mountains, alluvial plains beyond the fans, and a central base level plain between the mountains or adjacent to the coast (Fookes 1978). However, within these zones, there are wide variations in soil types, landscapes, and land use practices (De Pauw 2002). The two most common soil textures are sandy and loamy sand. These soils typically have low organic matter content, are calcareous, and have a high soil pH (Poppiel et al. 2021). The nature of the Middle East as a dryland system makes its soil highly vulnerable to any climatic changes or inappropriate human intervention. Therefore, there is a need for sustainable management practices to preserve these fragile soils.

The agricultural sector plays a very important role in food security, although its contribution to gross domestic product (GDP) varies between countries. For example, the agricultural sector contributes 12% to GDP in Iran, while it is 0.3% of the GDP in Bahrain (World Bank 2021). Important annual crops include wheat, barley, sorghum, and vegetables, while perennial crops include dates, bananas, fodder, and coconuts (FAO 2008). Agricultural development in most countries is facing two major challenges. The first is related to the soil (e.g., soil salinity, desertification, low organic matter, erosion, mountainous soil profiles etc.), while the second is related to the harsh climate and scarcity of water resources (Clawson et al. 1971; Kaniewski et al. 2012). It is of paramount importance to understand and manage these challenges to achieve sustainable agricultural development and food security. This chapter discusses some of the major constraints to the agricultural sectors in the Middle East and potential management strategies.

Soil constraints

Soil salinity

Soil salinity is a serious land degradation issue that threatens agriculture and global food security. In the Middle East, about 38,002,500 ha are severely affected by soil salinity (Shahid et al. 2018). The main causes of soil salinity are harsh climatic conditions (e.g., low rainfall, high temperature, and high evaporation rates), intensive agricultural activities with poor management of land and irrigation, increases in salt scalding (Sabkhas), and seawater intrusion due over pumping of groundwater in coastal areas (Al-Rawahy et al. 2010; Abdalla et al. 2010; Shahid et al. 2018). Figure 17.1 shows the percentage of freshwater withdrawal by agricultural, industrial, and municipal sectors in the Middle East. From this it can be seen that the agricultural sector is the main consumer of water in all Middle Eastern countries, which exerts extra pressure on the limited water resources. Excessive pumping of groundwater in coastal areas can cause seawater intrusion, which can later salinise groundwater and the soil. Additionally, over irrigation with low quality water can also lead to soil salinity problem (Darwish et al. 2005). Therefore, there is a need to strategically manage the limited water resources in the Middle East to achieve sustainable agriculture.

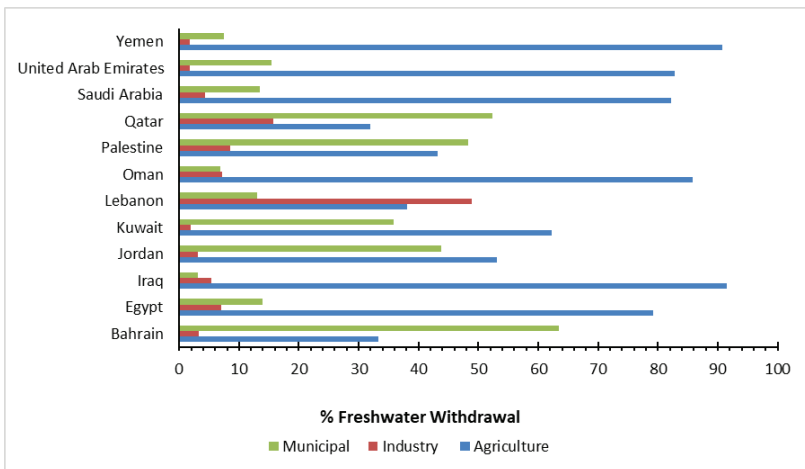


Figure 17.1 Percentage of freshwater withdrawal by sector in the Middle East (Source: FAO AQUASTAT 2017a).

Soil pH

Soils in the Middle East typically have a strong alkaline pH and low organic matter content, which is a major hindrance to crop production (FAO 2008). This high pH typically occurs due to low rainfall and high evapotranspiration rates, which lead to an accumulation of salts, such as sodium-bicarbonate and carbonates, that buffer soil pH in the alkaline range (Al-Busaidi and Cookson 2003). High soil pH reduces the mobility, solubility, and availability of nutrients to plants, which is a major cause of lower production in these soils (Neina 2019). High soil pH also increases the solubility of organic matter and the leaching of dissolved organic carbon and nitrogen (N), which reduces soil fertility and productivity (Evans et al. 2012). Additionally, soil pH affects the microbial community (Blagodatskaya and Anderson 1998) and microbial metabolic quotient (Neina et al. 2016; Neina et al. 2017). The microbial metabolic quotient was found to be two times higher in soils having a more neutral pH compared to alkaline soils (Blagodatskaya and Anderson 1998).

Low organic matter

Soil organic matter is crucial to soil physical, chemical, and biological health and long-term crop productivity. The soil's ability to store and retain moisture for crop use strongly depends upon the proportion of sand, silt, and clay. However, soil organic matter also plays a crucial role in soil water holding capability by improving the soil physical condition. Soil organic matter reduces soil crusting, reduces soil loss via erosion, and improves the soil's ability to absorb and store water (FAO 2005). Soils hold different amounts of organic matter depending upon soil geology, climatic conditions, and management (FAO 2017c). In hot and dry areas of the Middle East, plant biomass production and organic matter input into the soil is low and a major reason for the lower rates of carbon storage in this region. Soils rarely contain >1% soil organic carbon (SOC) and in most cases contain less than 0.5% SOC (Darwish et al. 2018). Despite their low carbon content, soils can be used for crop cultivation if water is available for irrigation. However, due to the high temperatures, degradation of organic matter is very high, and declines in SOC over time in cropping soils is common. This leads to declines in soil physical structure, fertility, and crop production (Alaboudi 2020). Thus, proper management practices must be adopted to increase organic matter and ensure continued crop productivity over the long term.

Erosion

Soil loss from the land surface by erosion is a widespread problem that substantially reduces the productivity of cropping systems (Pimentel and Burgess 2013). In the Middle East, more than half of the region receives <150 mm of rainfall annually, resulting in large areas with low levels of plant cover (Abahussain et al. 2002), particularly when intensive land use pressures from animals and humans disturb the topsoil (El-Kholei et al. 2012). Thus, wind erosion is a common problem and affects 60% of the land area (FAO 2015). Wind erosion reduces soil quality and fertility by removing the fertile topsoil. Additionally, accretion of eroded materials in agriculture fields, canals, and water harvesting areas severely affect crop production (Azimzadeh et al. 2008). Water erosion is also common in parts of the region where rain-fed agriculture is practiced on sloping land. The extent of water erosion depends upon the duration and intensity of rainstorms, land cover, and land use (FAO 2015).

Soil contamination

Soil contamination refers to the build-up of toxic compounds, salts, chemicals, and radioactive materials in soils, which adversely affect plant growth and yield (Ashraf et al. 2014). The rapid increase in industries, mining, wars, and intensive agriculture is a major cause of soil contamination across the globe (Bundschuh et al. 2012; Luo et al. 2017). In the Middle East, soil contamination is more prevalent in countries with high populations and oil production. In particular, increases in population, coupled with increases in liquid and solid waste disposal into soils and water resources, is responsible for soil contamination.

The overuse of chemical fertilisers and pesticides is also an important source of soil and water contamination. In Egypt, the construction of dams has decreased soil fertility due to a reduction in flooding and the deposition of alluvial sediments. This forces farmers to use higher amounts of N and phosphorus (P) fertilisers, with negative impacts on soil and water quality (NAP 2002). Additionally, industrial effluent also directly affects water quality, which in turn contaminates the soils and reduces crop productivity. Most of the industrial effluent in this region comes from cement and fertiliser plants, and iron, steel coke, and chemical manufacturing (FAO 2015). These pollutants accumulate in water ways and their field application causes chemical and biological impacts that adversely affect crop productivity and quality (FAO 2015).

Mountainous soil profiles

Although mountainous soils occur in almost all Middle Eastern countries, there is a limited literature about them. Mountainous soils are very vulnerable to climate change, desertification, soil salinity, deforestation, and human activities (Romeo et al. 2015; Moraetis et al. 2020). A common characteristic of these soils is that they are shallow, stony, and their development is very slow due to the harsh nature of the climate (i.e., high temperature and limited water), which limits biological activities (Spaargaren 2004; Egli and Poulénard 2016). They also have low fertility, often appreciable accumulations of salts, gypsum, or carbonates (De Pauw 2002), and steep topography can make them susceptible to water erosion, especially during rainy times. All of these make agricultural activities in mountainous soils challenging and greater understanding is vital for better sustainability.

Desertification

Desertification occurs under all climatic conditions and mainly results from human activities. However, dryland areas (arid, semi-arid and dry-sub-humid climatic zones), are more susceptible to desertification (Figure 17.2) due to their vulnerability to any climatic fluctuations, over-exploitation, or poor land use practices (Baartman et al. 2007; Imeson 2012).

The Middle East is one of the regions threatened by desertification as it is a dryland region and is experiencing high population growth and intensification of its socio-economic activities. The most important causes of desertification in this region are aridity of the environment, soil erosion, sand dune migration, soil salinity, deforestation, poor water management, and overgrazing (Abahussain et al. 2002; Amiraslani and Dragovich 2011; Heshmati and Squires 2013).

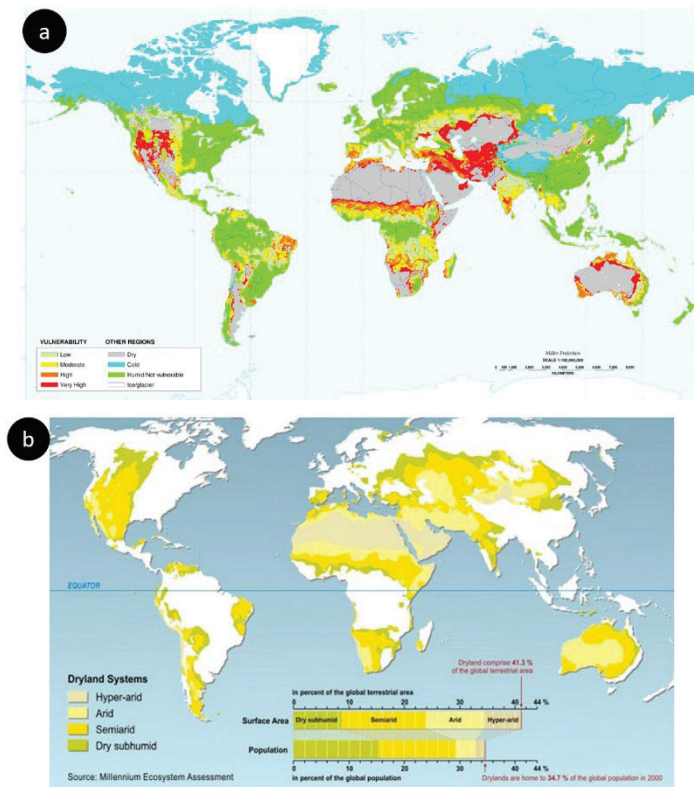


Figure 17.2 Word map showing a) desertification vulnerability (USDA-NRCS 2003) and b) distribution of drylands (Millennium Ecosystem Assessment 2005).

Management strategies

A variety of management practices are available to combat the soil constraints affecting Middle East soils. Extensive research and global efforts have been made to combat soil salinity and to reclaim salt affected lands. However, there is no single solution applicable to all sites and types of soil salinity due to the spatio-temporal variation of salt affected soils across different environmental and socio-economic settings. Therefore, reclamation techniques can be more successful if they follow an integrated

management approach (Horneck et al. 2007; Zaman et al. 2018). This involves employing a combination of several physicochemical, hydrological, and biological methods (Horneck et al. 2007; Machado and Serralheiro 2017). Several efforts have been made to combat desertification, although similarly, there is no single solution due to the complexity of the problem. Several projects across the region have been implemented to minimise soil erosion, soil salinity, stabilise dune movement, reduce overgrazing and deforestation, as well as conserve and augment water resources (Amiraslani and Dragovich 2011; Heshmati and Squires 2013).

Salt tolerant crops

The cultivation of salt tolerant crops can help to maintain production on salinised soils. Many glycophyte crops are available, which can withstand moderate salinity stress. However, only a few crops can be considered as halophytes. Among herbaceous crops, rapeseed (*Brassica napus* L.), guar (*Cyamopsis tetragonoloba* (L.) Taub.), wheat (*Triticum aestivum* L.), kenaf (*Hibiscus cannabinus* L.), barley (*Hordeum vulgare* L.), sorghum (*Sorghum bicolor* (L.) Moench), pearl millet (*Cenchrus americanus* (L.) Morrone), and cotton (*Gossypium arboreum* L.) can be moderately salt tolerant and grown in this region. Similarly, in case of vegetables, purslane (*Portulaca oleracea* L.) and artichoke (*Cynara scolymus* L.), and fruits like guava (*Psidium guajava* L.), guayule (*Parthenium argentatum* A.), pomegranate (*Punica granatum* L.), olive (*Olea europaea* L.), and grape (*Vitis vinifera*) can be considered as moderately salt-tolerant and grown in salinity stressed areas when proper management practices are adopted (Cuevas et al. 2019; Zuazo et al. 2004).

Variations in the adaptability of crops to salinity can be expected depending on soil characteristics and the agronomic practices adopted (Wagner et al. 2016; Apostolakis et al. 2017). Therefore, selection of suitable crops and the application of appropriate agronomic practices (heavy water application, nutrient management, seed priming, plant growth regulators) can both help to reduce the yield losses under salinity stress (Cabot et al. 2014; Koubouris et al. 2015). Salinity tolerance in plants is a complex trait at a physiological and genetic level that is controlled by polygenes (Linh et al. 2012). Traditional breeding approaches are a promising and efficient approach to improve salinity tolerance. Traditional breeding involves quantitative trait loci (QTL) identification using closely linked markers. One of the best QTL, saltol, was identified in rice crops using conventional breeding (Thomson et al. 2010). This QTL was effectively controlling the sodium/potassium ratio at the seedling stage in

rice. Therefore, identification of new QTLs could help to develop promising salt tolerant lines. The crop varieties developed in the Middle East have better salt tolerance owing to better salt exclusion and salt tolerant genes/QTLs compared to the varieties developed elsewhere.

Fertilisation

Appropriate management of plant nutrients and organic matter is an important approach that can ensure better crop performance in adverse conditions (Nasim et al. 2018; Roy et al. 2018). The application of organic amendments significantly improves the physio-chemical properties of salt affected soils and is considered as a highly sustainable practice (Leogrande and Vitti 2019). The application of organic amendments instead of, or in combination with, chemical fertilisers and amendments has also gained acceptance among farmers across the globe (Roy and Kashem 2014). For example, the application of gypsum (100%), or gypsum + farmyard manure to salt affected soils growing sorghum, rice, and wheat crops improved soil properties, crop yield, and biomass production (Mehdi et al. 2013). Similarly, the application of humic acid to salt affected soils improved nutrient availability [P, potassium (K), calcium (Ca), iron, zinc (Zn), and copper (Cu)], buffered soil pH, enhanced microbial activities, and improved plant growth and yield (Khattak et al. 2013). Likewise, the combined application of humic acid and gypsum substantially reduced the soil EC and SAR owing to a chelating effect, compared to the combination of either amendment alone (Shaaban et al. 2013). Sugarcane press mud, green manure, poultry manure, and sesbania growth can also be used in salt affected soils to reduce the impacts of salinity stress (Ismail 2009).

Seed treatments

Seed treatments, including seed coating and seed priming, are important methods to improve crop plant growth and germination (Hussain et al. 2018). Seed priming refers to the controlled hydration of seeds for a specific period to trigger metabolic activity before radical protrusion (Hussain et al. 2015). Seed priming is an important technology that can improve germination and subsequent growth and yield under both normal and salt stress conditions (Jisha et al. 2013), although its effects are often greater under adverse conditions (Chen and Arora 2011). For example, hydro-priming refers to the soaking of seeds in water for a specific time. This technique is effective in improving seed germination under conditions of salt stress (Li et al. 2011). Primed seed has better and more synchronised seedling growth.

Manganese seed priming of rice and wheat improved the grain yield of both species (Ullah et al. 2017; 2018). Likewise, Zn seed priming of rice seed improved the grain yield and quality of produced rice (Farooq et al. 2018). Similarly, priming using polyethylene glycol (PEG) also considerably increased seed germination, growth, and yield under salt stress (Zhang et al. 2009), as did osmo-priming with sodium chloride, potassium chloride, and calcium chloride. These compounds are inexpensive and easily available, and they substantially improved the seed germination and subsequent growth and yield under salt stress (Jiang et al. 2020).

Seed coating involves coating the seed surface prior to planting to supply different materials (growth regulators, nutrients, microbial inoculants) to the plant (Rocha et al. 2019). Seed coating can significantly improve growth and yield by increasing photosynthetic pigments, soluble sugars and proteins, free amino acids, relative water contents, and the activities of antioxidants (Rehman et al. 2020). Application of Zn through seed coating improved the grain yield in direct seeded rice (Farooq et al. 2018). Seed coating of chickpea with 5 mg Zn kg⁻¹ seed enhanced chickpea emergence/germination, seedling root and shoot length, and the dry weight of chickpea (Ullah et al. 2019). In a field study, it was reported that application of Zn through seed coating at 5 mg kg⁻¹ seed improved the nodulation, grain yield (44%), profitability, and grain quality of chickpea compared to control (no Zn application) (Ullah et al. 2020).

Agroforestry

Agroforestry is a highly effective approach to deal with salinity stress and infertility in cropping systems (Zhang et al. 2004). The benefits produced by agroforestry are of economic and environmental importance. Agroforestry improves profitability by increasing farm output and protecting the soil from the damaging effects of soil erosion. Arable crops and trees can also be grown in sequence, and this system improves soil fertility relative to continuous cropping. This is particularly the case when N-fixing trees are grown in combination with short duration and fast growing *Prosopis juliflora* and Kallar grass (*Leptochloa fusca*) for 4-5 years. During the tree phase (4-5 years), this system provides fuel, fodder, and wood. The increase in soil fertility and nutrients observed during this phase can then improve production under arable farming (Singh and Dagar 2005; Dagar et al. 2016a, b).

Agri-silviculture or agri-horticulture are also important systems that can be adapted on saline soils to improve productivity. These systems involve the concurrent growth of arable crops along with trees or fruit trees (Banyal

et al. 2017). Most fruit trees are sensitive to salt stress, however, some can be grown successfully. For instance, *Embllica officinalis* can be grown with cluster bean (Banyal et al. 2017). Salt affected soils planted with trees can show a marked improvement in SOC, soil N, and microbial activity (Sharma et al. 2014). Woody plants can also be grown in combination with pasture, which can produce fodder, wood, and reclaim saline soils by improving organic matter and reducing salt contents (Wicke et al. 2013).

Water treatment plants

Recycled wastewater can augment agricultural water resources in Middle Eastern countries. In recent years, wastewater treatment and reuse in the Middle East has gained in popularity in response to the rapid population growth and industrial development in the region, which has increased the demand for water. As a result, countries like the Gulf Cooperation (GCC) have adopted advanced wastewater treatment practices, such as ultra and nanofiltration technologies. These technologies enable those countries to efficiently treat wastewater to high degree of purity, which ultimately reduces the potential pollution from effluent and widens the utilisation of the treated wastewater for industrial and agricultural uses.

Intensive research has been conducted to study the environmental and socio-economic impacts of treated wastewater reuse in the Middle East (Shomar and Dare 2015; Shahid and Ahmed 2014; Abu-Madi and Al-Sa'ed 2009; Al-Busaidi and Ahmed 2017; Qadir et al. 2010). Reuse of treated wastewater can increase soil fertility and decrease the need for synthetic fertilisers, prevent pollution through improved wastewater management, and ultimately increase plant growth and yield and improve water and food security (Dare et al. 2017; Jeuland 2015; Al-Busaidi and Ahmed 2017).

Detailed information about the quantities of wastewater produced, collected, and treated and the area irrigated with wastewater in some Middle Eastern Countries is shown in Table 17.1. Although most Middle Eastern countries collect and treat their municipal wastewater, there are few countries that utilise the treated effluent for agriculture. There are several challenges that hinder the use of treated wastewater for agriculture in the region including: (i) poor public awareness about the quality and potential benefits of treated wastewater, which leads to poor public acceptance; (ii) lack of distribution networks to supply treated effluent to the end users and the long distances between wastewater treatment plants and agricultural areas; (iii) the high cost of the collection and treatment of wastewater, and the low tariffs for freshwater, which make farmers unwilling to replace freshwater with treated wastewater; (iv) lack of adequate facilities to store

Table 17.1 Status and use of wastewater in selected countries of the Middle East (FAO AQUASTAT 2017b)

Country	Produced municipal wastewater (GL year ⁻¹)	Collected municipal wastewater (GL year ⁻¹)	Number of municipal wastewater treatment facilities	Volume of municipal treated wastewater (GL year ⁻¹)	Area irrigated with TWW (x1,000 ha)	Area irrigated with non-TWW (x1,000 ha)
Bahrain	151	148.2	11	148.2	1.25	-
Egypt	7078	6497	382	4282	35.5	-
Iraq	850	579	21	400	-	-
Jordan	180	115	32	146.7	3.7	0.665
Kuwait	292	319	6	289	9	-
Lebanon	310	103	3	56	-	-
Oman	90	73	-	9	-	-
Palestine	71	122	-	83	-	-
Qatar	274	231.5	6	228.7	0.858	-
Saudi Arabia	1546	1144	30	1600	51.92	2.85
United Arab Emirates	500	-	41	735.6	17	-
Yemen	131.5	-	20	46	-	-

TWW = treated wastewater

treated wastewater; (v) lack of cost recovery mechanisms; and (vi) lack of long term studies about the impacts of treated wastewater on the environment and public health (Abu-Madi and Al-Sa'ed 2009; Ouda 2016; Qadir et al. 2010; Jeuland 2015; Mizyed 2013; Khanpae et al. 2020). The way forward to improve the reuse of treated wastewater in the Middle East can be achieved if these challenges are tackled.

Desalination units

Several Middle Eastern Countries depend to some extent, if not fully, on desalination to supply water for drinking and other domestic uses (Figure 17.3). Gulf Cooperation Countries (GCC) are among the top users of desalination, not only in the Middle East but worldwide, despite the fact that desalination is an energy intensive and expensive process (Dawoud 2005; Ghaffour et al. 2013; Zekri and Al-Maamari 2020). This is because most of the GCC are rich with oil resources and can afford desalination. The number of desalination plants in the Middle East are expected to increase with the increasing demand for water due to population growth and economic developments.

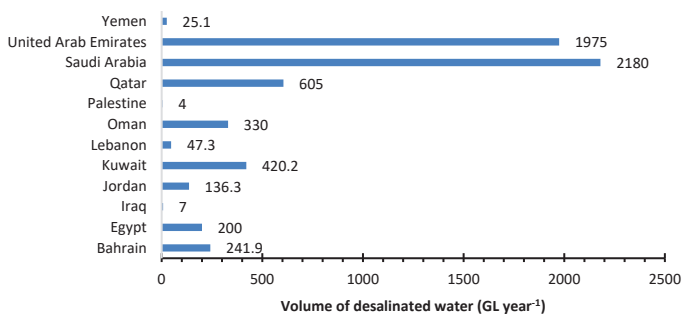


Figure 17.3 Volume of desalination water produced in some Middle Eastern Countries (Source: FAO AQUASTAT 2017c).

Along with the use of desalination technology to meet the increasing domestic demand for water, this technology can also provide a climate independent water resources for the agricultural sector. There are number of desalination technologies that can be used to produce water for agriculture including, reserve osmosis, electrodialysis reversal, nanofiltration,

and multistage distillation (Al-Jabri et al. 2019). However, reverse osmosis is considered the most energy efficient and produces larger quantities of water with less labour than other desalination systems (Burn et al. 2015; Al Jabri et al. 2019). To date, only 2% of the global desalination water production is used for agriculture (Burn et al. 2015). However, this figure is expected to increase with improvements in desalination technology and reductions in costs. The top seven countries that use desalination for agriculture in the world are Spain (22% of the total desalinated water), Kuwait (13%), Italy (1.5%), USA (1.3%), Saudi Arabia (0.5%), Bahrain (0.4%), and Qatar (0.1%) (Burn et al. 2015; Barron et al. 2015).

One of the benefits of desalination for agriculture is that it can steadily supply water of a desired quality in areas with limited water resources or saline groundwater. Since the water produced is of a pure quality [total dissolve solids (TDS) $<20 \text{ mg L}^{-1}$], it can be used for leaching salts from soil and therefore reclamation of salt affected soils. Figure 17.4a illustrates an example of the use of a small reverse osmosis unit for desalinating saline groundwater at a farm in Barka (a coastal area in northern Oman). Based on the perspective of the owner of this farm, the use of the desalination unit has provided pure water (electrical conductivity $< 0.3 \text{ dS m}^{-1}$), which helped ameliorate soil salinity (Figure 17.4b) and enabled the cultivation of crops that were not grown previously due to the high salinity of the soil (Figure 17.4c).

Despite the potential benefits of desalination, there are still many challenges that hinder its adoption in the Middle East, particularly cost. It is estimated that the energy cost accounts for about 20-70% of the total price of desalinated water (Djuma et al. 2016), although, this can be significantly reduced when solar energy is used to distilled water or generate electricity to run the desalination unit. Another challenge is the cost of the desalination unit itself and the operation and maintenance (O and M) expenses. For example, the instalment and annual O and M costs for a small size desalination unit (production capacity= 11.5 kL day^{-1} , energy requirement= 2.7 kWh m^{-3}) are about \$7,790.18 USD and \$10,90.63 USD, respectively. While costs for a large desalination unit (production capacity= $38 \text{ m}^3 \text{ day}^{-1}$, energy requirement= 3.5 kWh m^{-3}) can be double the costs for a small one (Al-Jabri et al. 2015). The high price of desalination water can be offset when farmers grow high value crops (e.g., mushroom, flowers, strawberry, etc) using efficient irrigation systems in controlled environments like greenhouses (Burn et al. 2015; Barron et al. 2015; Al-Jabri et al. 2015). Another strategy that can reduce costs is blending desalinated water with high salinity feedwater, e.g., brackish water (TDS = $500\text{-}17,000 \text{ mg L}^{-1}$). Although the blending process will increase water salinity, the FAO

standard for irrigation allows water salinity up to 700 mg L⁻¹. Therefore, this process will be beneficial as it increases the volume of water that can be used for irrigation.

Brine disposal is another major environmental problem that arises from desalination. Brine is a very saline by-product of desalination (EC=26 dS m⁻¹). Many farmers find brine difficult to get rid of and either disposed of it in pits, on the soil surface, in areas outside the farm (Figure 17.4d) or down old wells (Al-Jabri et al. 2015). Inappropriate disposal can lead to salinisation of soil and groundwater. A possible solution is to use brine for salt production.

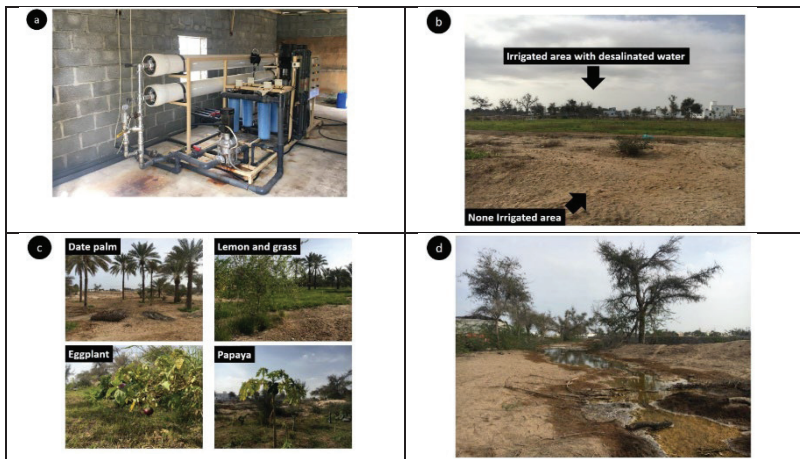


Figure 17.4 Example of a farm that uses a desalination unit for irrigation in Oman. a) Reverse Osmosis (RO) plant; b) reclamation of soil and improved growth of plants in an area irrigated with desalinated water; c) cultivated fruits and vegetables; d) brine disposal in an area adjacent to the farm (Photos taken by Ahmed Al-Mayahi).

Diversified cropping systems

Crop diversification is an approach that can increase the economic benefits gained from marginal lands. Compared to monocropping, crop diversification improves soil fertility, controls insect pests and diseases, and brings yield and nutritional stability (Lin 2011). It also serves as an excellent substitute for the use of chemicals to control pests and maintain soil fertility (Makate et al. 2016). Moreover, crop diversified systems are considered to

be more stable and resilient owing to reduced weeds, insect pressure, reduced erosion, and increased soil fertility and per unit area yield (Lin 2011). Additionally, crop diversification provides habitat for beneficial insects, which in turn reduces pests and parasites (Makte et al. 2016).

Different salt, drought and metal tolerant crops have been developed at various research stations across the globe to promote the rehabilitation of marginal soils. These crops include quinoa, maize, safflowers, sorghum, barley, mustard, sesbania, and perennial grasses. These crops have superior salinity tolerance, reduced water requirements, and can be used as feed, food, and for different industrial purposes. These characteristics make them promising candidates for diversified cropping systems in the light of their economic value (Al-Dakheel et al. 2015; Hussain et al. 2016; Al-Dakheel and Hussain 2016).

Crop diversification can be practiced at both a field and landscape level, and it can include different innovative practices including agro-forestry, legume cereal intercropping, relay cropping, perennial forages, and integrated crop and livestock systems. Moreover, adaption of agriculture systems for marginal soils must consider climate associated factors. This is imperative to combat climate change severity and its corresponding impacts on crop productivity. Many studies have emphasised that adaptations must be site specific and management practices most appropriate for the Middle East region must be identified. These can include changes in sowing time, introduction of stress tolerant cultivars, and using non-traditional water sources to help maintain the sustainability of agro-ecosystems on marginal lands (Hussain and Al-Dakheel 2015). Moreover, the application of organic amendments, crop livestock integration, and improved water management will also lead to marked improvements in soil characteristics and agro biodiversity (Hussain et al. 2020).

Future Research Areas

Future research into the sustainability of food systems in the Middle East needs to focus on: the adaption of diversified cropping systems; growing abiotic stress resilient crops (as tolerant to salt, drought, heavy metals, and heat stress); agroforestry, with emphasis on reduced water requirements and more biomass production; the treatment and use of wastewater for agriculture; and the use of environmentally friendly fertiliser approaches, including integrated strategies (growing of legumes) and seed treatments (coating, pelleting, and priming).

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CHAPTER 18

EUROPE

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Abstract

It is estimated that 60–70% of soils in Europe are more or less degraded and may continue to degrade due to climate and land use changes. The cost of soil degradation is huge and, in some estimates, erosion, loss of organic matter, salinisation, landslides and soil contamination cost the European Union (EU) member states up to €38 billion annually. Several policy instruments exist at a European and country level that either explicitly mention soil threats and/or soil functions; or implicitly offer some form of protection for soils. The European Commission has recently developed a new EU Soil Strategy 2030. The aim is to address soil- and land-related issues in a comprehensive way to help achieve land degradation neutrality by 2030. To address some of the soil constraints, the European Union launched the Mission “A Soil Deal for Europe”, with the aim to lead a transition towards healthy soils by 2030. This chapter describes four major soil constraints for crop production in Europe: soil drainage (Northern/Central Europe); soil erosion (Mediterranean region); soil compaction (Scandinavia); and soil salinity (Crete, Greece).

Keywords: Soil threats, erosion, salinisation, drainage, compaction.

Soil constraint and policy responses

It is envisaged that Europe's soil resources may continue to degrade due to climate and land use changes, mainly caused by human activities (Gobin et al. 2004), unless the drivers of soil degradation are addressed, and appropriate measures are taken. The European Commission (EC) has identified a number of threats to soil under the topic "Towards a Thematic Strategy for Soil Protection" (EC 2006). A brief definition and discussion of the main soil threats that directly and/or indirectly affect crop production in Europe is given in Table 18.1.

The cost of soil degradation is huge and, in some estimates, erosion, loss of organic matter, salinisation, landslides, and soil contamination in total might cost the European Union (EU) member states up to €38 billion annually (EC 2006), with the majority of these costs borne by society. Montanarella and Panagos (2021) estimated that 60–70% of soils in Europe are more or less degraded. There are several studies that show the magnitude of soil degradation in Europe (Table 18.2).

Several policy instruments exist at a European and country level that either explicitly mention soil threats and/or soil functions; or implicitly offer some form of protection for soils³. Environmental policies are likely to mitigate soil threats, for e.g., through improved soil management, land rehabilitation, green infrastructure development, or the limitation of urban sprawl. These measures reduce the effects of soil threats, such as the decline in organic matter (OM), loss of soil biodiversity, and soil sealing. Water management policies generally have a positive impact on soil by reducing fertiliser use and improving manure management (Nitrates Directive), and through the reduction of pressure from agriculture, restoration of rivers and ecosystems, and the stimulation of sustainable land use, including floodplain restoration (e.g., Water Framework Directive and Floods Directive). These directives have reduced levels of a range of threats, such as flooding and landslides and decline in OM.

The Common Agricultural Policy (CAP) is a key EU land management policy and a central driver for the management of agricultural land in particular. In general, the CAP includes a range of policy instruments that can positively or negatively impact most soil threats (excluding soil sealing) and soil functions. Key priorities for the CAP 2023–2027 include supporting the sustainable growth of food production and greener farm

³ https://ec.europa.eu/environment/soil/soil_policy_en.htm

practices through eco-schemes. Under these schemes, specific payments will be provided to farmers that adopt climate sensitive and nature-sensitive friendly practices in line with the European Green Deal objectives. Examples of these actions include organic farming, crop rotation, and preservation of carbon rich soils. During 2023-27, the CAP will be built around nine key objectives (Figure 18.1), which will be the basis upon which EU countries design their CAP strategic plans.

Table 18.1 Brief definition and concepts of the main soil threats in Europe (EC 2006).

Soil threat	Definition/concepts
<i>Soil erosion by water/ wind</i>	A three-phase soil process of: (i) the detachment of individual soil particles; (ii) their subsequent transport by an erosive agent; and, ultimately, (iii) their deposition.
<i>Organic matter decline in peat/ mineral soils</i>	A negative imbalance between the build-up of soil organic matter and rates of decomposition, leading to an overall decline in soil organic matter contents and/or quality, causing a deterioration or loss of one or more soil functions.
<i>Soil compaction</i>	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.
<i>Soil sealing</i>	Permanent covering of soil with a completely/partly impermeable material (e.g., asphalt, concrete) for housing, roads, or other land developments.
<i>Soil contamination</i>	The occurrence of a contaminant (any substance with the potential to cause damage, irreversible or not, in the environment) in soil above a certain level causing deterioration or loss of one or more soil functions.
<i>Salinisation</i>	A term that includes saline, sodic, and alkaline soils where the accumulation of soluble salts severely affects soil fertility and plant growth.
<i>Desertification</i>	A result from land uses that have become unsustainable through the action of one or a combination of processes, many of them arising from, or aggravated by, human induced activities and habitation patterns, and potentially aggravated by climate change.
<i>Floods and landslides</i>	Natural hazards, which can be aggravated or triggered by human interactions with the environment, causing loss of natural resources (soils, water), damage to buildings, infrastructures, and man's activities, and potentially claiming lives.
<i>Loss of Biodiversity</i>	Reductions of forms of life living in soils (both in terms of quantity and variety) and of related functions.

Table 18.2 Examples of soil threats affecting crop production in Europe

Soil threats	Evidence	References
<i>Soil erosion by water</i>	Loss of agricultural productivity is 0.43% annually, using results of combined biophysical and agronomic models.	Panagos et al. (2018)
<i>Soil erosion by wind</i>	About 0.53 t of soil loss ha ⁻¹ yr ⁻¹ from arable land.	Borrelli et al. (2016)
<i>Soil compaction</i>	About 23% of European soils have a critically high bulk density.	Stolte et al. (2016)
<i>Soil salinity</i>	Salt-affected soils are less fertile than non-saline soils and produce less biomass.	Stolte et al. (2016)



Figure 18.1 Nine objectives of the Common Agricultural Policy⁴

However, there is no binding framework that strategically defines policy priorities/instruments for soil protection at the EU level. In other words, key policies that offer some strategic vision are non-binding (Frelüh-Larsen et al. 2016) and they cannot be used as a basis for integrating and reinforcing the protection of soil within the EU. Soil protection measures are mostly derived as a consequence of delivering environmental objectives that are not explicitly soil focused. These include reducing contamination, offsetting greenhouse gas emissions, and preventing other environmental threats.

To address important soil constraints in Europe, sustainable soil

⁴ https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy/new-cap-2023-27/key-policy-objectives-new-cap_en

management practices are required for specific pedoclimatic and socioeconomic conditions (Barão et al. 2019). This will enable soils to function properly and provide the required ecosystem services. There is a large body of literature related to good soil management practices for crop production across Europe (e.g., Orsini et al. 2020). However, these practices are not reinforced by appropriate policy measures and not sufficiently implemented at the local level to enhance sustainable crop production in soils exposed to various forms of degradation.

To meet some of the soil constraints, the European Union has launched the Mission “A Soil Deal for Europe”⁵. The main goal of the mission is to lead a transition towards healthy soils by 2030 by establishing 100 living labs and lighthouses. The Mission will support the EU’s ambition to lead on global commitments, notably the sustainable development goals, and contribute to the European Green Deal⁶ targets on sustainable farming, climate resilience, biodiversity, and zero-pollution.

In the following sections, four major soil constraints, i.e., soil drainage (a case representing the Boreal region of Northern/Central Europe), soil erosion (representing the Mediterranean region of Europe), soil compaction (encompassing Scandinavia), and soil salinity (case of Crete, Greece), are described and discussed.

Soil drainage: Case of Central/Northern Europe (boreal, nemoral, climatic zones)

State of soil drainage

Climate change is expected to increase precipitation amount and intensity in northern Europe, particularly during extreme events. This increases the need for drainage measures (Jensen et al. 2021) to prolong the growing season (Wiréhn 2018).

i) Norway: Approximately 60% of the agricultural area in Norway is artificially drained, with the proportion in cropped land areas even higher. Much of the cropped land area is located on marine clay in Eastern and Mid Norway. Here, the soil is completely dependent on systematic drainage to provide good yield. In a survey among farmers in Norway, Hauge et al. (2020) found that there was a difference of around 300 kg ha⁻¹ cereal yield between a badly drained and optimally drained field, and up to 850 kg ha⁻¹ for newly drained fields compared with the “before-drain” period.

⁵ <https://ec.europa.eu/eip/agriculture/en/news/launch-eu-mission-soil-deal-europe>

⁶ https://ec.europa.eu/info/strategy/priorities-2019-2024/european-green-deal_en

ii) Sweden: In Sweden, 28% of arable land needs new tile drainage or refurbishment of the existing tile drainage system. However, only 6% of the arable land has plans for refurbishment or a new system installation within a five-year period (Wiréhn 2018). A Swedish study showed that improved drainage decreases the drought sensitivity of crop production (Jordbruksverket 2013). However, unless current drainage systems are excessively poorly dimensioned, it is not economically viable to update the main system before its technical lifetime has expired.

iii) Denmark: Rasmussen et al. (2018) predicted an expected 10% increase in drain flow by 2030–2059 and a 16% increase by 2060–2099 in Denmark. Møller et al. (2018) estimated that artificially drained areas were 49.4% of Denmark's agricultural area, based on a model ensemble and field observations. The ensemble mostly predicted artificially drained areas on the loamy tills of Eastern Denmark and in the large wetland areas of Northern Denmark.

iv) Germany: Only estimations exist concerning the extent of artificially drained areas in Germany (Tetzlaff et al. 2009). Artificial drainage systems can be found on 10–20% of agricultural land, while the need for artificial drainage is estimated to be around 30–35% of agricultural land. In North-Western Germany, this share is estimated to be circa 60% (Eggelsmann 1978).

v) Netherlands: Agriculture was intensified after the Second World War, leading to intensive drainage that lowered groundwater levels, increased drainage rates, and resulted in more drought stress in dry periods. This process was further intensified by the land consolidation practices employed to reduce the problems of fragmentation of land holdings (Van den Noort 1987).

Adaptation and best management practises

Current drainage guidelines are mainly based on research from the 1960s and 1970s for Sweden and Norway (Wiréhn 2018; Hove 1981). Drainage of agricultural fields in Denmark was primarily done between the 1860s–1890s and the 1930s–1970s (Breuning Madsen 2010). Wind (1976) studied the influence of drainage on soil workability in the spring in the Netherlands. The number of workable days was found to be hardly influenced by the drain spacing. Drain depth, however, greatly affected workability. It was found that on average there are 6.2 workable days in March/April when the drain depth is 40 cm, 12.5 days when 100 cm, 15.4 days when 150 cm, and 17.1 days when 200 cm. Deep drainage allows earlier spring planting, resulting in better yields. However, deep drainage

also makes light textured soils more drought sensitive, and the commonly applied drain depth of approx. 100 cm appears to be about correct for a sandy loam soil under the prevailing conditions in the Netherlands (Wind 1976). There is a need to revisit these drainage guidelines given the climate change projections for Northern Europe.

Adaptation to climate change by improving water management and drainage systems is essential. At the farm level, key drainage adaptation measures are to increase the capacity of the sub-surface drainage system. In existing systems, reducing the drainpipe distance from 15 m to 12 m can increase the drainage capacity by approximately 45% (Wiréh 2018). Hove (1981) compared 4, 8, 16, and 32 m drain distances in an experiment in Norway. Increased drainage intensity resulted in an increase of 250-300 kg ha⁻¹ of cereal yield at half the drain distance (4 m: about 3500 kg ha⁻¹, 8 m: about 3300 kg ha⁻¹, 32 m: about 2800 kg ha⁻¹). The crop yield increase was closely related to the possibility of earlier spring sowing. Mean sow time was delayed by an average of 5-6 days by doubling the trench distance. Deelstra (2015) reported that the drainage system in Norway was able to cope with high precipitation amounts like those predicted by climate change scenario studies. This suggests that no adaptation of the design is necessary, but rather there is a need to increase the number of drained fields and upgrade existing systems established in the 1960s and 1970s (Hauge et al. 2020).

Swedish experiments showed a decline in spring cereals of between 4-10% 4 m from a ditch, and between 5-14% 8 m from a ditch in trench trials with 16 and 32 m trench distances. Controlled drainage (CD) could be an alternative measure to improve soil drainage, and CD is one of the tested measures to, for example, reduce nitrogen (N) leaching from agricultural fields. CD can play an important role in saving water and nutrients and improving and optimising downstream water availability and quality.

Wesström and Messing (2007) showed that CD significantly lowered N and phosphorus (P) loads in drain outflow and altered N dynamics in soil in an experiment in Sweden. In their study, they found an increase in yield of 2-18% for CD plots in a crop-rotation experiment, where the crop N uptake increased by 3-14 kg ha⁻¹. For the Netherlands, it is suggested that drainage systems should be altered to move from a strategy of rapid removal of all excess water to one that continuously controls water levels individually in each agricultural plot (Ritzema and Stuyt 2015). In this study, CD resulted in a reduction in N losses and thus a positive effect on the quality of drainage effluent. Deichmann et al. (2019) found for an experiment in Denmark that CD did not introduce waterlogging stress when applied during winter, nor cause lower yields.

Soil erosion: Case of Southern Europe (Mediterranean climatic zone)

State of soil erosion for crop production

Mean sheet and rill erosion rates for the European Union and UK are an estimated to be $2.45 \text{ t ha}^{-1} \text{ year}^{-1}$ for rill and interrill for all land uses and ca. $2.7 \text{ t ha}^{-1} \text{ year}^{-1}$ for arable land only (Panagos et al. 2020). The highest annual soil loss rates are observed in Mediterranean areas, on land dominated by vineyards.

Based on a modelling study, Bakker et al. (2007) concluded that crop yields in Europe are unlikely to be significantly reduced by the levels of soil erosion occurring for the next century. However, there is a considerable spatial variability in the effect of erosion on crop yield. For the Mediterranean area, erosion-induced yield reductions are higher than Northern Europe. For example, Greece is likely to suffer the largest reductions, followed by Portugal, Italy, Spain, and the south of France (Bakker et al. 2007). Cerdan et al. (2010) showed that plot erosion rates have a distinct regional difference, with lower values in the Mediterranean than the rest of Europe. However, due to the very shallow soils in this area, any rate of soil erosion has a large negative effect on crop productivity.

In a more recent study, it was estimated that by 2050, the mean soil loss rates may increase by 13–22.5 % in agricultural areas of the EU and the UK compared to the 2016 baseline, mainly due to climate change (Panagos et al. 2021). Countries with currently low erosion rates, such as France, Denmark, and the Netherlands, are expected to experience increased rates, whereas areas currently experiencing high erosion (Italy, Slovenia, Portugal, Spain, and Greece), are expected to experience reduced rates in the future. For the EU, it was estimated that the loss of crop yield due to soil erosion is on average 0.4% per year, which is equal to a crop productivity loss estimate of €1.2 billion (Panagos et al. 2018).

Mulching as soil management practice

Mediterranean vineyards and orchards in France and Spain showed the highest soil erosion rates (Prosdocimi et al. 2016) under rainfed agriculture. The main reason is high erodibility of the bare soil exposed between the vines and fruit trees during the entire year. Similarly, wheat-based agricultural systems in Italy showed a soil fertility decrease over time, and crop productivity is seriously decreased as a result of soil erosion and organic matter depletion. This was attributed to the conventional soil

management system of ploughing and harrowing before seedbed preparation (Stagnari et al. 2014). Olive trees in Italy are also often cultivated in soils with high clay contents on steep hillslopes, resulting in unsustainable erosion rates (Bombino et al. 2021).

The above studies recommended mulching as an alternative soil management practice that can reduce soil erosion and increase soil fertility. Mulching reduces soil erosion by delaying runoff initiation and reducing discharge, slowing down surface runoff, improving soil infiltration capacity, and protecting the soil surface from raindrop impact (Cerdà et al. 2017). In addition to erosion reduction, mulching can lead to increased soil fertility. For example, Marinari et al. (2015) showed (under Mediterranean conditions), that mulching with cover crop species enhanced decomposition rate, tomato yield, and increased soil carbon and N storage. Stagnari et al. (2014) also showed that straw mulching led to significant improvements in soil water and nutrient availability. Keesstra et al. (2016) reported that in apricot orchards in Spain, covering soil with vegetation and chipped pruning residues increased vegetative cover and soil quality and reduced erosion rates.

However, there is a limited amount of research that directly connects mulching with improved crop yield. Implicitly, there is an expectation that crop yield will be positively influenced by mulching through the above-mentioned positive effects on soil health. However, Cerdà et al. (2017) showed that the cost of mulching was about twice the price of conventional soil management practices, which hinders adoption, despite its benefits in decreasing erosion rates by two orders of a magnitude.

Soil Compaction: Case of Scandinavia

Status

Soil compaction is a particularly important cause of soil degradation around the world (Montanarella et al. 2016). In Scandinavia, the risk of soil compaction is high, as soils are often wet during field operations due climatic conditions.

Reported crop yield losses in Sweden, Finland, and Denmark due to compaction range from 1.5–8% (Wahlström et al. 2021). Sonderegger and Pfister (2021) estimated that, at global level, the potential long-term yield losses due to soil compaction approximate 6.5% in mechanised agriculture. The area and severity of degradation attributable to compaction are not well known, and only a few studies distinguish between natural and anthropogenic causes (e.g., Schneider and Don 2019). Although the agricultural area in

most Scandinavian countries is relatively small, the impact of soil compaction is high. For example, Schjønning et al. (2016) estimated that 37% of the Danish agricultural soils are characterised by a critically compact upper subsoil.

Field operations with a high risk of subsoil compaction include conventional ploughing in spring or autumn, application of slurry during winter/early spring, and harvest of grain, potatoes, sugar beet, and silage maize in autumn (Keller and Arvidsson 2006; Thorsøe et al. 2019). According to Wiréhn (2018), the risk of compaction in Scandinavia will further increase under current climate change projections, due to decreased ground frost during winter and more extreme precipitation events, resulting in soils with a high moisture status.

The use of heavy machinery with high traction is also common (Kutzbach 2000), even on smaller farms (Seehusen et al. 2019), and contributes to compaction. For example, wheel loads reach 6 Mg in slurry applications and 12–15 Mg during sugar beet and potato harvest, while loaded combine harvesters reach wheel loads up to 12 Mg (Schjønning et al. 2015). The risk of an unfavourable combination of high loads and low soil strength is therefore high. This was also highlighted in the results of a survey of 1,126 farmers in Denmark. This showed that during potential problematic field operations (manuring in spring and the harvest of grain, root crops, grass, and maize in autumn), only 5–39% of the operations would be performed using wheel loads that roughly classified as being a low risk for subsoil compaction. On the other hand, 23–90% of the operations would be performed using wheels load that roughly classified as a high risk for subsoil compaction (Thorsøe et al. 2019). Another example of the prevailing risk of compaction comes from Arvidsson et al. (2003). They simulated the risk of soil compaction to 0.7 m depth in Southern Sweden over the period 1963–1988 and showed that the risk of subsoil compaction during field operations, such as manuring in spring and harvest in autumn, is critically high.

Sustainable soil management

Sustainable soil management from a soil compaction perspective ideally prevents soil stress exceeding soil strength, both in the topsoil and subsoil (Schjønning et al. 2015). This could, in theory, simply be done by following general guidelines. For example, Schjønning et al. (2012) proposed a rule-of-thumb, based on a range of field studies, for field traffic of wet soils: a maximum wheel load of 3.5 Mg for a tyre inflation pressure of maximum 0.5 bars. In practical terms, however, general guidelines such as these

greatly restrict current agricultural practices in Scandinavia due to their lack of practical applicability for farmers and contractors. As an alternative, the focus in Scandinavia has been on reducing the overall risk of soil compaction by minimising soil stress.

Risk assessments taking specific loading characteristics and actual soil conditions into account, such as provided by the decision support tool Terranimo® (Stettler et al. 2014), allow a quantitative comparison of stress and strength. Such assessments yield an estimate of the risk of compaction specific to the traffic-situation and may therefore be easier to implement by practitioners. Moreover, they allow investigation of the effect of technical measures such as different tyre, wheel load, and inflation pressures, and the effect of the soil moisture content. Although risk assessments are appreciated as an educational tool to (further) increase awareness of the risk of compaction (Thorsøe et al. 2019), it is unclear to which extent risk assessments are used in actual decision making. This is partly explained by the fact that farmers and contractors may lack flexibility to actively take measures to reduce the risk of soil compaction due to, among other factors, the available technology, economy, and temporality of field operations.

Technology

Examples of available technologies that reduce the risk of soil compaction for a given wheel load are wide, low-inflation pressure tyres (Ten Damme et al. 2020), rubber tracks (Alakukku et al. 2003), the option to adapt the number of wheels through a tandem of dual/twin wheels constructions (Keller and Arvidsson 2006) and central tyre inflation systems (CTISs) for on-the-go adjustments of inflation pressures. All these technologies can improve the contact stress distribution by increasing the tyre-soil contact area, which helps reduce contact and vertical soil stress. Techniques such as these are, however, limited in their own way. Wide, low-inflation pressure tyres are still considered expensive (Thorsøe et al. 2019), and while further development of tyres might further benefit the stress distribution, their contribution is expected to be limited. Tyre dimensions also affect the height and width of vehicles, and vehicle-dimensions are legally restricted for driving on public roads. For example, in Denmark, a width of 3 m is allowed for tractor-trailers only if the width beyond 2.55 m is related to wheels and wheel guards (Trafik- og Byggestyrelsen 2016).

Compared to tyres, rubber tracks have a great potential to reduce the risk of compaction due to the long contact area over which the load can be distributed. However, vertical stress tends to concentrate under the supporting rollers and the wheels of tracks (Arvidsson et al. 2011) and the

benefit of tracks compared to tyres is uncertain (Lamandé et al. 2018). A tandem configuration increases the number of wheels passing in the same wheel track (the impact of which is not fully understood), and dual/twin configurations increase the potential area of compaction (although with relatively lower stresses). While these techniques help reduce the risk of compaction for a given wheel load, they may in practice often be used to carry higher wheel loads, which actually increases the risk of subsoil compaction (Schjønning et al. 2012). The use of CTISs is not yet common practice in Scandinavia, but in, for example Germany, CTIS is promoted through subsidies.

Farm management

Farm management decisions that influence soil compaction include controlled-traffic farming (CTF) versus so-called random-traffic farming systems, reduced tillage versus “conventional” tillage versus on-land versus in-furrow ploughing, and choice of crop and sowing date. Although these decisions affect the extent or risk of soil compaction, it is unknown to what extent the risk of soil compaction drives farmers to take certain management decisions. As expressed by Jamali et al. (2021), farmers tend to manage agronomic challenges with immediate consequences, such as water, nutrients, and pests, while the effects of soil compaction may be too indistinct to consider. In CTF, soil compaction is restricted to small areas; a system many farmers have converted to, for example in Australia. The main barriers for farmers in Australia have been the need to modify machinery to fit the chosen system (Antille et al. 2019) and this may as well be a relevant constraint in Scandinavia. Edwards et al. (2016) showed for soil in the UK that the number of days of field traffic without irreversible soil structural deformation was higher under minimum tillage compared to soil under conventional tillage. In Denmark, the area under reduce-tillage has grown from approximately 10.8% in 2016 to 19.0% in 2020 (Statistics Denmark 2022). On-land ploughing can reduce the risk of compaction through lower soil stresses, in particular in the upper subsoil (Weisskopf et al. 2000). Obour et al. (2018) investigated the impact of sowing date and observed that for the early sowed treatment, for which the soil water content was highest, soil physical properties and workability (the ease at which an optimum seedbed for crop establishment is produced) were reduced compared to the timely sowed treatment.

Cropping practices

Crop harvest dates may greatly affect the risk of soil compaction since harvest campaigns usually inflict the highest wheel loads of all field

operations (Augustin et al. 2020). As precipitation generally increases and evapotranspiration generally decreases over autumn, the soil moisture content is often higher, and soil strength lower, during the harvest of potatoes and maize compared to harvest of, for example, spring barley (which is harvested approximately seven weeks earlier). This shows that the choice of crop, and taking the timing of sowing and harvest into consideration, may also reduce the risk of soil compaction.

Soil Salinity and Management Practices: The Case of Crete, Greece⁷

Status

Saline soils directly impact crop production due to high osmotic stress, low water potential of the root environment, ion-induced toxicity, nutrient imbalance in the cytosol leading to osmotic imbalance in plant cell, and oxidative stress mediated by reactive oxygen species (Negrão et al. 2017). Salinity has accumulated in the soils of most arid regions, where climate conditions enhance salinisation by lowering soil moisture and suppressing leaching events (Perri et al. 2021). This is very important in the Mediterranean regions (Iglesias et al. 2011), where 25% of the irrigated agricultural land is affected by a significant level of salinisation leading to moderate soil degradation (Mateo-Sagasta and Burke 2011). Greece is no exception, with about 9% of 1.4 million hectares of irrigated land affected by soil salinisation, mainly due to seawater intrusion increasing the salinity of irrigation water. The island of Crete is a typical example where water resources are rapidly depleted and water quality is constantly deteriorating.

Sources of salt in irrigation water supplies

Primary sources: include marine marls and proximity to the sea. Furthermore, the highly fractured nature, the dissolution, and the erosion of the limestone bedrock has created a great variety of karst features (Steiakakis et al. 2016), which often control sea water aquifer intrusion in the coastal regions and pose challenges to hydrogeological modelling (Malagò et al. 2016). A predictive simulation model by Schildgen et al. (2014) to investigate drought impact on the fresh water potential of a karst

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aquifer in Crete indicated that the transport of saline water may depend primarily on karst conduit network rather than matrix permeability (Steiakakis et al. 2016). Results from this, as well as similar physically based (Arfib and de Marsily 2004) and data-driven studies (Bekas et al. 2021), open up new perspectives for water resource management in karstic coastal zones of Crete.

Secondary source: occur mainly because of the tourism and agricultural water demand peak, which stretches the limits of surface and groundwater stores during the summer season, often leading to groundwater overexploitation (Tsanis et al. 2008). Wastewater reuse for irrigation is an attractive alternative (Petousi et al. 2015a), however, it can lead to the accumulation of macro-nutrients and trace elements and intensify soil salinisation (Petousi et al. 2019). It is also well-established that the vegetation of Crete will have to grow in a hotter, drier (Daliakopoulos et al. 2017), and in some cases, more saline environment (Chartzoulakis and Psarras 2005).

Control measures

Management, monitoring, and mapping

To control the threat of soil salinity, the Prefecture of Crete consistently practices Integrated Water Resources Management (Demetropoulou et al. 2019) and has deployed a groundwater quality monitoring network, measuring EC and groundwater levels (RoC 2009). Other observations are based on geophysical surveys (Vafidis et al. 2013). Specifically for Tympaki, the surveys of Vafidis et al. (2013) showed that groundwater salinisation occurs at depths of 30 m and that the biggest problem appears north of the stream Klematianos. Based on these measurements, several modelling applications quantified the state and rate of seawater intrusion (Pappa et al. 2017; Paritsis 2005). In terms of agricultural modelling, SALTMED (Ragab 2002) and modifications (Daliakopoulos et al. 2016a) have been used to quantify the impact of irrigation water salinity on crop yield and soil quality, both under current conditions (Daliakopoulos et al. 2016a) and a future warmer climate (Apostolakis et al. 2016).

Figure 18.2 shows areas where soil salinity is the primary or secondary limitation to agriculture, as well as areas of seawater intrusion in Greece (top) and specifically in Crete (bottom). Undoubtedly, remote sensing could help create a harmonised soil monitoring and mapping system, similar to the EEA (2000) land cover strategy (Daliakopoulos et al. 2016b).

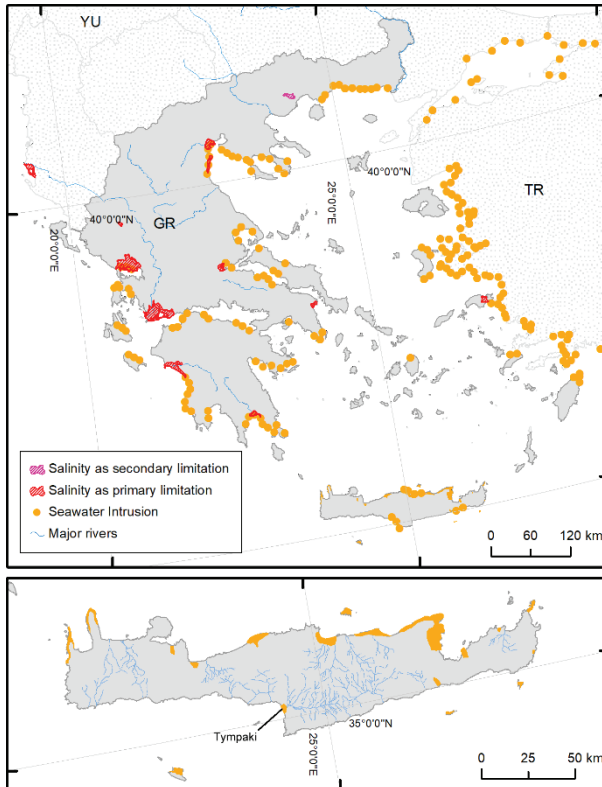


Figure 18.2 Soil salinity as a primary and secondary limitation to agriculture and areas of seawater intrusion in Greece (top) and specifically in Crete (bottom). Data from Daskalaki and Voudouris (2008), RoC (2009), Tóth et al. (2008).

Irrigation technologies

Balanced irrigation, in terms of salts introduced to and drained from the soil, is the cornerstone for preventing or halting salinisation (Cuevas et al. 2019). Micro-irrigation has several advantages when the irrigation water is saline, because continuous leaching of the soil is achieved through frequent irrigation treatments during the cultivation period, thereby preventing salt accumulation in the root environment (Cuevas et al. 2019). Driven by water scarcity and supported by increasing public water infrastructure (Tzanakakis et al. 2020), since the early 1990s farmers in Crete have invested in drip

irrigation technologies (Genius et al. 2014) and some early adopters are currently moving to smart irrigation (Daliakopoulos et al. 2020).

Fertiliser management

Another factor that usually compromises crop production is the amount of fertiliser salts applied, in addition to fertiliser scheduling. Machado and Serralheiro (2017) proposed the application of fertilisers through irrigation water, adjusting nutrient supply to plant specific requirements, and at the same time mitigating the osmotic effects caused by an overabundance of nutrient ions in the rhizosphere. Furthermore, the use of biofertilisers has frequently been reported to alleviate salinity impacts on vegetable production by utilising microorganisms that enhance plant nutrient status. One such example is the common symbiotic fungi of the *Trichoderma* genus, which colonises plant roots, improving root system development and increasing plant tolerance to abiotic stresses such as drought and salinity, facilitating nutrient and water uptake. Daliakopoulos et al. (2019) found that soil quality was significantly affected by an irrigation treatment, but that *T. harzianum* successfully reduced the effect of higher salinity irrigation, especially in the case of sodium (Na) absorption ratio, and positively affected bioavailable P concentration and could effectively reduce P fertiliser requirements.

Salt tolerant crops

Olive (*Olea europaea* L.) is a Mediterranean staple crop commonly cultivated in marginal lands and characterised as moderately salt tolerant. The salinity tolerance mechanism of *Olea europaea* L. involves salt exclusion from the shoot and the large use of osmoprotectant mannitol to regulate leaf osmotic potential. Chartzoulakis et al. (2002) investigated the effect of five salinity levels [0, 25, 50, 100, and 200 mM sodium chloride (NaCl)] on yield characteristics and ion content of six olive cultivars and concluded that shoot length and total leaf area was significantly reduced above 25 mM NaCl for cultivars Koroneiki, Kalamata, and Megaritiki and above 50 mM NaCl for Mastoidis, Amphissis, and Kothreiki and that the ion concentrations of Na⁺ and Cl⁻ (% d.w.) were higher in roots than any other part of the plant. Additionally, Petousi et al. (2019) demonstrate that after two years of irrigation of olive trees (*Olea europaea* L. cv. “Koroneiki”) with secondary and tertiary-treated water, Na concentrations of 50-413 mg kg⁻¹ did not significantly affect photosynthetic capacity (Fv/Fm) and relative chlorophyll concentration (SPAD).

Moreover, the cultivation of salinity-resistant endemic halophytes and glycolhalophytes such as *Portulaca oleraceae* (Portulacaceae), *Cichorium spinosum* (Asteraceae) and *Scolymus hispanicus* (Asteraceae) that have genetically and physiologically adapted to the local adverse soil and climatic conditions (Petropoulos et al. 2018) has recently gained interest. Chatzigianni et al. (2019) demonstrated a significant tolerance of a coastal-marine ecotype of *Cichorium spinosum* (a wild endemic species in Crete) at salinity levels of 20 and 40 mM NaCl in a soilless cultivation system.

Use of biological and organic materials

The combined use of biological and organic materials (Cuevas et al. 2019) as bioorganic amendments can improve soil structure and increase soil organic, humic, and nutrient status, which are the most growth-constraining factors in saline conditions (Bello et al. 2021). Nevertheless, few test cases have been documented in Crete so far (Panagea et al. 2016).

Finally, from a farmer's point of view, when salinity conditions are not favourable, options range from adopting different cropping systems to minimise economic impacts, to implementing control measures, to eventually abandoning the land (Cuevas et al. 2019). Farmer and stakeholder perceptions, as well as barriers and opportunities for the adoption of salinisation mitigation and adaptation practices, have been documented. Future research needs to focus on innovative ideas, including alternative water resources (Petousi et al. 2015b), improved cropping systems (Panagea et al. 2016), and social innovation (Okpara et al. 2020).

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CHAPTER 19

CLIMATE CHANGE IMPACTS ON SOIL SALINITY IN CALIFORNIA'S SAN JOAQUIN VALLEY

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Abstract

Changes in climate patterns have dramatically influenced some agricultural areas within North America. Examples include the historic 7-year drought in California's San Joaquin Valley (SJV) and the 20-year above average annual rainfall in the Red River Valley (RRV) of the Midwestern USA. Climate change has impacted soil salinity levels in the root zone of these agricultural areas. Inventorying and monitoring climate change impacts on salinity are crucial to evaluate the extent of the problem, to recognise trends, and to formulate irrigation and crop management strategies that will maintain the agricultural productivity of these areas. Over the past 3 decades, Corwin and colleagues at the U.S. Salinity Laboratory have developed proximal sensor (i.e., electrical resistivity and electromagnetic induction) and remote imagery (i.e., MODIS and Landsat 7) methodologies for assessing soil salinity at multiple scales: field (0.5 ha to 3 km²), landscape (3 to 10 km²), and regional (10 to 10⁵ km²). A case study for the SJV is presented that utilises these approaches to assess soil salinity changes due to changes in climate patterns. Discussion is given to the ramifications of the findings, which is causing a shift to high cash crops under micro-irrigation and subsequently potential future problems due to the buildup of salinity outside the area that micro-irrigation systems push salts vertically and laterally.

Keywords: Apparent soil electrical conductivity, drought, electromagnetic induction, electrical resistivity, Landsat 7, MODIS, remote sensing, soil

mapping, soil salinity, spatial variability.

Introduction

The world faces an unprecedented crisis in water resource management due to the increasing demand on finite water supplies from an ever-growing world population, and the changing climate patterns that disrupt the replenishment of surface and ground water resources. This has profound implications for global food security, protection of human health, and maintenance of aquatic ecosystems. Jury and Vaux (2007) provide an insightful look into the emerging global water crisis, identifying a definitive and imminent need for water conservation and management throughout the world, particularly for water-scarce and water-stressed countries.

The increase in population and urbanisation has resulted in severe water shortages throughout the world. In the mid-1990s eighty countries constituting 40% of the world's population were suffering from serious water shortages (CSD 1997). The World Water Council (2000) forecasts that by the 2020's the world will be 17% short of the water supply needed to feed the world's population. The United Nation's FAO (2011) indicates that by 2025 nearly 2 billion people will be living in countries or regions with absolute water scarcity, and two-thirds of the world's population could be living under stressed water conditions. In late 2011 the world population passed 7 billion and the United Nations forecasts that the world population will reach 9.7 billion by 2050. There are grave concerns that by 2050 40% of the global population (i.e., 3.9 billion people) may suffer from food shortage, economic deprivation, and poor health because of water stress.

These alarming statistics just scratch the surface of the compelling data pointing to the need for global water conservation and management. Most of these projections do not even account for the impacts of climate change on water supplies since high levels of uncertainty exist in climate model predictions, particularly at regional and local scales. However, climate change models do agree on some basic global trends. The models indicate that climate change is expected to affect primarily precipitation, potential evapotranspiration (ET), and temperature, which will likely influence the severity and occurrence of droughts and floods.

Evidence of the impact of the change in climate patterns is found worldwide. For instance, the USA experienced an increase in moderate to severe levels of drought, particularly in the Southwest, but other areas of the USA are not exempt (e.g., Midwest and Southeast). Arguably, the most notable example in North America from a public-awareness perspective is the California drought, which according to the U.S. Drought Monitor lasted

376 weeks beginning on December 27 2011 and ending on March 5 2019. This drought caused drastic reductions in irrigation water allocations to farmers in the agriculturally productive San Joaquin Valley (SJV) and heightened water conservation measures in urban areas, particularly over the period of 2011 to 2015. Other recent worldwide droughts include a one-in-a-thousand-year drought in Australia, which began in 1995 and continued until 2009; Spain's drought in Catalonia; northern India's drought in the first decade of the new millennium; and droughts in northern China, Syria, and south-eastern Brazil. Recent floods include Queensland, Australia; Tennessee, Arkansas, Texas, Wisconsin in the USA; Pakistan; and India. Even though there is no short-term extreme weather event that can be conclusively attributed to climate change, there is a statistical record of these events showing that they are clearly occurring with increased frequency and/or intensity (Dai 2011).

There is an expectation that climate change will bring increased frequency of extreme weather events around the globe, with unusually high rainfall events leading to floods; and low precipitation, higher temperatures, and higher potential ET resulting in longer, harder, and more frequent droughts. The National Centre for Atmospheric Research released a collection of drought severity maps projecting drought levels using the Palmer Drought Severity Index for the time period 2000-2099. The drought maps show that much of the Western Hemisphere and large parts of Eurasia, Africa, and Australia will experience extreme drought while higher-latitude regions from Alaska to Scandinavia are likely to become wetter and experience flooding.

Agriculture accounts for more than 70% of the freshwater drawn from lakes, rivers, and underground sources (CSD 1997). Ironically, some of the most crop productive areas of the world occur in water-scarce regions, such as the arid southwestern USA (e.g., California's San Joaquin and Imperial-Coachella Valleys) and other arid regions of the world including the Middle East; the Hai He, Huang He and Yangtze basins in China; and along the Nile River in Egypt and Sudan. In most cases these areas owe their successful crop productivity to mild year-round climates and available sources of surface and/or ground water for irrigated agriculture. Climate change will influence global rainfall patterns, influencing both the amount and distribution of rainfall. Global climate change model predictions indicate decreased precipitation for drier regions of the world, with annual average precipitation decreases likely to occur in most of the Mediterranean, northern Africa, northern Sahara, Central America, the American Southwest, the southern Andes, as well as southwestern Australia (IPCC 2007).

Climate change impact on soil in relation to crop productivity

Agriculture and climate change are directly linked. Crop yield, water use, biodiversity, and soil health are directly affected by changes in the climate. Changes in the frequency and intensity of rainfall, temperature, and other extreme weather events will impact agricultural productivity, with the net effect of climate change on world agriculture most likely negative. Even though some regions and crops may benefit, many will not. Increases in atmospheric CO₂ will likely increase organic matter in soil and stimulate growth and improve water use efficiency in some crops, but heat waves, droughts, and flooding may dampen these potential yield increases. Furthermore, indirect climatic impacts, such as greater competition from insects, weeds, and pathogens will decrease yield.

Brevik (2013) provides an excellent review of the potential impact of climate change on soil properties and processes that influence plant productivity. Recent research has indicated that increased atmospheric CO₂ may not have as large an influence on plant productivity as once thought (Poorter and Navas 2003; Zavaleta et al. 2003; Long et al. 2005; Körner 2006; Jarvis et al. 2010; Zaehle et al. 2010). Ostensibly, increasing ozone levels may counteract the CO₂ effect (Long et al. 2005). Furthermore, the negative effects of increased temperature on plant growth may also counteract the CO₂ effect (Jarvis et al. 2010). Nitrogen (N) and phosphorus (P) limitations may also play a role. Niklaus and Körner (2004) found in a long-term elevated CO₂ grasslands experiment that after two years N and P were limiting the biomass production expected from high CO₂ levels. There is also evidence that increased CO₂ levels may not necessarily increase C sequestration. Carney et al. (2007) found that due to increased microbial activity, soil organic C levels declined under increased CO₂ levels. Temperature can also exacerbate matters. Increased temperatures cause increased CO₂ to be released from soil into the atmosphere, which increases ambient temperatures more. This continues until a new equilibrium is reached.

Even though global projections of climate change paint a dark picture with respect to extreme weather event occurrence and frequency, and as many negative as positive potential impacts on agriculture are likely to occur, the fact remains that global projections have less associated uncertainty than regional predictions; consequently, effort is needed to monitor regional- and local-scale impacts. Continuous monitoring of climate change impacts on soil health and condition are essential to identify and quantify trends requiring the development of management strategies

that will ameliorate detrimental impacts to crop productivity, especially in highly productive agricultural areas. The sustainability of agriculturally productive arid and semi-arid areas, such as California's SJV, depends upon the timely knowledge of the geospatial impacts that changes in climate patterns will have on the soil properties influencing crop yield.

One notable soil property that is seldom mentioned by scientists when speaking about the impacts of climate change is soil salinity. This may be because salinity is among the most spatially complex and temporally dynamic soil property, with a coefficient of variation generally over 80-90% (Corwin et al. 2003). Or it may be because, unlike many other soil properties, salinity can be easily managed by the addition of water to leach salts. As long as the water source is plentiful and of sufficiently good quality, then salinity is generally not regarded as a problem of concern. However, if droughts become more frequent, as climatologists predict, then salinity will likely become an issue of concern.

Soil salinity is a worldwide concern in arid and semi-arid agricultural areas. Salt-affected soils are estimated to comprise 23% of the cultivated land, approximately 3.5×10^8 ha (Massoud 1981). In actuality, however, there are no directly measured global inventories of soil salinity. All known global inventories of soil salinity, and with only a few exceptions, all known regional-scale inventories are gross approximations based on qualitative and not quantitative data (Lobell 2010; Lobell et al. 2010). Current global inventories of soil salinity are estimates at best.

Any climatic pattern changes that influence the salinisation process will alter the accumulation of soil salinity. For instance, the west side of the SJV has shallow water tables due to local and basin hydrological effects. Under normal conditions, irrigation water applications leach salts from the root zone. However, during drought conditions no water is available for irrigation causing much of the land to be fallow. Upward water movement resulting from capillary rise from the shallow water table causes salts to accumulate at the soil surface.

Soils are spatially heterogeneous. This is particularly true for soil salinity, which is a dynamic soil chemical property exhibiting complex spatial patterns that vary over time. These complex spatio-temporal patterns make inventorying and monitoring salinity at a field scale or at larger spatial extents especially challenging. Techniques that provide rapid, reliable, and detailed spatio-temporal geo-referenced measurements of salinity at multiple scales are crucial for inventorying, monitoring, and managing soil salinity impacted by changes in climate patterns as well as by natural on-going salinisation processes associated with irrigated agriculture. It is the objective of this chapter to present a case study for the west side of

California's SJV, evaluating the impact of climate change on salinity development utilising the proximal and satellite sensor approaches developed by Corwin and colleagues for assessing soil salinity across multiple scales.

Proximal and satellite sensor approaches for assessing soil salinity across multiple scales

Historically, three methods have been developed for determining soil salinity at field scales and larger spatial extents: (1) visual crop observations, (2) geospatial measurements of apparent soil electrical conductivity (EC_a), and most recently (3) multi- and hyper-spectral imagery. Visual crop observation is the oldest method of determining the presence of soil salinity in a field. It is a quick method, but it has the disadvantage that salinity development is detected after crop damage has occurred. For obvious reasons, the least desirable method is visual observation because crop yields are reduced to obtain soil salinity information. Remote imagery, including multi- and hyper-spectral imagery, is increasingly becoming a part of agriculture and represents a quantitative approach to the antiquated method of visual observation that may offer a potential for early detection of the onset of salinity damage to plants. Even so, multi- and hyper-spectral remote imagery are still in their infancy, with an inability at the present time to differentiate osmotic from matric or other stresses, which is key to the successful application of remote imagery as a tool to map salinity and/or water content. The most common method of mapping salinity at field scale is geospatial measurements of EC_a .

For field-scale mapping applications, the measurement of salinity shifted in the 1970s and 1980s from the measurement of the EC of soil solution extracts to the measurement of the apparent soil electrical conductivity (EC_a , $dS\ m^{-1}$), which is the measurement of the EC of the bulk soil. The geophysical techniques of electrical resistivity (ER) and electromagnetic induction (EMI) measure EC_a . The EC of the bulk soil (i.e., EC_a) is the measurement of anything conductive in the soil, not just salinity. The shift to measuring EC_a occurred because the time and cost of obtaining soil solution extracts prohibited their practical application at field scales due to the high spatial variability of soil. In contrast to EC measurements of soil solution extracts, EC_a is a very rapid and easy to take measurement that can be readily mobilised to obtain tens of thousands of geo-referenced measurements of EC_a in a comparatively short time period, providing field-scale maps of EC_a . The use of EC_a to measure salinity has the further advantage of increased volume of measurement. However, EC_a suffers from

the complexity of measuring EC for the bulk soil rather than being restricted to the solution phase where salts reside, making interpretation of the spatial EC_a data difficult. Furthermore, EC_a must be converted to EC_e since EC_e is the reference point for salinity, requiring a calibration between EC_a and EC_e . Even with these challenges, the practical application of EC_a far outweighs the negative aspects. Corwin and Lesch (2005a) review the use of EC_a in agriculture.

At present, no other measurement provides a greater level of spatial soil information than that of geospatial measurements of EC_a when used to direct soil sampling (Corwin and Lesch 2005a). The characterisation of spatial variability using EC_a measurements is based on the hypothesis that spatial EC_a information can be used to develop a directed soil sampling plan that identifies sites that adequately reflect the range and variability of soil salinity and/or other soil properties correlated with EC_a . This approach is referred to as EC_a -directed soil sampling.

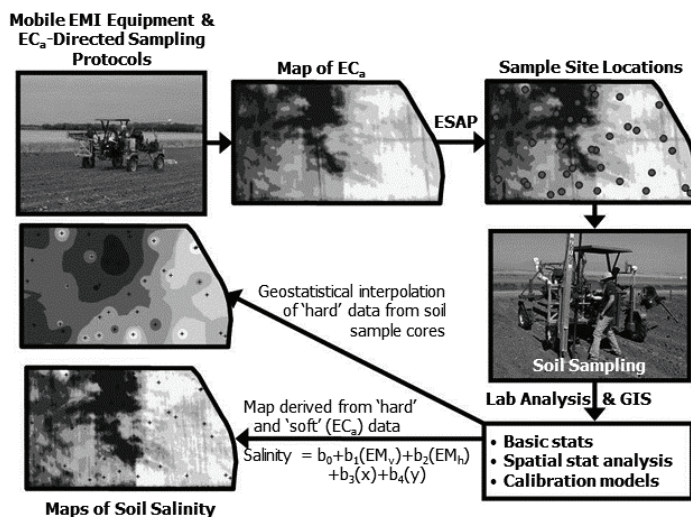


Figure 19.1 Schematic of the integrated system to assess field-scale soil salinity using apparent electrical conductivity (EC_a) directed soil sampling protocols, a mobile electromagnetic induction (EMI) rig, ESAP software, and geographic information system (GIS). EM_v refers to the measurement of EC_a by electromagnetic induction (EMI) in the vertical coil configuration and EM_h refers to measurement of EC_a in the horizontal coil configuration. Taken from Corwin (2015).

Scientists at the U.S. Salinity Laboratory have developed an integrated system for the measurement of field-scale salinity consisting of (1) mobile EC_a measurement equipment (Rhoades 1993), (2) protocols for EC_a-directed soil sampling (Corwin and Lesch 2005b; 2013), and (3) sample design software (Lesch et al. 2000). The integrated system for mapping soil salinity is schematically illustrated in Figure 19.1.

Since 1980, the U.S. Salinity Laboratory within USDA-ARS has been the centre of research related to the mapping and monitoring soil salinity at field scale and larger spatial extents using ER and EMI (Corwin 2008). Over that time, USDA-ARS scientists and visiting scientists have developed three approaches for mapping soil salinity at three distinct spatial scales: field (< 3 km²), landscape (3-10 km²), and regional (10-10⁵ km²). Each approach is based on the measurement of EC_a. The three approaches are: (1) EC_a-directed soil sampling (field scale, < 3 km²), (2) ANOCOVA of EC_a-directed soil sampling (landscape scale, 3-10 km²), and (3) satellite imagery calibrated from ground truth soil samples obtained from EC_a-directed soil sampling (regional scale, 10-10⁵ km²). Each approach has been reviewed in the papers by Corwin and Lesch (2005a; 2013; 2014; 2017) and Corwin and Scudiero (2019; 2020). The chapter on Soil Salinity of the current book also provides an overview of these approaches.

Impact of climate change on soil salinity development: West side of the San Joaquin Valley

The Central Valley of California, which includes the SJV, the Sacramento Valley, and the Sacramento-San Joaquin Delta, comprises less than 1% of the USA's farmland, but provides about 25% of the USA's table food (Cone 1997). However, agricultural practices are challenged by extensive areas of salinity and sodicity, particularly in the western San Joaquin Valley (WSJV). The WSJV soils are derived from saline alluvium originating from California's Southern Coast Range (Letey and Dinar 1986). In the early 1980s, Backlund and Hoppes (1984) estimated about 39% (i.e., 9×10^5 ha) of the irrigated land in the SJV was saline and/or sodic.

The California drought, particularly over the years 2011-2015, had the greatest economic impact on agriculture in the San Joaquin Valley in recorded history. The impact as estimated by University of California-Davis economists and scientists was \$2.74 billion (Howitt et al. 2015). The economic impact is based on insufficient water to grow crops, which caused hundreds of thousands of hectares of land to become fallow. According to the CropScape database (Han et al. 2012), during the California drought that began in 2011, fallow land went from 12.7% (2007-2010 average) to 19, 21,

22, 26, and 34% in 2011, 2012, 2013, 2014, and 2015, respectively. In 2014, approximately 1.95×10^5 ha of land in the Central Valley was fallow, resulting in \$810 million in lost crop revenue (Howitt et al. 2015). However, the estimated financial impact on agriculture did not take into account the future ramifications that placing land into fallow would have, such as the impact of drought on salt accumulation in the soil.

For centuries farmers have been well aware of the inexorable link between soil salinity and crop yield. Scientists have quantitatively described the relationship between salinity and yield through the piece-wise linear salt tolerance equation (Maas and Hoffman 1977):

$$Y_r = 100 - b (EC_e - a) \quad [1]$$

where Y_r represents the relative crop yield, a is the salinity threshold (dS m^{-1}) and b is the slope expressed % per dS m^{-1} . Representative salt sensitive, moderately sensitive, moderately tolerant, and tolerant crops include carrot, broccoli, wheat, and cotton with salinity thresholds (a ; dS m^{-1}) and slopes (b ; % per dS m^{-1}) of 1.0 and 14.0, 2.8 and 9.2, 6.0 and 7.1, and 7.7 and 5.2, respectively (Maas and Hoffman 1977). In short, beyond a specific soil salinity threshold, which is characteristic to each plant and variety of plant, crop yield will decrease linearly, where the yield decrement is also plant specific.

In arid or semi-arid irrigated agricultural areas such as the SJV, sustaining crop production requires irrigation management practices that prevent excessive root-zone salt accumulation. The primary source of salts is a consequence of the process of evapotranspiration where salts in the irrigation water accumulate in the root zone due to evaporation at the soil surface and removal of pure water by the plant roots, which leaves salts behind. To prevent the accumulation of salts in the root zone, additional irrigation water beyond the consumptive water use by the crop is applied, which leaches the excess salts. In areas where the water table is deep, the additional irrigation water simply percolates below the root zone carrying the salts with it, but in areas where the water table is shallow the ground water is degraded by the salts leached from the root zone and the water table becomes more and more shallow, unless it can be channelled away through the use of tile drains. Without tile drains and a place to dispose of the drainage water, the situation becomes problematic since salts can readily move back into the root zone as soon as the downward flux of water stops. Adhesive and cohesive forces result in the capillary rise of water from the shallow water table carrying salts in the ground water back into the root zone. The finer textured the soil, the greater the capillary rise will be, which

can ultimately result in the substantial accumulation of salts at the soil surface in high clay soils. If this occurs, then crop yields will decline.

The drought in California (2011-2015) resulted in drastically reduced irrigation water allocations to farmers rendering substantial land in the SJV left fallow. Drought and salinisation go hand-in-hand under conditions of shallow water tables and fine textured soils. Fallow land, particularly in the WSJV where water tables are often shallow and the soil is fine textured, becomes saline in a relatively short time due to the upward movement of salts in the soil solution resulting from capillary rise from the water table (Corwin 2012).

The California drought, resulting from changes in climate patterns, exacerbated soil salinity levels. Three studies provide insight into the impact of climate change on soil salinity accumulation in the WSJV. Each study used the previously described EC_a -directed soil sampling methodology as a basis for monitoring soil salinity in the root zone (i.e., 0-1.2 m depth increment) at three different scales: field ($< 3 \text{ km}^2$), multiple-field or landscape ($3\text{-}10 \text{ km}^2$), and regional ($10\text{-}10^5 \text{ km}^2$).

Recent studies by Corwin and his colleagues showed that reuse of saline irrigation water can reclaim saline-sodic soil (Corwin et al. 2006; 2008). In a long-term, field-scale study of the reclamation of a saline-sodic soil using $3\text{-}5 \text{ dS m}^{-1}$ drainage water on a 32.4-ha field located on Westlake Farm in WSJV, Corwin (2012) found that from 1999 to 2009 there was steady improvement in soil quality due to decreases in salts, molybdenum (Mo), boron (B), and the sodium adsorption ratio (SAR), resulting from leaching. Sodic soils have very low permeability due to the dispersive effect of sodium (Na) on clay particles, which causes the clay to flocculate reducing the hydraulic conductivity. The decrease in salinity, Mo, B, and SAR was attributed to the presence of calcium (Ca) in the low-quality drainage water used for reclamation. The Ca displaced the Na on clay exchange sites causing the soil particles to aggregate thereby improving the permeability. The improved soil quality from 1999 to 2009 is shown in Figure 19.2. However, in 2011 when the California drought began there was no fresh or drainage water available for irrigation; consequently, the field was left fallow, receiving only rainfall. Within 18 months after irrigation with drainage water had ceased and the field was left fallow, the levels of salinity, Mo, B, and SAR returned to nearly their original 1999 levels and in some instances the levels were even higher (compare 1999 to 2011 for EC_e , SAR, B, and Mo in Figure 19.2). This field is typical of fields located in the WSJV near the San Joaquin River where the soils are generally fine textured and the water table is perennially shallow (i.e., within 2 m of the soil surface). It can be expected that all fallow lands in the WSJV with shallow water

tables would experience a similar increase in salinity and trace elements due to the capillary rise effects from the shallow water table causing the upward movement of salts and trace elements.

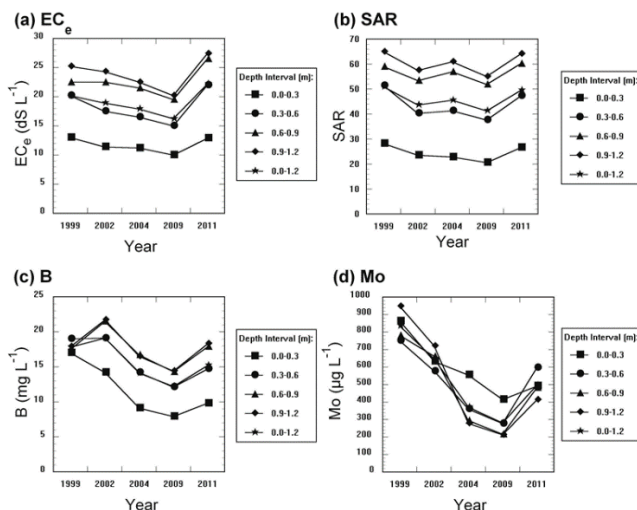


Figure 19.2 Graphs showing the improvement in soil quality from 1999 to 2009 for (a) salinity (EC_e), (b) sodium adsorption ratio (SAR), (c) boron (B), and (d) molybdenum (Mo) followed by the return to original levels by 2011 after drought interrupted further application of drainage water to reclaim the saline-sodic soil. Source: Corwin (2012) with permission.

A comparison of salinity levels for 2,400 ha of the former Broadview Water District in the WSJV from 1991 to 2013 shows an increase in soil salinity occurred (Figure 19.3). In February 2005, Broadview Water District land became fallow as water rights were sold to Westlands Water District (Wichelns and Cone 2006). Figure 19.3a shows a map of salinity for Broadview Water District in 1991 obtained from the E_{Ca} -directed soil sampling approach of Corwin et al. (1999) and Corwin and Lesch (2003, 2005b). Figure 19.3b shows the increase in soil salinity from 1991 to 2013 as indicated by the percentage of the total acreage falling within salinity classes of 0-2, 2-4, 4-8, 8-16, and > 16 dS m^{-1} . From 1991 to 2013 the 2-4 dS m^{-1} salinity class showed a substantial decrease from 27% to 3% while the 8-16 dS m^{-1} class significantly increased from 3% to 33%. The field average soil salinity increased by 43% within the root zone during this time

period. Essentially, all the non-saline and slightly saline soils in 1991 became moderately (4-8 dS m⁻¹) or strongly saline by 2013, which is primarily due to the upward movement of salts from the shallow water table resulting from capillary rise.

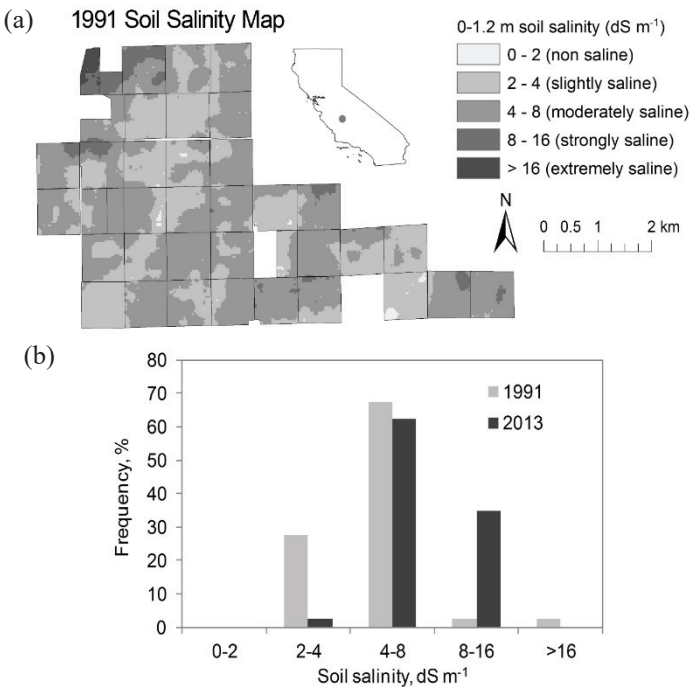


Figure 19.3 Broadview Water District: (a) map of soil salinity levels within the root zone (0-1.2 m) for Broadview Water District, CA in 1991 (Source: Corwin et al. 1999) and (b) histogram of soil salinity categories as predicted for 2013 using the soil salinity assessment model of Scudiero et al. (2014).

At a regional scale, there are strong indications that soil salinity for the root zone (0-1.2 m) increased for the WSJV over the past three decades. Estimates of salt-affected soils (i.e., $EC_e > 4$ dS m⁻¹) for the WSJV calculated from data presented by Backlund and Hoppes (1984) were approximately 0.45×10^5 ha in 1984. Scudiero et al. (2014; 2015; 2017) estimated salt-affected soils for WSJV were approximately 0.6×10^5 ha in 2013. The increased salinisation of the WSJV from 1984 to 2013 may or may not be attributed solely to changes in climate patterns since there are

no reliable estimates of salt-affected soils for WSJV in 2011 when the California drought began. Nevertheless, salinisation of the WSJV did occur between 1984 and 2013.

There is strong evidence suggesting that climate change patterns are impacting the salinisation of agricultural land in the SJV (Corwin 2021). From the field-scale observations for Westlake Farm, the landscape-scale observations for Broadview Water District, and the regional-scale estimates of salinity for the WSJV, it is surmised that significant increases in root-zone soil salinity are occurring due to impacts from climate change. Even though the occurrence of above average annual rainfall in northern California in 2016 ameliorated water scarcity conditions that existed in California's SJV from 2011-2015, there are clear indications that salt accumulation in soil is occurring and will likely continue to occur on lands with the right conditions (i.e., fallow, shallow water table, and fine-textured soil). Since 2015 there has been a steady shift in the SJV from flood and sprinkler irrigation of low-cash crops to micro-irrigation of high cash crops. The shift to micro-irrigation will subsequently result in potential future problems due to the buildup of salinity outside the area that micro-irrigation systems push salts vertically and laterally (Corwin et al. 2022). As a consequence of changes in climate patterns, salt accumulation will most likely occur in irrigated agricultural areas around the world where shallow water tables and fine textured soils exist.

The implications of the California drought and its impact on soil salinity are self-evident. Less available fresh water for irrigation necessitates a shift to the use of impaired water sources (e.g., municipal, ground, and drainage waters) as alternative sources of water and to high efficiency irrigation systems (e.g., drip, buried drip, and micro-sprinkler irrigation systems). Concomitantly, low quality water reuse management guidelines are needed. The potential accumulation of soil salinity in the root zone requires (1) spatial knowledge of soil salinity levels for the site-specific management of soil salinity at farm levels to optimise scarce water resources, (2) the development of water use and regulatory guidelines at state and federal levels, and (3) assessment of climate change impact trends at global levels. To obtain the necessary spatial soil salinity information, a multiple-scale infrastructure is needed.

Global, national, state/province, and local infrastructures are needed that will routinely monitor climate change impacts, including salinity, at multiple scales. The guidelines and protocols are currently available to monitor soil salinity cost effectively at multiple scales with ground- and satellite-based sensors. At local scales where salinity is a concern, irrigation and drainage districts need field-scale monitoring equipment (e.g., EM

conductivity meters and GPS equipment) and a trained member of their staff to conduct routine field-scale soil salinity assessments for producers without charge. Funds to cover the collection of this information can be shared by the irrigation/drainage districts, and state and federal programs. The field-scale data would be passed to producers for site-specific management use and to state and federal agencies for compilation, display, and interpretation to provide state and federal decision makers controlling water resources with the information to make informed water management decisions at state and national levels. A commitment from global organisations (e.g., United Nation's Food and Agriculture Organisation, FAO) is needed to compile national data throughout the world where it is available. Soil and agronomic researchers, irrigation specialists, producers, consultants, conservation and irrigation districts, commodity groups, cooperative extension, federal and state regulators, and agencies such as National Resource Conservation Service (NRCS), Environmental Protection Agency (EPA), U.S. Geological Survey (USGS), Bureau of Reclamation, and the United Nation's Food and Agriculture Organisation (FAO) will directly benefit from timely salinity monitoring maps and the spatial information derived from ground- and satellite-based sensors.

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Abbreviations

EC _a :	Apparent soil electrical conductivity (dS m ⁻¹)
EC _e :	Electrical conductivity of the saturation extract (dS m ⁻¹)
EM _h :	Electromagnetic induction measurement in the horizontal coil configuration
EMI:	Electromagnetic induction
EM _v :	Electromagnetic induction measurement in the vertical coil configuration
ER:	Electrical resistivity
GPS:	Global positioning system
SJV:	San Joaquin Valley
SP:	Saturation percentage
TDR:	Time domain reflectometry
WSJV:	West side of the San Joaquin Valley

CHAPTER 20

SOIL LIMITATIONS FOR CROP PRODUCTIVITY IN SOUTH AMERICA

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Abstract

The agricultural practices introduced by European colonisers have been practiced in South America for many years, resulting in depletion of the soil's natural fertility. The introduction of fertilisers and acidity amendments occurred in the mid-1960s and boosted production of many areas, but still with high soil erosion and low levels of organic matter. The widespread use of conservation systems, such as no-tillage, occurred latter and changed the relationship between soil indices and crop responses. Nowadays, South America represents 47% of the total global area under no-tillage, which covers an area around 56 million hectares. No-tillage reduces the annual rate of decomposition and increases the mean residence time of the soil organic matter. One of the great challenges in different countries nowadays, in the most diverse cropping systems, is to increase biodiversity through the proper use of different species of cover crops, mainly mixed cover crops, in

order to improve the microbiota, achieve better soil-plant equilibrium and contribute strongly to enhance the soil organic carbon (C) sequestration. In some countries, the challenge of no-tillage adoption is getting closer to being overcome. The current challenge is working on a cropping system with diversified species and high residue input. The adoption of legume cover crops as a source of nitrogen (N) seems to be very important, resulting in higher accumulation of soil organic matter compared to N fertilisation. The management of soil acidity was, and continues to be, one of the main factors limiting crop yields. The problems of diagnosis are mainly related to the sampled soil layer and the indicators used for decision making. Recent studies have demonstrated that neither the 0-10 cm nor the 0-20 cm soil layer is suitable for diagnosing soil acidity and the potential crop yield in no-tillage systems with chemical restrictions in the subsurface. In these areas, a stratified soil analysis is essential, covering at least one subsurface layer (10-20 cm). The incorporation of limestone may be the best and fastest way to eliminate problems related to soil acidity in the subsurface. Significant increases in crop yields have been observed when using agricultural gypsum based on the diagnostic soil layer of 20-40 cm. Doses between 2 and 3 Mg ha⁻¹ are sufficient to obtain 95% of the maximum crop yield. For phosphorus (P), there is no doubt that the biggest problem is restricted access: by farmers at the micro scale and by countries at the macro scale. When accessible, the inappropriate use of P fertilisers is often noticed. The correct approach would be to raise the available P content above the critical level in the 0-20 cm soil layer, and then reapply the amount exported by crops in the row at sowing time. Regarding potassium (K), although there is an assumption that the tropical soils found in South America have only minerals such as kaolinite and oxides, there are several studies that show that the mineralogy of these soils is not so uniform. It is common to observe situations where 2:1 clay minerals are present and crops do not respond to the addition of K fertiliser, or the available K content does not increase over time. The research on sulfur (S) has advanced and shown that in tropical soils there is a higher positive crop response to S addition than in subtropical soils, regardless of available soil S contents. The evaluation of the 20-40 cm soil layer can support decision making regarding S management. To enhance production of plants and to increase the soil organic matter content, it is necessary to encourage and promote the horizontal and vertical monitoring of soil fertility. Associated with this, it is necessary to establish research networks aimed at improving the establishment of critical levels of soil acidity and available nutrients in the soil to guide decision-making more assertively, thus maximising productivity and promoting more sustainable production.

Keywords: Organic matter, no-tillage, nitrogen, phosphorus, potassium, sulfur, soil acidity, aluminium.

Introduction

Agriculture in South America existed long before the arrival of Europeans. Local communities have used agriculture based on some conservation principles for millennia, such as direct sowing without soil plowing and maintaining a protective soil cover on the ground (Calegari et al. 2020). Although the production systems were not intensive or highly productive, they were sustainable and generated minimal environmental impact. However, after 1492, agriculture introduced by European colonisers was practiced, based on burning, fallow, monoculture, and intensive plowing. This model of agriculture rapidly depleted the soil's natural fertility and organic matter stocks (Tiecher et al. 2020). After the introduction of fertilisers and acidity amendments in the mid-1960s during the Green Revolution, previously unproductive areas were recovered and agricultural production boosted, but still with a high potential for soil loss through water erosion and with great difficulty in recovering the original levels of soil organic matter (SOM).

In the last three decades, the widespread use of conservation systems such as no-till, especially when associated with the use of crop rotation, has slowed soil degradation, increased soil carbon (C) stocks, and maintained soil fertility with less dependence on external inputs (Calegari et al. 2008). However, even with these improvements, technological advances in phytosanitary control, and genetic advances, in many cultivation areas it is possible to observe a stagnation in crop productivity. Much of this is due to the strong gradient of acidity and nutrient availability with depth due to the lack of soil disturbance associated with the addition of lime and fertilisers on the soil surface (Bellinaso et al. 2021). It is increasingly evident that horizontal monitoring of soil fertility alone is not enough to explain crop response, and it is increasingly necessary to understand the dynamics of soil acidity, nutrient availability, and SOM accumulation in areas under no-till. In this chapter, the main challenges of managing soil acidity, nutrients and organic matter in South America will be addressed.

Management of carbon and nitrogen in the soil

In the tropical and subtropical soils of South America, SOM accumulation is essential for more sustainable and resilient agricultural

systems, as SOM drives many soil properties that strongly impact crop yield and environmental quality (Lal 2004). The benefits of SOM are mainly related to water retention, improved soil structure, reduce Al activity, increased cation exchange capacity and improved nutrient cycling. However, the intense solar radiation and high precipitation covering the majority of South America favour highly weathered soils and high SOM decomposition rates. At the same time, these climatic conditions also favour the cultivation of a wide diversity of crops over the year. To favour SOM accumulation in the agricultural soil of South America, management systems that decrease the SOM decomposition rate and increase crop residue input are recommended to achieve a positive balance between the input and output of C.

Tillage systems

No-tillage (NT) is the basis of conservation agriculture and is considered an important tool for the sustainability of agricultural activities in tropical and subtropical regions (Lal et al. 2007). The area under NT has been growing steadily all over the world, but the highest rates of adoption have been achieved in South America. South America represents 47% of the total global area under NT, which covers an area around 56 million hectares (Derpsch and Friedrich 2009). In Argentina and Brazil, the countries with the greatest area under NT in South America, 70% of NT is permanent, which means that once started most farmers never till again.

Switching from conventional tillage (CT) to NT can result in the sequestration of an average of $0.48 \text{ Mg ha}^{-1} \text{ year}^{-1}$ of carbon in Brazilian subtropical regions (Bayer et al. 2006). In comparison to CT, the elimination of soil tillage leads to a marked reduction of the annual rate of SOM decomposition (0.040 year^{-1} in CT versus 0.025 year^{-1} in NT). This reduction is mainly associated with increases in soil macroaggregates, which physical protect SOM from microorganisms and their enzymes (Veloso et al. 2019). The less oxidative soil environment in NT increases the mean residence time of the SOM (the time between the photosynthesised C input in the soil until its microbial oxidation to CO_2) from 25 to 40 years. Thus, the permanence of the SOM is greater in NT compared to CT, which amplifies the benefits of the cycling of SOM.

Besides environmental conditions, the decomposition of SOM is strongly influenced by soil particle distribution and mineralogy. Bayer (1996) found higher decomposition rates and a greater impact of NT in reducing decomposition rates relative to CT in an Acrisol with 22% clay and predominantly kaolinite mineralogy than in a Oxisol with 68% clay

(with predominance of oxides). The organo-mineral interaction is the main mechanism controlling the long-term stability of organic matter (Kogel-Knabner et al. 2008), and microorganisms have difficulty oxidising organic matter bound at the mineral surface (Sollins et al. 1996).

Influence of cover crops

In some countries of South America, such as Brazil, Argentina, Paraguay, and Uruguay, the challenge of converting CT into NT is getting closer to being overcome. The current challenge is identifying cropping systems that have diversification of species and high residue input, without making big changes to the farm budget. One alternative to keep the traditional main cash crops (soybean, maize, cotton, coffee, and wheat) cultivated in South America, is the adoption of cover crops between the growing seasons. In this way, we can combine the benefits of high value crops with species that aid the environment in the same agricultural system.

The adoption of cropping systems, considering cash and cover crops, will directly impact the biomass input that drives SOM accumulation. In Southern Brazil, SOM accumulation rates strongly depend on the cropping system adopted and can vary between 0.11 and 0.68 Mg C ha⁻¹ yr⁻¹ (Bayer et al. 2009) and achieve values up to 1.24 Mg C ha⁻¹ yr⁻¹ (Calegari et al. 2008). The residue input from roots and shoots plays a key role in determining SOM accumulation and can be responsible for 80% of the differences in SOM stock in different cropping systems (Veloso et al. 2018). It is interesting to note that some researchers have found that the roots show greater conversion into SOM compared to the shoots and contribute more to SOM stocks (Tahir et al. 2015).

In addition to residue input, species diversification is another important point to consider when seeking to increase SOM accumulation. In cropping systems where maize is the main cash crop, the introduction of legume cover crops can further increase the potential for SOM accumulation. Legume cover crops add C and N from their own residues (roots and shoots) and increase biomass input from the subsequent crop. Veloso et al. (2018) found that legumes added between 82 and 115 kg N ha⁻¹ yr⁻¹ when one or two legume cover crops were added to the cropping system, which increased the maize biomass cultivated in the subsequent crop by 1 Mg ha⁻¹ yr⁻¹. Also, the greater lability of legumes may result in a greater conversion of residues into SOM. Compared to N fertilisation, the adoption of legumes more than doubled (0.22 vs 0.09 kg SOC) SOM accumulation: in other words, each kg of C input by the legume cover crops resulted in the accumulation of 0.22 kg of soil organic carbon, compared with only 0.09 kg soil organic carbon

for N fertilisation (Veloso et al. 2018).

The most cultivated legume cover crops during the autumn/winter period of subtropical regions of South America, are white clover (*Trifolium repens* L.), common vetch (*Vicia sativa* (L.)), hairy vetch (*Vicia villosa* Roth), blue lupin (*Lupinus angustifolius* (L.)), corniculata (*Lotus corniculatus*), among others. The radish (*Raphanus sativus* (L.)), although not a legume, also plays an important role in biomass input, and helps recycle N from deeper in the profile in the winter seasons. Regarding the legumes cover crops that can be cultivated during spring/summer seasons, some examples are: caupi (*Vigna unguiculata* (L.) Walp), crotalaria (*Crotalaria* sp.), feijão-guandu (*Cajanus cajan* (L.) Millsp), lablab (*Lablab purpureus* (L.) Sweet), and mucuna (*Mucuna pruriens* (L.) DC) (Bayer et al. 2019).

Management of soil acidity

Liming

Soil acidity was, and continues to be, the main factor limiting the crop productivity of tropical and subtropical agriculture in Brazil (Bellinaso et al. 2021; Pias et al. 2020), Argentina (Alvarez et al. 2020), Chile (Bonomelli and Artacho 2021), Uruguay (Beretta et al. 2017), Paraguay (Kawavata 2018), and Colombia (Caballero et al. 2019). Exceptions to this problem are places with low rainfall, in poorly weathered and poorly drained soils, or near the sea, where the acidity gives way to problems related to excess salts (Denardin et al. 2020). The use of soil acidity correctives has been a common practice since the beginnings of agriculture and is still used as a “key” input in highly weathered acidic soils in South America. Among the inputs used for agricultural production, lime is certainly the one that results in the greatest response in crop yields and with the greatest residual effect, therefore, with an excellent cost-benefit ratio.

Our understanding of the dynamics of soil acidity and its effects on soil chemical properties and plant response has greatly evolved from the 1960s to the late 20th century (Rheinheimer et al. 2018). Criteria were established to estimate the efficiency of correctives based on their chemical composition and particle size distribution, which were calibrated for cropping systems with intense tillage and soil mobilisation. However, these criteria may not be as efficient in conservation management systems, such as NT, in which lime is broadcast. In NT systems, which cover huge areas in Brazil, Argentina, Uruguay, and Paraguay, there is still great uncertainty regarding the best acidity diagnosis methods, lime rates, and the type and mode of

application of lime.

The problems of diagnosing soil acidity are mainly related to the sampled soil layer and the chemical indicators used for decision making. There is no doubt that for acid-sensitive species, the acidity of the entire volume of soil explored by the roots should be corrected. This would avoid problems related to toxicity with Al^{3+} and would also increase the availability of various nutrients. However, the correction of soil acidity for annual crops and cultivated pastures is generally restricted to the topsoil (up to 20-30 cm), as this is where most of the roots develop and concentrate, and also due to the large increase in corrective and operational cost when seeking to correct deeper soil layers (below 30 cm). In systems with soil tillage, or before the beginning of the NT system, the diagnostic layer must be 20 cm deep. The incorporation of lime promotes the correction of acidity evenly in this layer (Miotto et al. 2019). In NT systems, the diagnostic layer can be more superficial (0-10 cm) since the soil re-acidification is more intense in the first centimeters of soil (Rheinheimer et al. 2018).

Various chemical indicators can be used to diagnose soil acidity. The most commonly used are pH in water or salt (active acidity), saturation by aluminum (Al), and saturation by alkaline and alkaline earth cations [sodium (Na), potassium (K), calcium (Ca), and magnesium (Mg)]. As the soil pH influences the availability of all nutrients in the soil, and also Al toxicity, it is generally used as a key parameter to make liming decisions. But to define the lime dose, it is necessary to estimate the potential soil acidity (H^+Al), because it buffers the active acidity. Despite this, decision-making and the dose of lime to be applied are often established based on saturation by cations (Ca, Mg, K). Although Ca and Mg are commonly called bases or basic cations, even in soil fertility books, this term is not adequate and should be abolished (Lambers and Barrow 2020), as bases react or remove acids (H^+), which Ca and Mg do not. Of course, after liming, there is a simultaneous increase in soil pH and in the concentration of Ca and Mg, but the increase in the concentration of these nutrients is a consequence, and not the cause, of the decrease in soil acidity. Furthermore, we cannot assume that the relationship between Ca and Mg saturation and pH is linear and the same for all soils, as it varies widely depending on e.g., soil type and mineralogy.

The mode of lime application has been widely studied (Miotto et al. 2019; Rheinheimer et al. 2018; Bellinaso et al. 2021). Currently, there is great resistance on the part of technicians and farmers to incorporate lime in areas where NT is already established. In NT soils, generally, the pH of the superficial layer (0-5 cm) is corrected with superficial liming. Several studies have demonstrated the limitation of surface liming on the

neutralisation of subsurface acidity since the action of limestone is restricted to some centimeters below the deposition point (Rheinheimer et al. 2018). Changes in soil chemical properties related to acidity due to surface liming are insignificant below 10 cm in depth (Bellinaso et al. 2021). Furthermore, the higher pH of this layer can reduce or inhibit the reaction of larger particles of lime, since without acidity, carbonate solubilisation does not occur. Moreover, the soil pH and organic matter may be high enough to not adequately supply copper (Cu), zinc (Zn), and manganese (Mn) (Moreira et al. 2019), and at the same time, the root system suffers from Al toxicity a few centimeters below the soil surface (Miotto et al. 2019). Recent studies have demonstrated that neither the 0-10 cm soil layer nor the 0-20 cm soil layer is suitable for diagnosing soil acidity and the crop productive potential in NT systems with chemical restrictions in the subsurface (Bellinaso et al. 2021). In these areas, a stratified soil analysis is essential, covering at least one subsurface layer (10-20 cm). In the 10-20 cm layer, 20% of Al saturation is enough to reduce the maximum soybean yield by 20% (Bellinaso et al. 2021). In these situations, strategic tillage or occasional tillage with the incorporation of limestone may be the best and fastest way to eliminate problems related to soil acidity in the subsurface and ensure good root development and high crop yields.

Agricultural gypsum

After correcting the soil acidity with lime in the 0-20 cm layer, it is also possible to further reduce Al toxicity in the subsurface (below 20 cm) in NT areas using agricultural gypsum, which promotes high migration of Ca^{2+} , Mg^{2+} and K^{+} associated mainly with the sulfate anion (SO_4^{2-}) from the surface to deeper soil layers, as long as there is no mechanical impediment (soil compaction) (Alves et al. 2021). The increase of cations at depth promotes a reduction in the activity and toxicity of exchangeable Al^{3+} below 20 cm depth, enabling the deepening of the plant's root system and, consequently, improving the use of water and nutrients.

As with any other agricultural inputs, the use of gypsum should not be indiscriminate, but based on a technical position. According to a meta-analysis of the effect of using agricultural gypsum for grain crops under NT in Latin America by Pias et al. (2020), the 20-40 cm depth should be used as the diagnostic soil layer for decision-making on whether to apply agricultural gypsum, with the type of crop, Al saturation, and water availability also considered. In the absence of a water deficit, soybean responds positively to gypsum in soils with Ca content lower than 0.5 $\text{cmol}_c \text{ dm}^{-3}$ and Al saturation greater than 40% in the 20-40 cm layer. In the

presence of a water deficit, soybean responds when the Al saturation is greater than 10% and the Ca content is lower than $2.0 \text{ cmol}_c \text{ dm}^{-3}$. Corn and other winter cereals, on the other hand, have a high probability of productivity increase when the Al saturation is greater than 5% in the 20–40 cm layer, regardless of whether there is a water deficit or not.

When it is necessary to apply gypsum, doses between 2 and 3 Mg ha^{-1} are sufficient to obtain around 95% of the maximum crop yield (Pias et al. 2020). When the objective is to obtain maximum productivity, doses of up to 6 Mg ha^{-1} can be used and also guarantee a greater residual effect in the soil. However, care must be taken with the application of these high doses when the levels of exchangeable Mg in the surface layer are low ($<1.0 \text{ cmol}_c \text{ dm}^{-3}$), especially in sandy soils, because the application of large amounts of Ca via gypsum can induce Mg deficiency (Alves et al. 2021). An efficient strategy to reduce the undesirable effects of gypsum is the prior application of dolomitic limestone.

Deficiency of sulfur (S) in agricultural soils has become more common in recent decades due to the use of concentrated fertilisers with no S in their composition, reduction in the atmospheric deposition of S, reductions in SOM, and increases in the productive potential of crops (Pias et al. 2019). Agricultural gypsum is an excellent source of S, containing on average 15–18% S. Therefore, even when Al saturation in the subsurface (20–40 cm) is low, increases in soybean productivity can occur due to a greater supply of S to the plants. In these cases, the soybean response is even observed at low doses of gypsum.

Management of main macronutrients

Phosphorus-related limitations in agro-environmental systems

In South America, the literature contains abundant data on phosphate reserves and their use in agricultural production (Pavinato et al. 2020), the dynamics of phosphorus (P) in the soil (Rheinheimer et al. 2019), plant mechanisms for mining P from the soil (Lynch et al. 2021), biological interactions and their impacts on the use of P from the soil, and even the dynamics and impacts of the transfer of this nutrient from the terrestrial system to the aquatic environment (Bender et al. 2018). Thus, in this section, we will only list the authors' opinions regarding the main problems related to P in agro-environmental systems.

There is no doubt that the biggest problem related to P nutrition is the restricted access to it, either for farmers at the micro scale, or for countries at the macro scale. Poor countries have weathered soils that are low in

bioavailable P, and poor farmers are located on the more fragile soils with limited P availability. Even in rich countries that have very high yields, plants require increasing amounts of P, even depleting the soil's P reserves. Thus, the productivity of plants, and therefore of animal protein, is totally dependent on exogenous P. Therefore, there is a real "Phosphate Apartheid", either because of the situation of unequal P availability in the different biomes and, consequently, of the implemented agroecosystems, or because of the unequal location of P-rich mineral reserves. The two situations usually occur concurrently and play unfavourably against poor countries and farmers. Those with financial resources have access to suitable soil and can purchase phosphate fertilisers; those who are outside the capitalist system cannot afford to buy phosphate, among the other means necessary for production. Thus, agricultural production is concentrated in a few rich countries and in a few developing countries, thanks to rich farmers and agricultural companies. Most farmers are poor, do not have access to fertilisers, including phosphates, and do not produce enough even to support their families.

Regarding the absorption and use of P from the soil by plants, there are at least two major obstacles. Firstly, even in soils highly deficient in bioavailable P there is a considerable amount of total P (Rheinheimer et al. 2019). However, years and decades of research related to the dynamics of P in the soil, including all knowledge related to physicochemical and microbiological mechanisms, has not translated into technologies that are truly usable by farmers. Information is often too generic, such as: "mycorrhizae are essential", "there are plants with a greater ability to absorb and use P", and "there are solubilising bacteria", etc. Some biological products have even appeared on the market that are more magical than phenomenological. The "Phosphorus Legacy" is quantifiable (Gatiboni and Condron 2021; Pavinato et al. 2020), but there are no technological means available to farmers to "mind" this stock of P via commercial crops. Even with advances in agroecology, and all the implications for land use and management, it is still not possible to offer viable and quantifiable alternatives for using this P stock.

The second obstacle, and obviously closely related to the first one, is the inappropriate use of phosphate fertilisers. Farmers who access and consume most of these fertilisers are the same ones who produce major commodities (soybeans, corn, wheat, cotton, coffee, orange juice, sugarcane), are the owners of huge areas of land, and are those that use any and all "modern" technology (total mechanisation of activities, GMO plants, and pesticides). The use of the NT system is representative of this type of farmer, especially in Brazil and Argentina. Phosphate fertilisers are just an input like any other

and, for this type of farmer, the cost of P is very low compared to other production costs. The effectiveness of fertiliser application is prioritised for its agro-environmental efficiency, which can lead to overuse and eutrophication of surface water sources (Bender et al. 2018). But this overuse is currently only a major concern for scientists and urban populations, and not for fertiliser users. At the same time as overuse of P is a problem in some sectors, in other sectors problems arise from not correcting the severe lack of P in the soil profile when natural biomes (Cerrado, Tropical Forests and Campos Sulinos – Pampa) are destroyed and the high toxicity of Al leads to limitations in deep root growth. This severely decreases the efficiency with which added phosphate is utilised. The scientifically correct approach would be to raise the available P content above the critical level in at least the 0-20 cm layer, and then reapply the P exported by crops in the row and at the time of sowing. Unfortunately, this proven approach is not adopted by all users of phosphate reserves.

Potassium

Due to its chemical properties, K is not part of the structure of any organic molecule, and therefore only exists in the soil in the following inorganic forms: (a) at low concentrations dissolved in the soil solution in the form of hydrated K^+ ions; (b) as part of primary minerals resistant to weathering (structural K); and (c) adsorbed to functional groups in surface complexes with varying degrees of energy. It is common in the literature to find the terms “exchangeable K” and “non-exchangeable K”, but this terminology has no relationship to the type of complex formed between the K^+ ion with the functional group, and only makes sense when properly associated with laboratory conditions used in the method (solution concentration, stirring time, soil:extractor ratio, number of extractions, among others) (Bourder et al. 2021). Plants absorb K from the soil solution, and as its content decreases, it triggers a rapid and stoichiometric release of the K retained with lower energy electrostatically adsorbed to functional groups, and more slowly of the K retained with greater energy in the interlayer of 2:1 clay minerals, or by hydrolysis of structural forms of the solid phase in soils with primary minerals containing K.

The total K content in soils varies greatly depending on the source material and weathering. More weathered soils derived from basalt can have a total K content of up to 3,000 mg kg⁻¹ (Almeida et al. 2021), while younger soils derived from granite can have up to 30,000 mg kg⁻¹ (Flores et al. 2021). Even with high total K contents, only a small fraction is available to plants and the bioavailable fraction is commonly estimated by routine soil analysis

methods. These generally use cation exchange resin, ammonium chloride, ammonium acetate, or the Mehlich-1 solution as extractors. The K extracted with these methods is called “exchangeable K”, which accounts for the soluble K and part of the K adsorbed to soil functional groups with lower energy (Bourder et al. 2021). However, these methods may not show good correlation with plant uptake due to the contribution of non-exchangeable forms of K from the soil that are not accessed by these extractors. In fact, it is known that there is a considerable contribution of non-exchangeable K to plant nutrition, which is attributed to the release of K from 2:1 clay minerals and the dissolution of K-containing minerals (feldspar and mica, for example) present in coarse soil fractions (silt and sand) (Moterle et al. 2019). However, the proper estimation of the contribution of these forms still represents a challenge, although the importance of the contribution of non-exchangeable forms of K to plant nutrition has been recognised for over 40 years in Brazil (Mielniczuk 1977; Mielniczuk and Selbach 1978) and Peru (Oelsligle et al. 1975).

In Southern Brazil, it is estimated that the non-exchangeable K contribution to the K nutrition of plants can range from 50 to 73% without K fertiliser addition, and from 1 to 18%, with K fertiliser addition (Steiner and Lana 2018). In Uruguay, the non-exchangeable K contribution to K nutrition can be up to 43% (Núñez and Morón 2017). In Argentinean soils, the initial exchangeable K and illite concentrations accounted for 90% of the variations in the K taken up by plants, demonstrating that mineralogy can help to better estimate soil capacity to supply K (Zubillaga and Conti 1996).

In highly weathered oxidic soils (Oxisols and Ultisols), with no or low 2:1 clay minerals, the supply of K to plants is primarily derived from the release of exchangeable K adsorbed to the surface functional groups of mineral clays and organic material. Under these conditions, the available K content in the soil estimated by routine methods (Mehlich-1, ammonium chloride, or cation exchange resin) represents almost the totality of the K absorbed by the plants. However, in soils with 2:1 clay minerals, the K dynamics are more complex, as these mineral structures can act both as a drain and a source of K (Flores et al. 2021).

Although traditional soil fertility assumes that the tropical soils found in South America have only secondary minerals, such as kaolinite and Fe- and Al-oxides, there are several scientific reports that show that the mineralogy of these soils is not so uniform (Almeida et al. 2021; Firmano et al. 2020; Schaefer et al. 2008), and this has greatly limited the understanding of South American soil chemistry. Although in many cases 2:1 clay minerals are present in much lower concentrations than oxides and kaolinite, they still

have a marked influence on the chemical properties of the soil due to their high reactivity, and directly affect K dynamics (Britzke et al. 2012). When the K supplied via fertilisers is lower than that taken up by plants, K retained in the siloxane cavities of clay minerals such as mica may be released. When added K is greater than absorbed K, 2:1 clay minerals such as vermiculite can retain excess K in their interlayers (Moterle et al. 2019), which can decrease the availability of K for plants due to competition between the soil and the plant for the added K. In both cases, there may be changes in the mineralogy of the clay fraction due to the entry or exit of K from the 2:1 clay mineral interlayers. For this reason, it is common to observe situations in soils with the presence of these clay minerals where crops do not respond to the addition of fertiliser, or the available K content does not increase even with large additions of K via fertilisers (Firmanno et al. 2020). In these soils, adequate K fertilisation should focus on the repositioning of exported K to prevent temporary adsorption, rather than being based on soil testing and raising the soil available K up to a critical content. Moreover, soil K management guidelines should consider both topsoil and subsoil nutrient status and variables related to soil K buffer capacity. The K supply capacity in the 0-20 cm layer has been estimated to average 48% in production systems in Argentine soils (Correndo et al. 2021), and about 60% for the 0-30 cm layer in soybean production systems in Brazil (Oliveira et al. 2021).

Sulfur

Sulfur (S) is a macronutrient nutrient absorbed in large amounts by crops. Historically, S deficiency is rarely described in the literature, and isolated cases where it has been detected are mainly sandy soils with low organic matter content (Pias et al. 2019). Because of this, S is the least studied macronutrient in the world and in South America. However, with the advance of agriculture in recent decades, the occurrence of S deficiency has become more common. There are several factors that have contributed to this, including the: (i) increased productive potential of crops, (ii) use of concentrated NPK fertilisers, (iii) reduction in the atmospheric deposition of S, (iv) reduction in the use of S-based agricultural pesticides, (v) reduction in SOM contents, and (vi) occurrence of chemical and/or physical barriers to the deepening of plant root systems.

There is still no consensus in Brazil regarding the critical levels of soil S and the diagnostic layer that should be used. Thus, differing criteria are used by official agencies for S recommendations in different regions. In the states of Rio Grande do Sul and Santa Catarina, the critical content of available S varies from 5-10 mg dm⁻³ in the topsoil (0-10 cm) depending on

whether the crop has a high or low demand for S (CQFS-RS/SC 2016). In São Paulo State, the critical levels are the same, but the diagnostic layer is 0-20 cm (Raij et al. 1997). In the state of Paraná, the application of S is recommended when the available S content in the 0-20 and/or 20-40 cm layer are below the critical levels of 3 and 9 mg dm⁻³, respectively (Pauletti and Motta 2017).

The content of S available in the soil, as discussed above, is dependent on several factors and has a large temporal variability. This makes it difficult to accurately determine fertiliser requirements. For this reason, studies evaluating isolated cases have not found a good relationship between the S content available in the soil and the grain yield of crops (Osório Filho et al. 2007). On the other hand, a joint analysis carried out in a recent systematic review for Brazil, found that the available S content was the main factor influencing crop response to S fertilisation in soil under NT (Pias et al. 2019). In this study, no crop responded positively to the addition of S when the soil available S was greater than 7.5 mg dm⁻³ in the 0-20 cm layer. However, in soils with available S content lower than 7.5 mg dm⁻³, there was a positive response in 50% of crops. It was also shown that 63% of the crop response to S (positive or no response) could be explained by available S content in the soil in the 0-20 cm layer, climatic zone, crop species, and productivity expectation.

Sulfur deficiency is more common in highly weathered tropical soils, which naturally tend to have low organic matter content (high mineralisation rate), especially when poorly managed, thus resulting in soils with low fertility. Studies with S in tropical NT soils (Cerrado region) found a positive response in 48% of crops, while in subtropical soils (Southern Brazil) the responses occurred in only 18% of crops. However, if we consider only soils deficient in S, in a tropical environment there was a response in 92% of crops, while in a subtropical environment only 22% of crops responded to S. Thus, it should be considered that in subtropical environments, even in soils deficient in S, the probability of a positive response by applying S is lower. Considering only grain crops in NT system in Brazil, Pias et al. (2019) determined critical levels of available S of 7.5 mg dm⁻³ for the surface layer (0-20 cm, 48 crops) and 8.5 mg dm⁻³ for the subsurface layer (20-40 cm, 34 crops). The evaluation of S contents in the 20-40 cm layer can support decision making regarding S management, as soils usually accumulate higher S contents in these layers.

Conclusions

One of the biggest challenges and limitations of crop productivity in South America is to increase the quantity of organic residues added by plant biomass to the soil and to reduce its rate of decomposition. There is no doubt that we should have plants growing and photosynthesising for as long as biologically possible during the year. In many places in South America, this means plants growing 365 days a year. Even so, unfortunately there are still vast agricultural areas that remain fallow for several months during the year, without the use of cover crops, and often without using conservation systems such as NT. The combined use of NT and cover crops should be mandatory, as it brings benefits both inside and outside the farm. For the farmer, this means less dependence on N fertilisers, greater recycling of nutrients such as P, K, and S, greater cation exchange capacity, less soil and nutrient loss, greater soil water retention and, consequently, greater productivity. Outside the farm, it benefits society due to greater CO₂ sequestration and climate regulation, greater regulation of the hydrological cycle, and greater food production.

However, to maximise plant growth and biomass production, it is necessary to provide conditions so that plants can have adequate development. Undoubtedly, the major limitations for this are soil acidity and low nutrient availability, especially P. Although not relevant in some specific areas in South America, soil acidity in the surface and subsurface remains one of the main obstacles in tropical agriculture. While the need for sulfate and phosphate fertilisation can be predicted by properly evaluating the levels of S and P available in the soil, thus increasing the efficiency of fertiliser use, there is still a very large limitation to using this approach for K. To enhance production of plants and plant biomass, and consequently to increase the organic matter content of the soil and the sequestration of C from the atmosphere, it is necessary to encourage and promote the monitoring of soil fertility, not only horizontally, but also vertically. Associated with this, it is necessary to establish research networks aimed at improving the establishment of critical levels of soil acidity and nutrient availability to guide decision-making more assertively, thus maximising productivity and promoting more sustainable production.

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CHAPTER 21

AUSTRALIA

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Abstract

Most of Australia's grain production occurs under rainfed conditions and crops often experience drought stress, making the efficient use of soil water critical for productivity. The highly weathered nature of many Australian cropping soils is confounded by a range of common physicochemical constraints, both inherent and induced, which limit crop root growth and the utilisation of soil water and nutrients. Sodicity is widespread, impacting more than 60% of cropping soils, and while surface applications of gypsum or lime predominantly benefit surface soil conditions, they have little impact on deeper subsoils. Research into the deep placement of organic, nutrient, and mineral amendments into sodic and poorly structured subsoil is currently being undertaken to provide financially viable management options for farmers. Subsoil acidity is also common, and surface applied liming amendments are taking decades to increase the pH below the topsoil. Incorporation of lime using once-off strategic deep tillage is being undertaken on suitable soils and direct deep placement of lime continues to be assessed. High soil strength, both naturally occurring and machinery induced, is common. Adoption of controlled traffic systems is ongoing and in sandier-textured soils deep ripping is regularly implemented. Topsoil water repellence is a problem in sandy soils of the Western and Southern zones and managed using a range of strategies, including soil wetting agents, strategic deep tillage, and the addition of clay-rich subsoils. Australia's grain growers have adapted to many of the other constraints typically present by embracing new technologies, varieties, and practices.

This enables them to remain competitive despite the challenges that arise from an increasingly variable climate and distortions in overseas markets. There has been widespread adoption of reduced tillage and stubble retention systems that improve water use efficiency and reduce erosion risk; the use of tolerant crops or alternative species on saline areas; and the use of soil and tissue testing for nutrients to guide fertiliser management. Methods to improve the efficiency of nutrient use, such as timing, placement, and variable rate application are subject to ongoing research. Growers are increasingly seeing the benefits of investments to overcome soil constraints and further improve water and nutrient use efficiency. Research to find solutions for some of the more intractable subsoil constraints is ongoing.

Keywords: Sodicity, acidity, compaction, soil fertility, water repellence, erosion.

Introduction

Australia's crop production is dominated by grains cropping and occurs primarily under rainfed conditions. Most grain (>70%) is exported and there are few subsidies (Angus and Grace 2017), which means growers readily adopt new technologies to remain competitive. Much of the grain growing region experiences a semi-arid climate, with rainfall typically averaging between 300 and 800 mm yr⁻¹ (decreasing away from the coast) and characterised by a moderate to high degree of variability. Both drought stress and intense storm events are common and increasing under climate change. Summers are warm to hot and characterised by high rates of evapotranspiration, while winters are mild and cool.

The grains growing region can be divided broadly into three major agroecological zones (Figure 21.1) (Williams, Hook, and Hamblin 2002). These are characterised by differing climates and soil types, which influence the type of cropping that occurs and the factors that constrain crop production. Specifically, the:

Western region – has a Mediterranean climate with winter dominant rainfall and hot dry summers. Texture contrast soils, typically with a sandy A horizon, along with deep sands and earths are the most common soil types. Winter cropping of wheat, barley, canola, oats, and lupins dominates production.

Southern region - has a predominantly Mediterranean-type semi-arid climate with winter dominant rainfall, cool winters, and hot dry summers. Some areas (southern Victoria, SE South Australia, and Tasmania) have high moisture availability in the winter and spring, but moderate availability

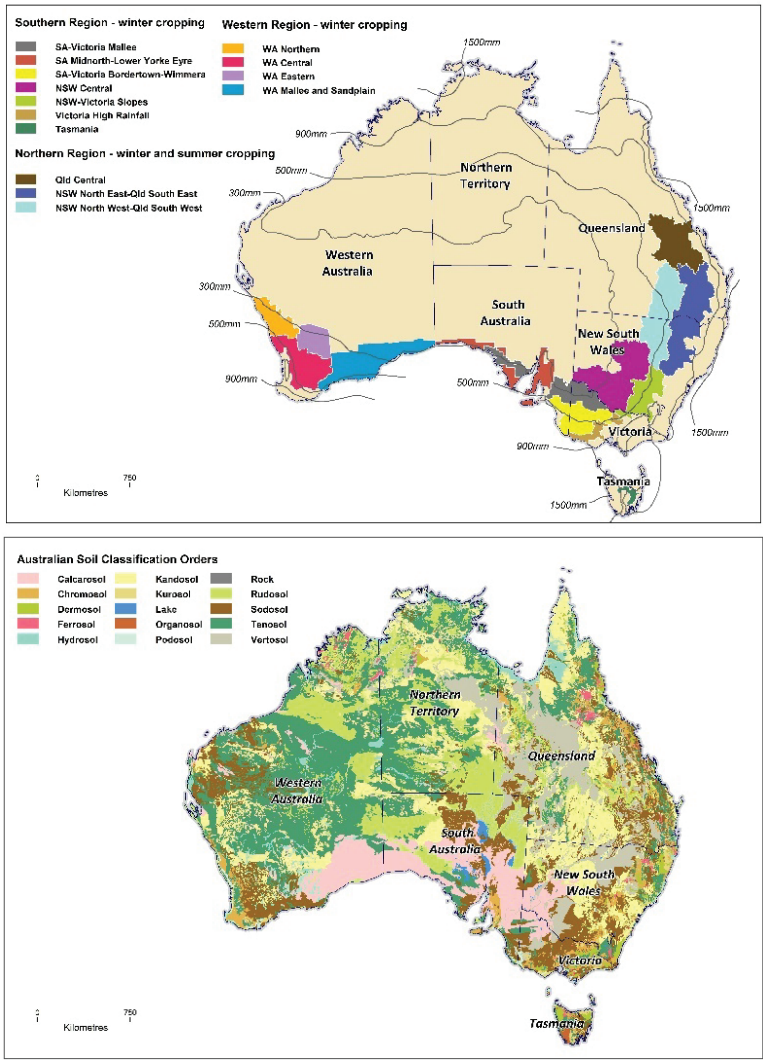


Figure 21.1 National maps showing (top) agro-ecological zones, cropping system types, and climate (modified from Australian Grains Export Innovation Centre 2020 <https://www.aegic.org.au>, accessed June 2021); and (bottom) major soil group areas (Australian Soil Classification; Isbell 2002) and distribution across agricultural zones (modified from Australian Soil Resource Information System 2013 www.asris.csiro.au, accessed June 2021).

in summer. Cropping is conducted on a diverse range of soils (Figure 21.1b), with most containing a neutral to alkaline clay subsoil. Crop production is dominated by winter wheat, with legumes (lentil, chickpea, field pea, and faba beans), canola, and ley pastures (sub-clover) included in various areas across the region.

Northern region – has an increasingly summer dominated rainfall, which allows for either winter or summer cropping. Winter cropping is dominated by wheat and barley, with legumes (chickpea, faba beans, lupins, field pea), canola, safflower, and linseed used in rotation. Summer cropping is dominated by sorghum, but can also include sunflower, maize, mungbeans, soybeans, cotton, and peanuts. Cropping is predominantly conducted on high clay soils (Vertosols) with high inherent fertility and water holding capacity, although smaller areas of lighter textured soils are also used.

The dominance of rainfed production across Australia and the predominantly semi-arid and variable rainfall conditions, means that good crop growth depends on maximising the infiltration and storage of rainfall in the soil (both in crop and during the fallow), and the extraction of this water by the crop. In the Australian context there are several soil characteristics that limit these processes and impact crop production. Many of Australia's soils tend to be deeply weathered with low fertility, contain high levels of accumulated salt. They are also fragile, making them prone to erosion and structural decline if mismanaged (McKenzie et al. 2004; Pain et al. 2012). Texture contrast profiles are common, as are both strongly acid and strongly alkaline soils, which impact nutrient availability, toxicities, root growth, and biological activity (McKenzie et al. 2004). The major soil constraints limiting production in Australia and their appropriate management are discussed in detail below.

Constraints in the cropping soils of Australia

Sodicity

Soil sodicity is widespread in Australia affecting >60% of cropping soils (Rengasamy 2002), representing one of the highest concentrations globally (Page et al. 2020). Of the major physicochemical constraints, sodicity is the largest cause of yield gaps across most dryland cropping systems in Australia, and worth AU \$1.3 billion per annum in lost wheat production alone (Orton et al. 2018). The dominance of sodicity reflects both the relatively low rainfall and high rates of evaporation, which result in impaired leaching and salt accumulation in the profile, especially for clay subsoils (Rengasamy 2002).

In Australia, soils are classified as sodic if the exchangeable sodium (Na) percentage (%ESP) is >6 (Orton et al. 2018). More recently there has been a growing realisation of the need to focus on “dispersive” behaviour rather than just Na concentration in recognition of the influence factors such as magnesium (Mg) and salinity have on soil behaviour, agricultural productivity, and likely responses to amendments (e.g., Bennet, Marchuk, and Marchuk 2016). Sodic/dispersive soils can occur both in the absence and presence of salinity ($EC_e >4 \text{ dS m}^{-1}$), acidity ($pH <6$), and high alkalinity ($pH >8.2$) and in surface soils, subsoils, or both (Page et al. 2020). More than 85% of dispersive soils in Australia are alkaline (Rengasamy and Olsson 1991).

The mechanisms by which soil sodicity/dispersion can reduce crop production are well established (Rengasamy and Olsson 1991). Page et al. (2020) has summarised these as (i) physical mechanisms, including the formation of surface crusts, high soil strength, and reduced hydraulic conductivity leading to perched water-tables and anoxia; (ii) chemical mechanisms, including a range of toxicities (e.g., Na) and increased osmotic potential; and (iii) impacts on soil biological activity. The net effect of these mechanisms is to reduce crop utilisation of soil water and nutrients, which is highly problematic given that both water and nutrients are the major constraint to productivity in Australian cropping systems (Connor 2004).

Salinity

In Australian cropping soils, high concentrations of salt within the root zone may be an inherent soil characteristic (primary salinisation) or have been brought to the surface by rising water tables following the clearing of native vegetation and increased groundwater recharge (secondary salinisation). Soils affected by primary salinisation develop in drier regions where natural weathering/depositional processes combined with limited leaching below the root zone (due to low rainfall / high evapotranspiration and/or low soil hydraulic conductivity) have led to a build-up of salt over time (Dang et al. 2006; Rengasamy 2002). Following clearing of these areas for cropping, decreased transpiration rates can lead to increased profile leaching and move salt below plant rooting zones (Dang et al. 2006). However, in some soils, the presence of dense clay sodic layers may impede water and salt movement. In these soils, salt can remain in the root zone after clearing, and crop growth can be limited due to temporary waterlogging during wet periods and salinity during dry (Rengasamy 2002).

Estimates of the areas affected by salinisation across Australia vary. In the past, estimates have focused on salinisation due to rising water tables,

however, more recent analyses have suggested that primary salinisation is likely to affect production on a much larger proportion of Australia's cropping area (67%) compared to secondary salinisation (16%) (Rengasamy 2002). Recent modelling suggests that over 9 million hectares (24%) of Australian cropping soils have sufficiently high salt concentrations within the top 2 m of the profile (primary and secondary) to limit yield. Southern Queensland and WA are the states most significantly impacted (46 and 32% of cropping soils affected, respectively, compared to 15% SA/NSW and 3% Victoria) (Orton et al. 2018). This salinity is estimated to be costing AU \$182 million annually in lost wheat production alone (Orton et al. 2018).

Soil fertility

The generally ancient, highly weathered nature of Australian soils has resulted in widespread nutrient deficiencies, particularly of phosphorus (P) (Wild 1958), making agricultural production highly dependent on fertilisers (McLaughlin et al. 2011). Some soils, such as the Vertosols that dominate cropping in the northern region, were initially of high fertility, but continuous cropping with little use of legumes or fertiliser has resulted in widespread declines in organic matter and soil fertility, especially of nitrogen (N) (Dalal and Mayer 1986). Consequently, the productivity of cropping systems in Australia since the 1840's has closely followed trends in soil nutrient supply and the development of new strategies viz. the use of superphosphate in the early 1900's to overcome widespread P deficiency, the development of ley-legume pasture rotations to supply symbiotically fixed N to subsequent cereal crops in the middle of the 19th century, to the large increases in the use of N fertilisers allowing intensification of continuous cropping systems based on cereals and canola since 2000 (Connor 2004).

Nitrogen and P are the dominant fertilisers required (Angus et al. 2019). Sulfur (S) and potassium (K) have become increasingly deficient in some soil types and high rainfall environments (Bell et al. 2012; Norton 2017), which may reflect the dominant use of high analysis (ammonium based) fertilisers in recent decades. Trace elements such as zinc (Zn) and copper (Cu) are also naturally deficient in some soil types, most notably highly leached sands and those with high pH, but deficiencies have also developed on intensively cropped soils where only macronutrients have been applied (McKenzie et al. 2004 pp. 20). Traditionally most fertiliser strategies to manage nutrient deficiencies in Australia have focused on the topsoil, which has resulted in the depletion of immobile nutrients, such as P, in the subsoil (Wang et al. 2007). In contrast to other international cropping systems, there is little use of animal manures as a primary source of nutrition in Australia.

Acidity

Many Australian soils have inherently low soil pH, which when combined with further acidification arising from agricultural production, can result in surface soil pH's (measured in 0.01M CaCl₂) less than 5.5 and subsurface pH's less than 4.5 (Li et al. 2019). It is estimated that between 11 and 21 million hectares have strongly acidic topsoil, while 23 million hectares have moderate to extreme subsurface acidity (Kanowski and McKenzie 2011). Ongoing acidification due to agricultural production, coupled with insufficient lime application for maintenance and correction of soil pH, means that the cropping area impacted by acidity is increasing. Agriculture enhances acidification through a range of processes including the use of ammonium-based fertilisers, conversion of ammonium to nitrate, nitrate leaching, and product removal.

Soil acidity reduces crop productivity due to:

1. Release of phytotoxic concentrations of aluminium and manganese into the soil solution (Li et al. 2019; Azam and Gazey 2020).
2. Reduced availability of essential plant nutrients, in particular P, calcium (Ca), Mg and molybdenum, and a reduction in nutrient cycling (Condon, Burns and Li 2021).
3. Reduced nodulation and N fixation in legumes (Hackney et al. 2019; Yates et al. 2021).

High soil strength

High soil strength is one of the most common constraints in Australia's agricultural soils (Hamza and Anderson 2005; Unkovich et al. 2020). In the Western region, 13.2 million hectares, or 70% of agricultural soils, are considered susceptible to subsurface compaction (van Gool 2016). Both inherent (Rengasamy 2002) and induced processes (Hamza and Anderson 2005) can contribute to these hard layers and soil horizons, which reduce crop root and shoot growth, soil porosity, aeration, water infiltration and biological activity (Hamza and Anderson 2005; Chen et al. 2014). Deeper subsoil compaction is induced from the traffic of heavy agricultural machinery (Keller et al. 2019; Antille et al. 2019). The depth and severity of compaction is directly related to axle load and contact pressures and the contact area of tyres and tracks has increased as machinery sizes and axle loads have increased (Keller et al. 2019; Antille et al. 2019). Repeated passes of agricultural machinery, soil moisture content, and soil type can impact the extent of compaction (Chan et al. 2006). In addition to

compaction, soils can be poorly structured and dense, often as a result of poor structural stability, hard-setting, and cementation (MacEwan et al. 2010; Unkovich et al. 2020; Sale et al. 2021).

Contributing to the severe impact of compaction and hard layers is the fact that many Australian soils lack the shrink-swell characteristics that can result in some “self-repair” of soil structure, particularly in southern and western Australia. Creation of structure by soil macro-fauna can also be relatively low in our moisture limited, rainfed cropping systems (Hamza and Anderson 2005), though this is of value where it does occur (Evans et al. 2011). Low soil organic matter also contributes to a lack of structural stability and resilience in response to compaction and cementation (Hamza and Anderson 2005).

Topsoil water repellence

Topsoil water repellence occurs where hydrophobic organic compounds and residues, including waxes and particulate organic matter, coat the surfaces of the topsoil particles and this repellent layer then resists and prevents rapid and even infiltration of water. These repellent compounds generally arise from plants and microbial sources. As coarse-textured sandy topsoils have relatively low surface area, they are more prone to developing and expressing topsoil water repellence, though some higher clay-content soils with high organic matter can also be repellent. The largest areas of agricultural soils at moderate to high risk of topsoil water repellence occur in the Western region with 9.9 million hectares (van Gool 2016), with a further 2.2 million hectares in the Southern cropping region (Unkovich et al. 2020).

Soil water repellence is a dynamic constraint. Its severity and expression vary in space and time, primarily mediated by climatic and seasonal conditions. In wetter periods the severity of repellence can reduce but may increase with warmer and drier conditions over summer (Roper 2005). Within dryland cropping systems several factors have increased the expression of topsoil water repellence, including:

1. Minimum and no-till seeding practices with stubble retention that concentrate organic matter, including those compounds that cause repellence, in surface layers (Roper et al. 2015).
2. A reduction in the number, frequency, and scale of break-of-season Autumn rainfall events (Scanlon and Doncon 2020).
3. Increases in the proportion of crops being dry sown (Fletcher, Lawes, and Weeks 2016), which increases the expression of water repellence

(Roper et al. 2015).

4. The common use of narrow knife point seeding systems, many of which allow dry repellent topsoil to flow into the base of the furrow where the seed is sown (Desbiolles et al. 2019).

Water repellence can result in slow and uneven water infiltration, which can result in poor and staggered germination, preferential and overland water flows, increased risk of wind and water erosion, and poor crop yields (Blackwell 2000; Roper et al. 2015). Weed control is compromised due to poor crop competition and staggered weed emergence, which can reduce herbicide effectiveness (Roper et al. 2015).

Erosion

Both wind and water erosion pose a significant risk to cropping across Australia (Metcalf and Bui 2016). Wind erosion predominantly affects lighter textured soils, and is particularly prevalent in WA and SA, while water erosion is prevalent across the country, but particularly in higher rainfall zones, on sloping land, and where soils are dispersive (Loughran et al. 2004). Erosion risk tends to be seasonal, for example, in the Northern region water erosion is most likely to occur during intense summer rainfall events (Freebairn and Silburn 2004), while in the Western region, wind erosion occurs when strong winds combine with low levels of soil cover during dry summer seasons (Carter and Laycock 2013).

Erosion has been decreasing since the early 1990s, in part due to the widespread uptake of conservation tillage practices, and the recognition among farmers that maintenance of soil cover is essential for soil retention (Metcalf and Bui 2016). However, the increases in climate variability, decreases in rainfall, and increases in extreme weather events that are predicted due to climate change are expected to increase erosion risk and the maintenance of good management will be essential (Carter and Laycock 2013; Carey, Leach, and Venz 2004).

Managing soil constraints in Australian cropping systems

Four basic strategies are used for managing soil constraints in Australian cropping systems: (i) tolerance (genetic solutions), (ii) amelioration, (iii) avoidance, and (iv) “living with the problem”. The choice of strategy will vary with the constraint being targeted and the production system (production potential). Australian cropping systems are relatively low input by global standards, and a combination of high seasonal risk and lack of

financial subsidies makes financial risk a key management consideration.

Sodicity/dispersion

Sodic/dispersive soils are closely associated with a range of other constraints including salinity and acidic/alkaline pH (Rengasamy 2010; Page et al. 2020) and it is critical that all these factors, including where they occur in the profile (topsoil, subsoil, or both), are considered when selecting management options.

Although a variety of strategies are available (see Page et al. 2020), the management of sodicity/ dispersion in Australian cropping systems has centred on ameliorating with Ca, using topdressing of gypsum in alkaline soils and lime in neutral/acid soils. While this strategy can successfully improve crop establishment through improved topsoil structure (Dang et al. 2010), productivity improvements can be unreliable. This unpredictability of response reflects several factors. Most Australian cropping systems experience low rainfall (by global standards). This limits both the dissolution and subsequent downward movement into the subsoil of gypsum/lime applied to the topsoil (Dang et al. 2010), where sodicity tends to be most severe (Rengasamy and Olsson 1991). Often the rates of Ca applied are insufficient to remove Na from the exchange complex throughout the profile, reflecting the influence of financial and seasonal risk on Australian farmers' management decisions. Until recently, analysis of whether a soil will respond to Ca application had been based on %ESP rather than dispersion (Bennet, Marchuk, and Marchuk 2016), thus not accounting for the range of factors that can influence flocculation and dispersion, such as the co-constraint of salinity (Rengasamy et al. 2003). Despite research into alternative ameliorants to Ca, including biochar and acidifying the rhizosphere using microbial mediated acidification of elemental S (Page et al. 2020), these approaches have not generally translated to field studies successfully.

There has been considerable interest recently in injecting high rates ($>15 \text{ t ha}^{-1}$) of nutrient enriched organic matter (referred to as subsoil manuring) into dense, often dispersive clay soils in the high rainfall areas of southern Australia. Although large ($>50\%$) yield responses have been recorded under experimental conditions in the field (e.g., Sale et al. 2019), yield responses to subsoil manuring can be unpredictable. Subsoil manuring appears to improve yields through a combination of short-term increases in nutrition and longer-term enhancement of soil structure (Celestina et al. 2019) and current research is focusing on the role of soil water availability on responsiveness. There are also logistical constraints associated with

accessing enough ameliorant and physically placing it into clay subsoils on a commercial scale (Sale et al. 2021).

A major challenge in developing improved tolerance through breeding has been the need to pyramid tolerance to the range of factors associated with sodic/dispersive soils, including water logging, high soil strength, and Na toxicity. There has been success, however, with identifying improved tolerance to some aspects, such as sealing of soil surfaces (Anzootman et al. 2018) and tissue Na toxicity and osmotic adjustment (Genc et al. 2019).

Acidity

The primary methods for managing soil acidity in Australia is the use of acid tolerant species/varieties or the addition of neutralising amendments. In addition, practices such as the use of less acidifying fertilisers, split N-applications, and overcoming constraints to allow better root growth and reduce nitrate leaching are also important to reduce the rate of acidification. Acid tolerance is largely seen as an adaptation enabling crop production on acid soils while neutralising amendments are applied to correct acidity (Tang and Gazey 2003; Ryan 2018). Acid tolerant germplasm has been shown to increase yields by up to 24-60% in wheat (Tang et al. 2001; Tang and Gazey 2003) and 15-60% in barley (Paynter 2016). Genetic potential to improve the acid tolerance of canola and grain legumes exists, but its value under field conditions is not known (Ryan 2018). Selection of acid tolerant strains of *Rhizobia* can sustain N-fixation in acid soils, further improving the adaptability of legumes (Yates et al. 2021).

Neutralising amendments can be organic or inorganic, but given the volumes required, product availability, and the cost associated with correcting acidity, agricultural limes are the most widely applied (Kanowski and McKenzie 2011). While application rates have increased, they broadly remain below what is required to prevent and correct acidity (Kanowski and McKenzie 2011). Liming products have typically been surface applied and the long-term maintenance of topsoil (0-0.1m) pH above 5.5 through regular liming has been shown to also increase or maintain soil pH in the top 0.2-0.3 m over 10-24-years in long-term field experiments (Li et al. 2019; Azam and Gazey 2020). However, increases in subsurface pH from surface applied lime in Australia's dryland farming systems remain slow, increasing at 0.04-0.05 pH units per year in the 0.1-0.2 m layer, or even less in the 0.2-0.3 m layer or in drier regions (Li et al. 2019; Azam and Gazey 2020).

To rapidly treat sub-surface acidity, a one-off (strategic) incorporation or direct deep placement of lime and/or organic amendments into acidic subsurface layers is required (Davies, Gazey, and Gartner 2008; Davies et

al. 2019a; Azam and Gazey 2020; Condon, Burns, and Li 2021). Direct deep placement of lime has been achieved through delivery of lime behind deep ripping (subsoil) tines, with the aim of creating a continuous vertical seam of limed soil through an acidic subsurface layer (Davies, Gazey, and Gartner 2008; Li et al. 2019). Achieving effective and even lime distribution throughout the seam has been a challenge for such approaches and only a limited volume of the soil profile is treated. Where effective, direct deep placement of lime was shown to increase wheat grain yield by 24% (Davies, Gazey, and Gartner 2008).

More thorough incorporation of lime into an acidic profile is possible through once-off strategic deep tillage (Davies et al. 2019a), though the incorporation method used will influence the depth and effectiveness of topsoil mixing (Scanlan and Davies 2019). Such approaches not only modify pH profiles but also alter soil strength, topsoil water repellence, and incorporate topsoil, with associated organic matter and nutrients, which may have both positive and negative consequences (Davies et al. 2019a). Surface application of a combination of organic amendments with lime may increase the movement of alkali into the subsurface soil without the need for deep tillage (Butterly et al. 2021). However, further research needs to elucidate the mechanism of the effect and the longer-term impacts on pH change and stratification over time (Condon, Burns, and Li 2021).

High Soil Strength

Amelioration of soils with layers of high soil strength can involve physical, chemical, and biological interventions. Physical loosening and the creation of structure via deep ripping (subsoiling) is common in many cropping areas (Ellington 1986, Crabtree 1989). Soil type, crop type, soil conditions at the time of ripping, depth of ripping, soil water, soil stability, crop establishment, and seasonal conditions all have an impact on the yield outcome (Ellington 1986, Kirkegaard et al. 2008; Hamza and Anderson 2008; Parker 2017; Davies et al. 2019b). Responses tend to be largest and most consistent on deeper sandier-textured soils where the ripping depth extends below subsurface hardpan (Parker 2017). Traditionally soils were ripped to 0.2-0.4 m depth (Ellington 1986; Crabtree 1989) but with increasing depth of compaction (Keller et al. 2019) ripping to 0.5m or more has increased, typically resulting in higher yields (Parker 2017; Davies et al. 2019a; Davies et al. 2019b). For example, ripping a deep sandy earth to 0.4m increased cereal grain yields 0.14-0.33 t ha⁻¹ (4-15%) while ripping to 0.7m increased yields by 0.54-0.83 t ha⁻¹ (14-39%) over 3-seasons (Davies et al. 2019b).

Topsoil slotting can also be combined with deep ripping. Soil openers are attached to the trailing edge of the ripping tines and operate below the soil surface. This allows topsoil with associated organic matter, nutrients, and surface-applied amendments, such as lime or gypsum, to fall into the open slots, often to depths of 0.35-0.45 m (Parker 2017). However, crop yield benefits have been inconsistent and further refinement is required to maximise topsoil incorporation and minimise compaction and/or smearing on the sides of the slot (Parker 2017; Davies et al. 2019b).

Physical loosening alone may be of limited benefit where soils are dispersive and subject to structural decline. In these cases, the application of gypsum, lime, or organic amendments are required to improve structural stability (Ellington 1986; Hamza and Anderson 2003; Sale et al. 2021) and maintain the benefits of physical loosening (Hamza and Anderson 2003; Chan et al. 2006).

Plant roots can also mediate improvements in soil porosity and structure by creating root channels, releasing root exudates, and/or stimulating soil microbiology (Chan and Heenan 1996). Increased density of root-induced macropores in dense subsoils has been shown to improve crop water extraction and grain yield (Chan and Heenan 1996; Ward, Dunin, and Micin 2002; McCallum et al. 2004).

The amelioration of hard layers is costly and thus prevention of compaction is vital (Chan et al. 2006; Antille et al. 2019). Controlled traffic farming (CTF) systems restrict machinery to defined wheel tracks minimising the area of soil impacted by machinery traffic (Antille et al. 2019). Under CTF systems the area of the paddock trafficked can be as low as 15% compared to 50% of the area trafficked under non-CTF and 80% if conventionally tilled (Antille et al. 2019). Financial benefits can be substantial and may increase profitability across a range of farm-types by 51-67% (Kingwell and Fuchsbichler 2011).

Topsoil water repellence

Seeding systems, such as winged and paired row seeding points, can grade repellent topsoil away from the sown furrow and into the inter-row to improve crop establishment (Desbiolles et al. 2019). Sowing crop rows near, or on, the previous season's crop rows has also been shown to improve cereal crop establishment by 17-50% and increase grain yield 14-36% compared to inter-row sowing (Desbiolles et al. 2019; McDonald et al. 2019). This approach takes advantage of the preferential infiltration of rainfall via the crowns and remnant root systems of the previous season's crop row (Roper et al. 2015; Desbiolles et al. 2019). Utilising this mechanism,

no-till cropping systems with maximum residue retention can enable effective soil wetting even in the presence of water repellent topsoil (Roper et al. 2013).

Infiltration of water into repellent soil can also be facilitated via soil wetting agents, which can have both surfactant and humectant properties. Wetting agents are typically applied as liquids, banded either below the surface near the seed, on top of the sown furrow (usually after the press wheel), or occasionally as targeted blanket applications (Crabtree and Henderson 1999; Desbiolles et al. 2019; McDonald et al. 2019). Coating of wetting agent directly onto crop seed has only been investigated recently (Anderson, Erickson, and Davies 2018). In responsive situations, soil wetting agents can significantly improve the timeliness and number of established crop seedlings (Desbiolles et al. 2019; McDonald et al. 2019).

The application and incorporation of clay-rich subsoil into repellent sandy topsoil is well-established in Australia (Cann 2000). Addition of clay increases the soil surface area, masking the hydrophobic layers present on the sand particles (Ward and Oades 1993) and is known to be long-lasting (Cann 2000; Hall et al. 2010). Repellent sandy topsoil typically has a low clay content, <5%, which may only need to be increased 1-2% to overcome soil water repellence. Clay application is expensive, and clay needs to be sourced close to where it is applied. Subsoil-clays can be a source of K and S, but can also be sodic, highly alkaline, and contain toxic concentrations of boron and salt. Dilution of the clay subsoil in sandy topsoil and leaching reduces the concentration of toxic compounds.

More recently, one-off strategic deep tillage practices have been used to dilute and bury water repellent topsoil at depth while bringing wettable subsoil to the surface (Davies et al. 2019a; Hall et al. 2020). The reduction in surface water repellence is dependent on the extent of topsoil burial. Deep mixing approaches using rotary spaders or offset disc ploughs reduce repellence to low levels, while soil inversion with a mouldboard plough or deep working one-way plough will typically make the surface completely wettable (Roper et al. 2015). These approaches are also an opportunity to incorporate lime, nutrients, and other soil amendments and can aid weed control and reduce frost damage (Davies et al. 2019a; Betti et al. 2019).

Salinity

To ameliorate soil salinity, increased downward flux of water through the profile to leach salts out of the root zone is required (Adcock et al. 2007). However, achieving this in the rainfed, semi-arid production systems dominant throughout Australia is very difficult, if not impossible, particularly

where downward water movement is restricted due to dense subsoil layers. Where secondary salinisation is present, it may be possible lower water tables below plant rooting zones using a combination of artificial drainage and the revegetation of groundwater recharge areas with deep rooted, perennial, and high water use plants. However, the economic and logistical constraints of such approaches are significant (Clarke et al. 2002).

Due to the difficulties in ameliorating salinity, management focus is often on improving productivity or profitability without amelioration. Practices that have been identified as useful include:

- Accurate identification of saline areas so that these can either be excluded from cropping or managed more profitably by matching inputs (e.g., fertiliser, pesticides) to expected yield (Dang et al. 2010).
- Use of salt tolerant species/cultivars that can provide greater economic return on salt affected soils (Dang et al. 2010; Nuttall et al. 2010), although only small improvements have been achieved to date.
- Optimising plant nutrition. For example, the addition of P and Zn can increase grain yield at saline sites (Dang et al. 2010; Singh et al. 2006).

Erosion

Maintenance of soil cover to reduce raindrop impact, slow water and wind movement, and improve soil cohesion and structure is the primary management strategy to prevent erosion (Carter and Laycock 2013; Carey, Leach, and Venz 2004; Freebairn, Loch, and Cogle 1993). As an example of the effectiveness of this approach, Figure 21.2 illustrates the difference between water erosion rates on areas of bare fallow compared to those with greater soil cover. No- or reduced-till with stubble retention, and the effective treatment of constraints to crop growth (e.g., acidity, sodicity, water repellence, infertility, pests, disease etc.) are particularly important to maintain good plant cover and preserve soil structure (Carter and Laycock 2013). Avoidance of practices that decrease cover, such as stubble burning, over grazing of residues, or long fallows, are also essential, and tillage events should be avoided during high risk erosion periods (Carter and Laycock 2013; Carey et al. 2015).

Physical structures such as windbreaks can also be effective to help decrease wind speeds and rates of wind erosion (Carter and Laycock 2013). Similarly, contour banks and grassed waterways are used to slow and direct

runoff to prevent water erosion in sloping areas - although their use has declined in Australia as steeper areas have been retired from cropping and no-till and residue retention have increased (Carey, Leach, and Venz 2004; Freebairn, Loch, and Cogle 1993).

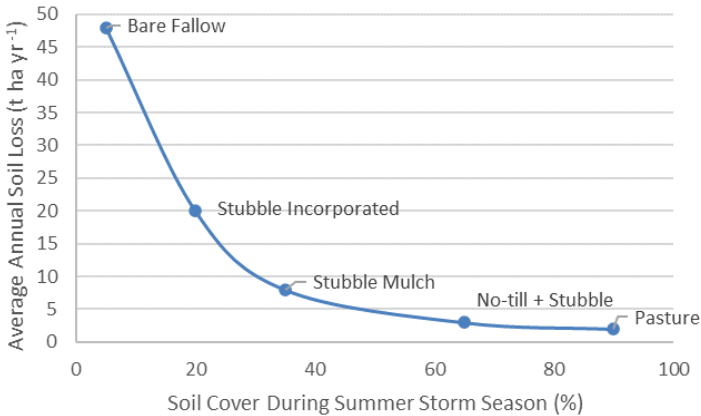


Figure 21.2 The change in average annual soil loss (t ha yr⁻¹) observed under a variety of management practices depending on their level of soil cover [reproduced with data from Freebairn and Silburn (2004)].

Managing soil fertility

There have been major changes in the nutrient management strategies used in Australian cropping systems over the past 30 years (Angus et al. 2019). Previously mixed cereal-livestock (sheep) systems and mechanical cultivation dominated production, whereas now continuous cropping systems based on minimum tillage/stubble retention to improve rainfall use efficiency (Kirkegaard and Hunt 2010) and involving cereals, pulses, and canola are dominant. Nutrient management has consequently changed from having a heavy reliance on soil supply and mineralisation, following in more drought prone environments, and the supply of N by pasture-leys legumes (Connor 2004) to crop demand being met mainly by fertilisers. None-the-less, the average fertiliser application rates to dryland cereals and oilseeds (45 kg N ha⁻¹) is low by international standards, with a tendency to undersupply because of low yield potential and the general lack of subsidies (Angus and Grace 2017).

Soil testing remains the key strategy to determine fertiliser responsiveness and application rates (Conyers et al. 2013). The accuracy of these predictions, however, is heavily influenced by seasonal (rainfall) conditions and the wide diversity of soil types and crops grown. Because fertilisers are the single largest variable costs input in many systems (ABARE 2020), a key aspect of nutrient management has been to balance the economic risk of investing in fertilisers against the production risk associated with seasonal (principally water deficits) conditions (Monjardino, McBeath, Brennan et al. 2013).

Fertiliser use efficiency in Australian cropping systems, whilst commensurate with those globally, is poor. Some studies show that only between 7-22% of P fertiliser is utilised by dryland crops (McBeath et al. 2012) and on average ~40% of applied N is recovered by the crop in the year of application (Angus and Grace 2017) but this can range from 10-70% depending on seasonal conditions (Wallace et al. 2021). There has been considerable research into the development of enhanced efficiency fertilisers to reduce losses of N via denitrification and volatilisation (Chen et al. 2008) and P via chemical fixation/precipitation (Bertrand et al. 2006). However, their relative high cost has precluded commercial use in dryland cropping systems. Recent technological developments in proximal and remote sensing, when combined with variable rate application, have provided the opportunity to improve management, especially of N, although rates of adoption remain low (Colaco and Bramley 2018).

There has been a major shift from the dominance of phosphate fertilisers to the dominance of N fertilisers since the 2000's (Angus et al. 2019), corresponding with cropping intensification. The dominance of surface application of nutrients, especially immobile ones such as P has also seen recent interest in "deep" placement of P and K fertilisers at ~20 cm. Deep placement has produced yield responses of 5-25% compared to "conventional" placement, presumably because the fertiliser is not "stranded" by frequent drying of the topsoil (Angus et al. 2019).

Conclusion

In Australia's rainfed cropping system, soil constraints, which limit crop access to water and nutrients, remain a fundamental barrier to crop productivity and profitability (Rengasamy 2002; Adcock et al. 2007; Dang et al. 2010; MacEwan et al. 2010; Unkovich et al. 2020). Many improvements in productivity have resulted from managing seasonal variability and the adoption of strategies to improve the capture and utilisation of water (Kirkegaard and Hunt 2010).

Changes in climate resulting in less reliable rainfall at critical times during the crop growing season (Scanlon and Doncon 2020) are likely exacerbating the impact of soil constraints (Hochman, Gobbett, and Horan 2017). The key to future productivity improvements, therefore, will be the better management of soil constraints. Australian grain growers have shown a willingness to invest in, and utilise, new technologies and practices that improve the efficiency and profitability of grain production (Hochman, Gobbett, and Horan 2017). As farm size, machinery size, and land values have increased, so has the capacity of growers to invest in practices to manage and overcome soil constraints. Removal of soil constraints allows the value of improved crop genetics, agronomy, farming systems, and crop protection practices to be more fully realised. Ongoing research and development is required to further improve our understanding of how to better integrate genetic tolerance, amelioration, and avoidance strategies to economically manage the range of constraints that currently limit productivity.

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PART F:

PERSPECTIVES

CHAPTER 22

MULTIPLE SOIL CONSTRAINTS IN DRYLAND CROPPING SYSTEMS

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Abstract

Many soils, especially those that dominate semi-arid dryland cropping systems, contain multiple soil physicochemical constraints, making their management much more complex than soils affected by single constraints. These constraints (or limitations) comprise any factor that prevent a plant achieving its maximum growth rate and include limits of soil resources, toxicities, and diseases. These constraints often exhibit high spatial (three dimensional) and temporal variability making diagnosis and prognosis for management even more challenging.

Until the late 20th Century, thinking about managing soil constraints centred on the *Law of the Minimum* and various variations *viz.* that no growth response can occur until the most limiting constraint is corrected. Inherent to this approach was that there were no interactions between constraints. While the *Law of the Minimum* approach may apply in natural, resource poor environments, plants in more productive agricultural systems

were often found to experience multiple constraints with synergistic responses to the addition of multiple resources (*Multiple Limitations Hypothesis*, which was subsequently modified to recognise that plant growth is proportional to the intensity of all stresses - referred to as co-limitation). Several experimental approaches can be used to assess these multiple constraints, including factorial and gradient designs, boundary level functions, and biophysical simulation modelling. Due to the complexity that arises from the potential interactions between multiple soil constraints (as well as environmental factors, such as rainfall), the identification and analysis of potential impacts on plant productivity is often highly complex. A range of approaches to analyse these impacts including mechanistic simulation modelling, multivariate regression modelling, and recent developments in machine learning algorithms are discussed.

Whereas the diagnosis of single soil constraints is well developed, the identification of multiple constraints is less advanced. However, recent developments in rapid soil testing and soil sensing (using both proximal and remote approaches) can be used to assess the impact of multiple soil constraints on crop growth rates, thereby accounting for the inherently high spatial and temporal variability that characterise these constraints at a paddock scale.

Currently most management strategies available to farmers focus on single soil constraints. These can be categorised as (genetic) “tolerance”, “amelioration”, “avoidance” and “live with the problem”. Each approach has its limitations from both a crop productivity, logistical, and economic perspective. We conclude, however, that if crop productivity (and farmer profitability) is to be maximised, multiple constraints will often require a pyramiding of these approaches based on a reliable knowledge of their spatial and temporal variability and account of crop type and climatic and economic risk.

Introduction

In dryland cropping systems, soil physicochemical and nutritional constraints can significantly limit the ability of grain crops to achieve their water limited yield potential (Armstrong et al. 2019; Lobell et al. 2009). These constraints are particularly problematic in semi-arid environments that dominate dryland production in the Middle East, North Asia, and Australasia (Bot Nachtergaele, and Young 2000), where restrictions to root growth and water and nutrient uptake ultimately limit grain yield.

The management of soil constraints remains a major challenge for farmers for several reasons. Firstly, most soils contain multiple constraints

(e.g., Nuttall et al. 2003), thus traditional approaches based on managing a single constraint will most likely produce only marginal yield increases compared to overcoming two or more constraints (Nuttall and Armstrong 2010). Secondly, although novel strategies are available to manage multiple constraints, such as “subsoil manuring”, high upfront costs and logistical challenges make farmers wary of the economic risk (Nicholson 2016), despite the potential long-term benefit demonstrated by economic analysis (Sale and Malcolm 2015). Thirdly, many soil constraints exhibit high spatial and temporal/seasonal variability in their location and severity (Armstrong et al. 2009). This variability may be managed by the farmer if variable rate approaches are adopted, particularly when more expensive amelioration strategies are used. However, successful variable rate management is dependent upon accurate, reliable, and cost-effective diagnosis of which soil constraints are priorities for management.

This review outlines a conceptual framework to understand the nature of multiple soil constraints, including approaches to identification, analysis of their impact on productivity, and strategies for management. Most examples are from dryland grain production in semi-arid areas, although many of the principles apply to other production systems.

The nature of multiple soil constraints

We define a constraint as any factor that produces sufficient stress that it reduces the rate of a physiological process below the maximum rate that plant could otherwise sustain (Lambers, Pons, and Chapin 2008). Plant growth (and yield) will therefore be inversely related to the intensity of stress (Sadras 2005). Soil constraints can be abiotic, biotic, or a combination of both. While some studies use the terms constraint and limitation interchangeably, limitations typically apply to the lack of a resource e.g., water or nutrients, while the broader term constraint encompasses resource limitations, toxicities, and disease.

The categorisation of constraints, e.g., those that limit access to resources, toxicities, and disease, is challenging. Most soil constraints limit root growth and therefore access to the resources of water, oxygen, and nutrients. Letey (1985) lists several soil physical factors that directly limit plant growth via their effect on the root: water potential, rate of gas exchange through “aeration” and supply of O₂, and mechanical resistance. Other constraints limit metabolic function, including some root disease, herbicide residues, and toxicities such as boron (B) toxicity in cereals (Yau and Ryan 2008). These categories are not exclusive, as some can impact production via multiple mechanisms. Some elements fall into both categories depending

on the circumstances, their concentration, and the species. Boron for example can be either deficient or toxic and whereas high B limits root growth and therefore water extraction in lentils (Hobson et al. 2006), high B is toxic to leaf function in cereals (Nable 1988), resulting in reduced transpiration efficiency.

Identifying soil constraints is challenging, particularly where the constraint is indirect, or the effects are separated in space or time; multiple constraints add another dimension of complexity. For example, in highly alkaline soils, crop growth can be limited by bicarbonate (HCO_3) toxicity if soil water is high, but subsequently limited by high soil B and salinity during dry finishes. However, for most dryland agricultural production systems, and particularly those of semi-arid areas, soil water is regarded as the primary constraint to crop yield (Boyer 1982; Letey 1985; Cossani and Sadras 2018).

Soil constraints (singular or multiple) can vary in intensity at temporal and spatial scales. Plant and soil water potential, for example, can be at equilibrium in the pre-dawn, with the plant subsequently being water stressed at midday. This results in reduced stomatal conductance and photosynthetic rate even if the soil contains potentially adequate supplies of plant available water (Lambers, Pons, and Chapin. 2008). However, for annual grain crops, the primary time scale unit will be the growing season as the critical response variable is grain yield. Crop growth may change during the season due to varying physiological demand or ability to acquire resources, and changes in sensitivity to particular constraints. For example, the early vegetative growth stages of cereal crops are more sensitive to plant available phosphorus (P) supply than latter in the growing season (Grant et al. 2001). Constraints affecting crops on the time scale of years may occur when a soil constraint reduces the use of soil water or nitrogen (N) to a sensitive crop, potentially resulting in increased supply to a subsequent, less sensitive crop in the rotation. Plant responses to constraints can also operate across generations via genetically induced adaptation in a natural system, and in crop management through selective breeding for tolerance.

Spatially, constraints can occur at different levels of biological organisation ranging from the cell, plant, community of plants (a crop), to an ecosystem / farming system (Sperfeld, Raubenheimer, and Wacker 2016). At a plant level, morphological plasticity can be expressed as the distribution of assimilate and N during haying-off, which in turn can alter harvest index and grain yield (van Herwaarden et al. 1998); increased root proliferation in zones of enhanced soil N and P availability (Drew 1975); or remobilisation of P from older to younger leaves in plants with inadequate P (Marschner 1986). At a population/crop scale, constraints can range from

the level of adjacent plants, where one plant shades another, to the whole paddock, where all plants are genetically identical but variation in the underlying soil influences the type and severity of constraints. From a farmer's perspective, it is the paddock "population" and grain yield that matter most.

The ability to quantify the level of stress induced by a constraint, and its effect on a plant, is critical for making informed assessments of potential management responses. Because plants can acclimatise physiologically and morphologically to stresses, biomass is regarded as a better indicator of constraint than seed yield (Kho 2000). It is yield however that will most influence farmer's management responses.

Because of the infinite combinations of stresses that can occur in time and space, terms such as "major limitations" and "principal limiting factor" are widely used (Kho 2000). The current theory about the nature of constraints, however, suggests that these terms may not be always appropriate at a biological level. Many approaches to studying constraints have provided a qualitative assessment – generally as a yes or no response. The use of quantitative approaches where both the frequency (e.g., in relation to seasonal rainfall) and degree of constraint is determined not only improve our understanding of the processes (Sadras 2005) but also facilitates decision making.

Approaches to assessing multiple constraints

An understanding of the approaches to assessing the effect of constraints (soil and other factors) on plant growth underpins the choice of strategy used to identify, measure, and manage these constraints. Thinking in this area was dominated by the Sprengel-Liebig "*Law of the Minimum*" until late in the 20th Century (see van der Ploeg, Böhm, and Kirkman 1999). This Law states that plant growth is limited by a single factor at a time and that growth can only be increased when that constraint is removed. Originally focused on nutrient deficiencies, the Law was subsequently extended to include other factors, such as soil condition (e.g., pH, water), by Blackman's (1905) "*Law of Limiting Factors*". The concept was qualitative (binary) as a factor was either limiting or not and a crop could not substitute one limiting resource for another (Sinclair and Park 1993). Subsequent conceptual advances were made by Liebscher (1895), Mitscherlich (1906), and Baule (1918) so that growth was related to the supply of each nutrient (N, P, etc) by a diminishing-returns relationship and growth constraint factors (zero for no growth, one for unconstrained growth) from each nutrient resource could be multiplied together to represent the effect of several constraints on

overall growth.

In recent decades there has been a growing recognition that multiple constraints/stresses can interact both synergistically and antagonistically to co-limit primary production (Sadras 2005; Sperfeld, Raubenheimer, and Wacker 2012). This has resulted in two basic paradigms (i) Sprengel-Liebig Law of the Minimum (LoM), which states that growth is limited by the resource in shortest supply; and (ii) the multiple limitations hypothesis (MLH) where growth can be limited by multiple constraints simultaneously and growth is maximised when all constraints are equally limiting (Sadras 2005; Cossani and Sadras 2018). MLH stresses can be positive or negative (Sadras 2005). These differing concepts are best illustrated (Figure 22.1) for the LoM where there are distinct break points described where the crop response shifts from nutrient A to nutrient B, whereas in the MLH approach the change is gradual. Both paradigms have focused on nutrients but have been extended for other resources such as water (e.g., Cossani and Sadras 2018). Conceptually, both paradigms could also be applied to constraints resulting from toxicities at a range of timescales varying from daily to the growing season (Cossani and Sadras 2018).

The pattern of co-limitation, and the underlying mechanisms operating, vary with the level of biological organisation - cell to plant to community/crop (Sperfeld, Raubenheimer, and Wacker 2016). A major mechanism underpinning the MLH is the ability of plants to reallocate internal resources to the organ that provides the highest return on investment (Bloom, Chapin, and Mooney 1985; Sinclair and Park 1993). For example, when nutrients are constraining growth, greater photosynthate is allocated to roots to acquire more nutrients. Because of this reallocation within the plant, growth is maximised when all resources are equally limiting. Subsequent studies have shown that the LoM generally applies at low levels of resource availability, e.g., natural ecosystems, whereas the MLH frequently occurs at the higher levels of resource availability that characterise most modern agricultural systems (Wallace 1990; de Wit 1992; Sinclair and Park 1993; Kho 2000; Agren, Wetterstedt, and Billberger 2012). An interesting variation of the MLH paradigm is Letey's (1985) *non-limiting water range* concept, where root function is constrained unless a range of soil physical conditions are simultaneously met.

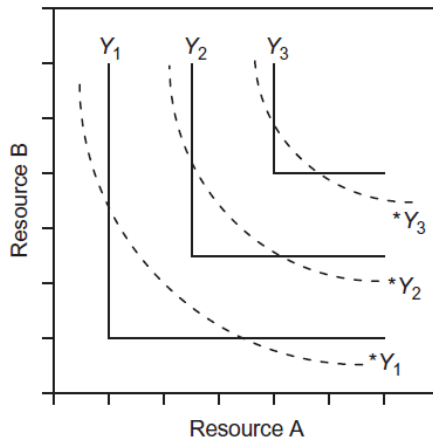


Figure 22.1 Response of crop yield to the availability of two resources, A and B. Isolines Y_1 , Y_2 , and Y_3 represent low, medium, and high yields according to the “Law of the Minimum”, whereas isolines $*Y_1$, $*Y_2$ and $*Y_3$ represent yields according to the multiple limitations hypothesis. Source: Cossani and Sadras (2018).

A range of experimental approaches have been proposed or utilised to both identify the type and severity of constraints (either of soil or other factors) to plant production (Kho 2000; Sperfeld, Raubenheimer, and Wacker 2016; Cossani and Sadras 2018), including:

- Factorial experiments in the field, or a controlled environment, for analysis of either limiting resources e.g., Dunsford, Armstrong, and Mason (2012) or toxicities e.g., Nuttall, Armstrong and Connor (2006). Responses measured are typically dry matter production but can include other measurements, such as resource use efficiency or capture e.g., plant use of soil water. A requirement for this approach is having sufficient levels of resource supply (or toxicity) constraint at or near the asymptote of the response and at suboptimal, co-limiting levels, including accounting for the interactive effects between different constraints (Sperfeld, Raubenheimer, and Wacker 2016).
- Boundary Line Functions: A widely used example of a boundary line function was that of French and Schultz (1984a) who assessed the relationship between grain yield and rainfall to understand what

proportion of cereal crops were achieving water limited yield potential. Yields below the boundary line were attributable to other constraints such as suboptimal nutrition (French and Schultz 1984b). The approach has also been successfully extended to the co-limitation of water and N on crop production (Cossani and Sadras 2018).

- **Simulation models:** Biophysical simulation models commonly used in agriculture, including APSIM (Keating et al. 2003), calculate stress indices ranging from one (no stress) to zero (total limitation) on a daily time-step basis for potential constraints such as water and N. These stress indices can be used to identify the magnitude of various constraints through the growing season (Cossani and Sadras 2018).
- **Stoichiometry and biomarkers:** These are mainly used to investigate nutrient limitations in natural systems, although they can also be used in agriculture e.g., the use of biomarkers such as PreDicta B (Heap and McKay 2009) to predict disease risk in crops.

Common to all approaches to assessing soil constraints is the consideration that the relative impact of the (multiple) soil constraints change during the growing season and depend on plant factors that vary with species and stage of growth. For example, tolerance to waterlogging during germination differs from that during maturity (Setter and Waters 2003), making assessment challenging, especially at a paddock scale.

Interactions between multiple soil constraints

A necessary consideration in the identification and management of multiple constraints is the range of potential interactions, which can be:

- **Additive**, where a statistical analysis shows no significant interaction between the constraints.
- **Synergistic**, where treatments that remove multiple constraints cause a yield increase that *exceeds* that of treatments that remove only a single constraint.
- **Antagonistic**, where treatments that remove multiple constraints cause a yield increase that *is less than* that of treatments that remove only a single constraint.

Examples include:

Rainfall- soil chloride (Cl) interactions. Because grain yields in nearly all dryland cropping systems are limited by lack of water, assessments of multiple soil constraints should consider interactions with soil water. For example, in dryland cropping systems of north-eastern Australia, high subsoil Cl restricts yield by limiting the capacity of crops to extract soil water through osmotic effects (Dang et al. 2010). Across soils with a wide range of subsoil Cl, years of high in-crop rainfall (>150 mm) produced yields 1.6 t ha^{-1} higher than years of low in-crop rainfall (<100 mm), and across all season types, sites with low subsoil Cl ($<400 \text{ mg kg}^{-1}$) yielded 1.6 t ha^{-1} less than sites with high subsoil Cl ($>1600 \text{ mg kg}^{-1}$). There was no interaction between in-crop rainfall and subsoil Cl, and the effects were purely additive.

Boron-salinity interactions. An example of potential interactions between two soil chemical constraints is the synergistic benefit to grain yields of pyramiding both B and salinity tolerance in lentils (Nuttall et al. 2010). Under glasshouse conditions, lentil lines possessing tolerance to both high B and salinity yielded more than a variety with only salinity tolerance on a soil with both of these constraints. These lines, in turn, yielded more than a line with tolerance to neither of these stresses, indicating synergistic interactions between salinity and B.

Sodicity-salinity interactions. Adcock et al. (2007) concluded that where salinity and sodicity occur together, their effect was antagonistic, with growth primarily limited by salinity effects. In these cases, osmotic effects limit yield rather than the sodium adsorption ratio of the soil solution, and thus the *Law of the Minimum* applies. A field study in the cropping zone of north-eastern Australia found no relationship between subsoil exchangeable sodium percentage (ESP) and wheat grain yield despite moderate to high ESP (6-25%) (Dang et al. 2010). This was attributed to the concurrent presence of soluble salts preventing soil dispersion and waterlogged layers, or may reflect an overriding effect of soil salinity *per se*. This study concurs that the relationship between subsoil ESP and Cl follows the *Law of the Minimum*.

Data analysis and prediction of multiple constraints

A key consideration for assessing the impact of multiple constraints on the crop are different data analysis approaches. A wide range of approaches can be used to study the relationships between the environment, management, soil constraints, crop growth, and yield. These can be broadly summarised as mechanistic simulation modelling, statistical (stochastic) procedures, and machine learning.

Mechanistic simulation modelling

In mechanistic simulation modelling, a fundamental understanding of physicochemical and biological processes is represented by algorithms. The effect of different factors on crop yield, or any other variable, can be represented as a factor/variable in these algorithms. A range of biophysical models have been used in grains research, including Decision Support Systems for Agrotechnology Transfer (DSSAT) (Jones et al. 2003) and APSIM (Keating et al. 2003). Simulation models have been successfully used to describe interactions between soil factors (such as texture) and the response of grain yield to fertiliser at both a “point scale” (Asseng, Milroy, and Poole 2008) and spatially within a paddock (Christy et al. 2018). A key consideration when using simulation models is the large input data requirements for calibration and that the underlying mechanisms are well understood. An ability to account for interactions over a range of seasonal conditions using long-term records (or future climate predictions) can be highly advantageous (Harmsen 2000). Two examples using simulation models to quantify the effects of soil constraints on crop growth are Hochman et al. (2007), who tested the effect of ESP and chloride on the soil water lower limit of a crop, and Barber and Mackay (1985), who quantified the effects of soil water and root properties on P uptake.

Statistical analysis of multiple soil constraints

In research on soil constraints, regression has proven to be a useful technique to identify critical values for nutrients such as N or P (Peverill, Sparrow, and Reuter 1999). Where the yield response is expected to follow a diminishing returns relationship, the data can be linearised by a suitable transformation (e.g., Dyson and Conyers 2013). Adaptation of this approach to multi-factor interactions can be difficult to interpret if there are large number of variables. For example, classical regression techniques provide only limited insight into observed relationships, and it has proven difficult to separate responses to causal factors from responses to variables correlated with those causal factors in both mono- and multi-factor models (Shatar and McBratney 2004). Several studies have used statistical approaches to evaluate the effect of multiple soil constraints on crop yield, mainly by fitting the boundary line function discussed above. Examples include Shatar and McBratney 2004; Cossani and Sadras (2018) and Milne, Ferguson, and Lark (2006).

Machine learning algorithms

Machine learning algorithms (MLA) are of particular interest for the study of multiple soil constraints due to their ability to handle both continuous and categorical variables, and their robustness to data complexity. In contrast to mechanistic simulation modelling, prior understanding or conceptualisation of the physicochemical and biological processes that under-pin crop growth and yield are not required (Breiman 2001). This is because MLAs use data-driven approaches to discover relationships between variables (Elith, Leathwick, and Hastie 2008). Another prospect for the use of MLA in multiple soil constraint research is their ability to uncover the relative importance of explanatory variables. This is useful to identify soil constraints potentially impacting crop growth across different scales.

MLAs belong to three main groups: regression, classification, and clustering, based on the type of tasks they perform. Regression-based algorithms [e.g., random forest (RF), boosted regression tree (BRT), cubist, etc.] are used to model and predict continuous or numerical variables. These are useful in generating dense soil physio-chemical data for the broad scale assessment of soil constraints. In contrast, classification-based algorithms, such as classification and regression tree (CART), are used for modelling and predicting categorical variables. These are useful for interpolating simulated stress indices across a broader scale. Finally, cluster-based algorithms such as fuzzy-k-means (FKM) and k-nearest neighbour (KNN) are used for finding natural groupings or clusters of observations based on the inherent structure within the dataset.

Despite their potential, relatively few studies have employed MLAs to model the relationship between crop yield and soil constraints. Exceptions include Filippi et al. (2019), who used RF to predict the depth to soil pH constraints and their relationship with cotton yield. Filippi, Jones, and Bishop (2020) also used RF to predict the depth to soil sodicity using different thresholds of ESP.

Overall, diagnosis and the interpretation of management effects on the amelioration of multiple constraints will potentially require adoption of all the three data analysis approaches. Mechanistic modelling could be used as a validation and/or data-generating tool to supplement experimental/survey data. This analysis approach may provide a complimentary tool to both machine learning and statistical approaches.

Diagnosis of soil constraints

The identification and quantification (diagnosis) of single soil constraints to grains crops is well developed and can be generally divided into either soil or plant-based approaches. Plant-based assessments can be based on visual symptoms or elemental analysis of plant tissue samples. Most commercially available soil tests focus on chemical properties (nutrition, pH, and toxicities such as Al^{3+}) rather than physical constraints. Recent advances in low-cost molecular techniques have extended soil testing to properties such as disease risk (Heap and McKay 2009) and these procedures are expected to gain greater acceptance in the future.

Two key considerations are often overlooked when diagnosing the effect of soil constraints (Sumner and Farina 1986). Firstly, it is important to consider that the crop does not react to any treatment of a constraint *per se*, but to the soil's response to the treatment (usually indicated by soil analysis). Furthermore, routine soil tests are not a direct measure of the conditions experienced by the plant root over the growing season, but are approximations (Sumner and Farina 1986). Consequently, plant assessments of soil-based constraints (especially nutrients) are arguably more reliable than those based on soil analysis.

For quantification of multiple constraints, we propose the measurement of changes in relative growth rate (biomass) (Chapin et al. 1987; Saito and Goepfert 2008; Lambers, Pons, and Chapin 2008). Growth rate is a sensitive and widely used indicator of most constraints and relatively adaptable to measurement by many of the recent innovations in rapid sensing (proximal, aerial, or satellite-based) and reallocation of resources (especially C and nutrients) between different plant components (e.g., shoot:root).

The rapid development of new field-based sensing technologies using platforms ranging in scale from proximal devices to unmanned aerial vehicles, aircraft, and satellites, provide the opportunity to not just indirectly assess the presence of a soil constraint/s but also directly assess the location and impact on the plant itself on both a spatial and temporal basis (Weiss, Jacob, and Duveiller 2020). Most applications of this technology have focused on providing data to support in-season crop management e.g., N fertilisation, and more recently in crop phenotyping (see review by Potgieter et al. 2018), but could potentially be used to assess the impact of multiple soil constraints on the crop over the season. Whilst grain yield is the ultimate variable of interest to growers, plant assessment approaches enabling measurement of the rate of change in biomass over time e.g., normalised difference vegetation index (NDVI), will be a key diagnostic indicator of whether the plant is “constrained”.

Management strategies for multiple constraints.

Strategies for managing multiple constraints can be grouped into several main categories, as outlined below.

Tolerance

Genetic tolerance through plant selection has been developed for several soil constraints, both physicochemical and biological in nature. Typically, the tolerance is species specific (although in theory genetic engineering could be used to develop multiple species tolerance). There have been improvements in the development of specific tolerances to some major soil constraints in grain crops e.g., B tolerance in cereals (Yau and Ryan 2008) and salinity in durum wheat (Munns et al. 2003). Poor soil physical conditions, such as high soil strength, have arguably proven more elusive to overcome (Botwright, Acuña, and Wade 2012; Lynch and Wojciechowski 2015). This apparent lack of success in breeding for tolerance to specific soil constraints results from the need to satisfy multiple selection criteria including high yield, the existence of both genetic variability and stability, the persistence of traits across generations, and the availability of suitable procedures to enable rapid and cost-effective quantification (Sadras and Richards 2014).

Breeding selection for multiple (soil) constraints requires a gene pyramiding approach where several traits are combined using either a conventional or molecular breeding approach, e.g., B and salinity in lentils (Nuttall et al. 2010). The prospect of developing tolerance for soil structural constraints, however, seems less promising, although there are programs currently in place to address constraints indirectly arising from poor soil structure e.g., waterlogging tolerance (Manik et al. 2019). Similarly, there has been considerable research to identify “nutrient efficient crops” (especially P and N) e.g., Bovill et al. (2013). The current lack of commercial varieties with clear advantages in nutrient use efficiency has, in part, resulted in farmers managing nutrients via both rotation choice (for N) and ongoing refinement of soil testing and variable fertiliser rate strategies. Development of species with true “tolerance” to a constraint often involves radical changes to metabolism that would render the plant unsuitable for commercial agricultural production e.g., succulents with crassulacean acid metabolism (CAM) pathways to minimise transpiration losses of water by transpiration during the day. There is an additional problem that tolerance may be achieved to a particular level, beyond which the crop will fail without removing the constraint. This is particularly the case for salinity and

acidity where the problem continues to increase without mitigation.

Amelioration

The “amelioration” of soil constraints is defined as the chemical or physical change of key soil attributes to overcome a constraint or change the soil to fit the requirements of the plant. The effectiveness of different amelioration strategies for grains systems was recently summarised by Davies et al. (2019). Where multiple constraints are present, a combination of ameliorants, or the use of ameliorants with capacity to address multiple constraints, will be required.

For example, the application of animal manures and other forms of nutrient enriched organic matter (OM) have produced significant increases in crop production on poorly structured clays in southern Australia (Armstrong et al. 2007; Gill, Sale, and Tang 2008 Sale et al. 2021). Although the mechanism/s underpinning this crop response remain unclear (Davies et al. 2019), it is believed to result from the ability of the OM to improve both soil structure and nutrient supply (Sale et al. 2017). Recent research also indicates that the application of some forms of OM high in nutrients can overcome toxicities arising from waterlogging and improve N nutrition (Dunsford et al. 2021) by stimulating root growth (Wang et al. 2020). On light sands in Western Australia, deep tillage alone has been highly beneficial in mitigating multiple constraints including water repellence, machinery-induced compaction, subsoil acidity, and fertility of the A horizon (Davies et al. 2019).

Many constraints, but especially multiple soil constraints, arise from long-term pedological conditions and are also strongly linked to other soil properties. For example, high B concentrations and sodicity in alkaline subsoils are strongly correlated with low rainfall and high clay content (Nuttall et al. 2003), which limit the leaching of both B and sodium from the root zone. An important consideration when using amelioration strategies for the treatment of multiple constraints is whether synergies over time can be achieved based on a phased approach. For example, in dense sodic subsoils with high sodium and B, soil structure may first need to be improved by addressing dispersion (e.g., via gypsum application), and once adequate drainage is established, more intractable constraints such as salinity high B may be reduced through subsequent increases in leaching (Adcock et al. 2007). Once soil structure has been improved through application of gypsum and/or organic matter, management practices such as controlled traffic that maintain soil structure should be adopted.

Developments in materials sciences provide potential opportunities for new ameliorants to manage multiple soil constraints. Early research on the use of polymers, such as polyacrylamide (PAM), to improve plant production on poorly structured soils showed promise, especially when used in combination with other amendments such as composted manure, creating significant synergies (Wallace and Wallace 1986). The synergies arising from the use of dual ameliorants may help to overcome the logistical barriers that currently challenge the adoption of some amelioration technologies.

There has also been recent interest in the use of nanotechnology to improve the effectiveness of soil ameliorants that can address issues relating to both soil structural and nutritional constraints. Some examples include hydrogels, nanoclays, and nanofertilisers (Fraceto et al. 2016; Mandal et al. 2019). An advantage of these nanomaterials is their ability to move more easily into subsoils where soil multiple physicochemical constraints are most severe. Unfortunately, the relatively high costs for the development and production of such materials may limit their use in broadacre dryland cropping systems, especially where relatively large quantities of ameliorant are required (e.g., leaching of Na in sodic soils).

Avoidance

Often plants adapt to severe constraints via avoidance mechanisms. Many so-called examples of tolerance are actually avoidance. For example, native grasses adapted to nutrient poor soils reduce growth rates to better match rates of plant demand with the soil's ability to supply the nutrient (Armstrong and Helyar 1993). Many "drought tolerant" plants avoid drought by remaining dormant (or changing flowering date) until sufficient water is available, or alternatively produce root systems with access to soil water supplies located deep in the profile (Lambers, Pons, and Chapin 2008). In grains crops, the analogy would be to alter the timing of flowering to reduce the probability of water stress and/or frost during flowering and grain filling by using both quicker maturing varieties and early sowing (Barlow et al. 2015). This contrasts with attempts to breed for increased physiological "tolerance" of crops to frost, which to date has met with limited success. The utilisation of avoidance strategies recognises that for many constraints, the physiological "cost" of tolerance to a constraint exceeds the potential benefit. At an extreme level of "avoidance", sections of (or whole) paddocks with severe constraints that are difficult to remediate can be removed from crop production.

Potential strategies for crops to “avoid” areas with multiple soil constraints are based on space and time. Many soil constraints exhibit high spatial variability within a paddock, both at a 2D scale (Armstrong et al. 2009) and within a profile (Nuttall et al. 2003), creating a 3D effect. Knowledge of where these constraints occur could allow use of a variable rate type approach where varieties with varying physiological tolerance to the multiple constraints present are sown in certain parts of the paddock, or specific amelioration strategies are applied to those areas or part of the soil profile where the constraints are most severe. Alternatively, because most physicochemical constraints become greater with increasing soil depth in clay soils (Nuttall et al. 2003), strategies that concentrate root growth in the upper part of the soil profile could “avoid” these constraints. Whereas this approach seems to contradict the goal of selecting for deep rooting (Lynch 2013), an alternative approach is to develop new root ideotypes and associated farming systems that focus on making better use of soil water in upper parts of the profile (Rao et al. 2021). Where large reserves of subsoil water are already stored in deeper sections of the potential rooting zone, a species with greater tolerance to particular soil constraints could be grown in that season rather than a more intolerant species e.g., barley grown rather than lentils in saline subsoils. Examples of “avoiding” soil constraints in time include rotating with non-host varieties to minimise the effect of soilborne disease e.g., *Pratylenchus* spp. (Hollaway et al. 2000) and the use of very early or late sown crops to avoid waterlogging arising from sodicity and perched water tables.

“Live with the problem”

Management of crops should aim to maximise profitability rather than yield. Where there is no readily available option to manage multiple constraints, profitability can be increased by minimising input costs, such as N fertilisers, where soil conditions limit the chance of a favourable response. This approach accepts that multiple soil constraints cannot be economically reduced, that low yields are inevitable, and the only way to increase profit is to reduce inputs. In some cases, a small portion of a paddock may make consistent losses but cannot be taken out of crop production due to logistical constraints. Variable rate approaches make this relatively easy to achieve within a paddock.

Conclusions

The productivity of dryland cropping systems can be limited by multiple soil constraints, which can vary both spatially (in three dimensions) and temporally. Their diagnosis and management are much more difficult than for single constraints due to the range of potential interactions with each other, as well as other non-soil factors, especially climate. In most modern cropping systems, the impact of any one single soil constraint is likely to be limited but the combined impact of multiple constraints can be significant, especially from a financial perspective. The impact of multiple constraints is best understood by the paradigm that the intensity of a constraint is proportional to the degree of co-limitation between different constraints and that productivity will be maximised when all constraints are equally limiting. This approach requires a much more complex response than a simple strategy based on the LoM paradigm where the most limiting constraint is identified and managed.

Traditionally soil constraints have been identified and diagnosed using both soil chemical and plant tissue analysis. These approaches are not necessarily effective or economically feasible for crops experiencing multiple soil constraints that vary in space and time. However, crop growth rate is a sensitive indicator of most constraints and relatively adaptable to measurement by a range of recent innovations in rapid sensing of both soils and crops. The collection of the allied data needed to interpret analysis outputs, as well as its use in the parameterisation of simulation modelling, needs to be exploited. We argue that the common factor linking a range of soil physical, chemical, and biological constraints on crop productivity is typically water, especially in dryland systems, so that assessments of water use and N (Ryan et al. 2009; Sinclair and Ruffy 2012) and extreme temperatures will need to be considered. It is also important to acknowledge that marketable yield rather than biomass will ultimately be central to any form of economic analysis. Recent developments in data analytics can provide useful complementary approaches to mechanistic simulation modelling, which has proven useful for accounting for interactions with climatic factors on crop production as well as traditional statistical approaches.

An important implication of the MLH paradigm is knowing the critical limits where crops can adjust for both limiting resources and toxicities. Farmers have access to several strategies to improve adaptation to potential multiple constraints: genetic tolerance, amelioration of soil constraints, avoidance, and “live with the problem”. Managing multiple co-limitations, however, will most likely require a pyramiding of approaches. An important consideration is not which stress or stresses is/are most important, but what

is the marginal return of applying different interventions (Sinclair and Park 1993). Many farmers currently utilise a “live with the problem” approach to maximise profitability, especially in environments where seasonal production risk is high. Novel, cost effective approaches will be required to achieve the transformational increases in productivity needed to overcome the “yield gap”.

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CHAPTER 23

SOCIOECONOMIC CONSEQUENCES OF SOIL CONSTRAINTS

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Abstract

In this chapter we address the issues of the worldwide social and economic effects of soil constraints; the importance of considering social and economic factors when identifying appropriate management options for soil constraints; and areas for further research. We discuss the poverty, food security, social and micro and macroeconomic consequences of soil constraints at the farm, regional, national, and multinational levels. The extra food required for the increasing world population will largely have to be grown in the existing cropland area of developing countries. This will place additional strains on soil quality because it will require increased yields and cropping intensity, which are currently leading to soil degradation in many regions. Unless new approaches to cropping are developed that are sustainable, increased pressure on existing cropping soils may result in reductions in GDP and farm viability, with increases in poverty, malnutrition, and food insecurity, which in turn will increase the probability of outmigration and conflict. Development of solutions to these issues will need to involve the target communities in transdisciplinary and participatory methodologies that include gendered approaches. Policy responses will need to take account of the social, cultural, and economic constraints faced by many of these farming communities and use a combination of mechanisms to address the relative levels of public and private benefits and costs of the solutions developed. Unless these changes are managed efficiently and effectively, it could lead to a collapse of soil resources with consequent catastrophic outcomes.

Keywords: Social, cultural, and economic constraints, poverty, food security.

Introduction

Demand for food is expected to increase by between 60-100% by 2050 as the world's population increases to around 9.7 billion (Elferink and Schierhorn 2016). More worryingly, the number of people affected by hunger has been rising since 2014, with an estimated 2 billion people, or one quarter of the world's population, not having regular access to sufficient, safe, and nutritious food (FAO et al. 2020). With current trends, the world will not achieve Zero Hunger by 2030 and in fact the number of people affected could surpass 840 million by then. A range of issues will need to be addressed to meet the demand for food including a) reducing wastage; b) improving supply chains, public expenditures, and investment policies; and c) implementing policies to encourage a shift to healthier diets, reducing the cost of healthy diets to the poor, and increasing their purchasing power. In addition, significant increases in agricultural production will be required (FAO et al. 2020; Davis et al. 2016). However, improvements in productivity using current technologies will not be achieved without important negative effects on the environment, so enhancements will be required to increase environmental sustainability (Davis et al. 2016).

Of the nearly 5 million hectares of agricultural land area (crop and grazing land), about one-third (1.6 million hectares) is used as cropland (arable land and permanent crops) (Table 23.1). However, the percentage used as cropland in different regions varies from 13% in Oceania to 94% in India. While dramatic increases in the land used for agriculture have occurred worldwide since 10,000 BCE when agriculture first began, more recently the area used has levelled off, and since 2000, even begun to decline in many locations (Ritchie and Roser 2013). However, while a decline in cropland has been observed in some regions, in others, particularly in Latin America, Africa, and Asia (excluding China and India), cropland has increased, either due to land clearing or due to the conversion of grazing land to crop production. At the same time, increasing competition with cropland from other human activities (e.g., urbanisation), reductions in the area of new cropland available due to the negative impacts clearing has on natural ecosystems (e.g., reductions in carbon storage), and losses of cropland due to unsustainable land management (e.g., soil erosion, salinisation) (World Bank 2007; Godfray et al. 2010) mean that we will increasingly have to rely on existing crop land and their soils to produce the required increases in food.

Table 23.1 Cropland area as percentage of agricultural area by region (million hectares in 2016)

Region	Cropland Area ^a	Agricultural Area ^b	% Crop
Africa	297	1185	25
Brazil	82	278	30
Canada	48	62	77
China	122	628	19
Europe (excl. Russia)	187	285	66
India	169	179	94
Latin America & Caribbean (excl. Brazil)	119	484	24
Middle East	34	236	15
Oceania	52	393	13
Rest of Asia (excl. India & China)	206	514	40
Russia	122	222	55
United States	155	408	38
World	1 593	4 873	33

^a Cropland refers to the area defined by FAO as “arable land and permanent crops”.

^b Agricultural land includes crop and grazing land, with grazing land referring to the area defined by FAO “permanent meadows and pastures”.

Source: Developed from: Goldewijk et al. (in prep.).

A recent decomposition analysis by Blomqvist, Yates, and Brook (2020) using data from 1961 to 2015 found that the majority of the increase in crop production over the last 54 years has come from increases in aggregate yield (89%) rather than cropland area (11%). Pure yield (output of crop per unit of harvested area in a location) was found to be the most important driver of rising aggregate yield (56%), with other drivers such as cropping intensity (harvested area as percentage of cropland area), crop composition, and country share accounting for about one third. Cropping intensity was the most important of these, with it having a greater effect over the last 15 years of the period. Ray and Foley (2013) also found that between 2000 and 2011, harvested land area grew roughly four times faster than total standing cropland area. However, they suggest that this increase in cropping intensity could have significant environmental impacts, including long-term deterioration of soils and the land base. Conversely, improvements in pure yield have slowed in the last 20-30 years (Blomqvist, Yates, and Brook 2020), with yields stagnating in some regions.

The increases in productivity and production required to meet the growing demand for food will mostly need to come from developing countries, and to a lesser extent transitional countries, from where productivity growth has mostly originated in recent years (USDA 2019). If, as implied by the findings of Ray and Foley (2013) and Blomqvist, Yates, and Brook (2020), increases in total factor productivity (TFP) will most likely come from reductions in yield gaps (increases in pure yield closer to the water-limited yield potential possible in a region) and reductions in harvest gaps (or increased cropping intensity) and not from increased cropland area, then there is a danger of increasing and expanding soil degradation. New approaches will need to be found that both overcome existing soil constraints and reduce the use of technologies that are currently degrading soils.

Yield gaps and harvest gaps have many causes and vary substantially in different geographic, social, and agronomic contexts (Ray and Foley 2013), with the biggest gaps occurring in Africa and parts of Latin America. For example, increased cropping frequency can lead to declines in soil organic carbon, diversity of soil microbiota, arthropods, and other species that affect soil structure and fertility, and ultimately reduce yield and widen yield gaps (Ray and Foley 2013). Similarly, soil constraints (both naturally occurring and human induced) can also limit yield and cropping intensity (Orton et al. 2018; Page et al. 2021) and hence economic returns to cropping. Page et al. (2021) also suggest there is little peer-reviewed literature on crop tolerances and yield responses to multiple soil constraints in sub-tropical cereal, pulse, and oilseed crops and that these tolerances and responses are variable for the different constraints. This makes it very difficult to assess the economic impacts of these soil constraints.

In this chapter, we examine the socio-economic consequences of soil constraints. We first discuss the importance of socio-economic factors that influence soil constraints, and then outline the common social and economic impacts of soil constraints globally, drawing examples from the literature. We then outline why it is important to consider social and economic factors when identifying appropriate management options for soil constraints.

Importance of socio-economic factors in considering soils constraints

Physical, chemical, and biological soil constraints lead to direct and indirect socioeconomic impacts that are observable at farm, regional, and national scales. Empirical studies focused on understanding these socioeconomic impacts largely centre around financial consequences (e.g.,

Orton et al. 2018; Panagos et al. 2018; Diao and Sarpong 2007), but also extend to other metrics of development and wellbeing such as poverty and food security (Wanjiku et al. 2003). These topics are of interest to a range of disciplines within the social sciences. Researchers apply qualitative and quantitative methods to the study of these socioeconomic consequences to capture the historic, current, and future expected impacts of soil constraints, develop strategies to minimise negative consequences, and inform policy aimed at improving environmental conditions to increase positive socioeconomic outcomes in agriculture.

Current understandings of the relationships between soil constraints and socioeconomic consequences are nuanced as the relationships can be approached by examining causality in two directions – that is, a) the impact of socioeconomic factors on soil constraints; and conversely, b) the social and economic impacts of soil constraints. There is a substantial body of research investigating the impact of growth and development on soil constraints (i.e., Erk 2010; Osgood and Lipper 2001). Studies have found that soil quality declines as agricultural production and population levels increase (Stocking and Murnaghan 2002; Vlek et al. 2010; Vu et al. 2014). Socioeconomic factors subsequently influence and interact with soil fertility and farmers' ability to deal with soil constraints.

Soil quality contributes to the provision of a wide range of ecosystem goods and services, and therefore soil constraints can have negative socioeconomic consequences stemming from a range of associated environmental factors. Palm et al. (2007) provide an overview of the processes and services provided by soils, highlighting their nutrient and water provision, cycling, and regulation functions that deliver provisioning, regulatory, cultural, and supporting services. Most commonly, the socioeconomic consequences of soil constraints are measured in direct relation to the food and natural resource provisioning services of soils, as income and profitability decreases with declining yields caused by soil constraints. These impacts have spill-over effects on other factors related to poverty, while changes in agricultural production also directly impact food security. Empirical findings supporting these trends are highlighted in the section below. This is followed by an overview of the ways in which these factors can be best considered in identifying improved management options regarding soil constraints.

Common social and economic impacts of soil constraints worldwide

Soil constraints can cause producers to suffer from decreased yields, increased input requirements, and increased output variability (Osgood and Lipper 2001). These factors reduce agricultural productivity and cause a range of detrimental social and economic spill-over effects. As a result, soil constraints have significant consequences for food and financial security and progression towards rural development objectives.

Establishing a representative figure, or drawing all-encompassing conclusions, about the socioeconomic consequences of soil constraints is not feasible as physical, chemical, and biological soil constraints impact agricultural productivity in different ways, and to different extents across different regions (Lipper 2001). Still, examining farm-level, regional, and national impacts offer insights into the magnitude of the consequences and provide evidence to inform future policy directions. The methods used to capture socioeconomic impacts include qualitative interviews (e.g., Mango and Hebinck 2016) and production output models (e.g., Orton et al. 2018), and the results point to a need to improve soil quality conditions and prevent subsequent land degradation to safeguard rural livelihoods. This section highlights the socioeconomic consequences of key concern associated with soil constraints, as exemplified through case studies in different geographic regions around the world.

Social consequences: Examples from Africa, Asia, Australasia, and Europe

Increased exposure to rural poverty is a primary social consequence of soil constraints (Wanjiku et al. 2003). Much like there are many different indicators used to measure soil constraints, there are several metrics that are applied to capture poverty impacts. These include human capital indicators, such as life expectancy, and financial indicators, such as GDP and household assets (Barrett and Belvis 2015). Because of their poverty implications, soil constraints impede progress towards meeting the first Sustainable Development Goal; restricting rural households' ability to fulfil their basic needs.

The poverty consequences of soil constraints are exacerbated by their inequitable impacts on different socioeconomic groups. For example, evidence in Ghana has shown that soil erosion has caused poverty reduction rates to be slower in the nation's poorest regions, relative to the national average (Diao and Sarpong 2007). This trend can be explained by

impoverished communities more often cultivating on more heavily degraded lands. Similarly, in Morocco, the vulnerability of rural populations was highlighted by the fact that 75% of impoverished populations depend on agriculture for their livelihoods, the industry most heavily impacted by soil constraints (Dahan 2018). Soil constraints are said to contribute to poverty traps wherein declining yields impact farmers' income, which then inhibits their ability to make investments to improving soil quality (Barrett and Bevis 2015). These impacts are especially severe for smallholder farmers (Eriksen and Watson 2009) and can further interact with factors such as land ownership and individuals' knowledge and resources (Dahan 2018).

Utilising ethnographic interviews in Kenya, Mango, and Hebinck (2016) set out to understand farmers' experiences and perceptions of the change associated with soil constraints. The authors found that farmers' returns to labour inputs had decreased so substantially with land quality changes that farming operations were approaching a threshold of economic unviability. As a result of these changes to returns, poverty rates were on the rise. Under an alternative qualitative approach, Eriksen and Watson (2009) undertook a structural analysis to review the processes through which land degradation impacts different facets of sustainability in the African savannas. This analysis highlighted the compounding interactions between processes impacting environmental systems, such as declining soil quality, on conflict, social vulnerability, and farming viability. The finding that soil constraints and degraded land are worsening political tensions has also been reported by Dahan (2018), who identified increased occurrences of conflict between local tribes and communities that collectively manage 70% of Eastern Morocco's land base.

Soil constraints impact food security and malnutrition in rural communities indirectly, because of poverty, and directly, through the ability to grow food for consumption through subsistence farming. In developing economies, core crops grown as part of agricultural operations are also a key source of calories and nutrition, and thus declining yields worsen malnutrition (Wanjiku et al. 2003). These consequences also have unequal impacts across socioeconomic groups, where poor rural communities tend to experience higher levels of food insecurity than national averages (Diao and Sarpong 2007).

In China, Ye and Van Ranst (2009) evaluated food security more broadly as a comparison between per capita food supply and demand and found that while there was an 18% surplus in food supply in 2005, a shift to a food supply deficit was expected by 2030 to 2050. The magnitude of this expected deficit was approximately 4, 16, or 26% when soil degradation

was modelled under zero degradation, business as usual, and increased degradation scenarios, respectively. Such figures call into question our global capacity to feed a growing population under current growth trajectories and environmental impacts of food production.

In addition to poverty and food security impacts, soil constraints can have negative cultural influences in agricultural communities. One such example is in the Italian Alps, where terraced landscapes are key both to agricultural production, and to tourism as iconic aesthetic landscapes (Stanchi et al. 2012). As soil quality declines on these terraces, values associated with the culture and imagery of these landscapes come under threat. In addition, the cultural legacy of historic family farming traditions can be disrupted if farmers need to adapt to soil constraints by changing the crops grown on their land. Beyond a change in growing portfolios, where the viability of smallholder farming becomes uncertain, individuals might come to question whether they should pursue alternative livelihoods (Eriksen and Watson 2009). In combination, these social impacts can then lead to outward migration of impoverished people away from rural farming communities to urban communities or across borders (Dahan 2018).

Economic consequences: Examples from Africa, Asia, the Middle East, Australasia, Europe, and Central and South America, and North America

The economic consequences of soil constraints are commonly quantified using measures of soil erosion control, the market price of soil, replacement costs of fertilisers compensating for nutrient loss, and the opportunity cost of crop productivity loss (Panagos et al. 2018). These consequences can be considered at a range of scales but are often discussed in the literature at state or national levels.

In Central and South America, the annual production loss in maize, potato, soybean, and wheat crops due to soil erosion was estimated to be US \$90.1 million from soil databases mapped with geographic information systems, findings from a synthesis of erosion productivity studies, and global agricultural production statistics from the FAO (Biggelaar et al. 2004). In a study from Iowa in the US with a similar focus on soil erosion, costs have been estimated based on changes in land values from loss of corn and soybean yield potential and land rents (Duffy 2012). Erosion impacts were found to decrease land values by approximately \$340 USD acre⁻¹, ~ 4.8% of 2011 land values.

An estimate of the economic impacts of soil constraints using land values in New Zealand used hedonic pricing methods to capture the value

of soil quality through land sales data (Samarasinghe and Greenhalgh 2013). Rural lands with higher soil quality were linked to higher land values. Mirzabaev, Stokov, and Krasilnikov (2018) also used hedonic pricing methods in an economic assessment of soil erosion and land degradation but used net agricultural profits instead of land values. Net agricultural profits were modelled as a function of agricultural inputs and outputs, their prices, climate variables, farm capital, and land quality using information obtained from farm surveys and remote sensing data. The authors found that in Central Asia, thirty years of land degradation resulted in an average 27% decline in household agricultural profits in the 2009-2010 season.

Orton et al. (2018) evaluated the impacts of different soil constraints on wheat production in Australia under an opportunity cost framework. Using a model that incorporated detailed land quality and growth condition data, pricing information, and costs of treatments to manage and improve soil quality, the authors estimated yield gaps equivalent to annual costs of \$1300, \$400, and \$200 million Australian dollars for soil sodicity, acidity, and salinity, respectively. With a focus on state-level impacts in Australia, Herbert (2009) looked at the cost of land degradation in the South-West agricultural region of Western Australia by evaluating lost productivity through differences in farm operating surplus. Their model integrated historic gross agricultural production data and individual farm physical and financial data obtained from farm surveys. In a single year snapshot of impacts, they estimated the average annual opportunity cost of various soil constraints due to lost productivity to be \$498, \$344, \$333, and \$251 million for soil acidity, salinity, subsurface compaction, and water repellence, respectively. At a lower scale of impact, wind erosion, waterlogging, soil structure decline, and water erosion were estimated to cost \$71, \$29, \$15, and \$10 million/yr, respectively. The authors highlighted the importance of looking at the relative values across soil constraints, rather than the absolute number estimated, which indicates that soil acidity is of particular concern in the state.

Emadodin, Narita, and Bork (2012) assessed the economic and environmental impacts of soil degradation in Iran under a framework that reviewed the sustainability of agricultural practices. The economic costs of soil degradation were quantified under a replacement cost approach, such as the cost of chemical fertilisers as a cost representing the loss of soil nutrients. The authors estimated the annual total cost of soil and water degradation to be US \$12.8 billion. This figure represented approximately 35% of the agricultural sector's GDP and 4% of national GDP.

Barrett and Bevis (2015) observed correlations between soil degradation and poverty in sub-Saharan Africa using readily available GDP measures to

represent poverty rates. Causal relationships between changing soil constraints and economic impacts can also be estimated and have traditionally been based on incorporating national resource accounting into GDP measures. These GDP approaches have been undertaken for many years. For example, in the 1980's, the economic cost of soil erosion as a percentage of agricultural GDP was estimated to be 4-6% in Mali (Bishop and Allen 1989) and 1.6% in Java, Indonesia (Magrath and Arens 1989). In Malawi, the cost of soil loss in 2018 was estimated to be 0.42-2.1% of agricultural GDP, and 0.25-1.28% of total GDP with soil loss scenarios ranging from 10-50% (Solomon et al. 2018).

At the European Union scale, the cost of soil erosion has been estimated based on lost agricultural productivity and GDP using a general equilibrium model (Panagos et al. 2018). Soil erosion models were used to predict where severe erosion is expected to occur and estimate the resulting likely loss of crop productivity for 10 crops, before macroeconomic impacts were evaluated for the agricultural sector and GDP. The annual cost of lost agricultural productivity was estimated at €1.25 billion. Looking at national figures within the European Union, Italy experienced the greatest annual economic impact at €619 million, representing a 0.75% decline in agricultural production, and a 0.002% decline in national GDP. Under a similar framework in Ghana, Diao, and Sarpong (2007) developed a multimarket model that compared future scenarios of soil loss and accounted for interactions between outputs of eight different crops in urban and rural divisions. When comparing status quo levels of soil loss to alternatives of no soil loss, the authors found that soil constraints were expected to reduce the country's total agricultural GDP outputs by approximately 5% (US \$4.2 billion). These examples highlight the substantial macroeconomic impacts soil constraints can have.

The importance of considering social and economic factors when identifying appropriate management options for soil constraints

Managing and addressing soil constraints

While there are wider societal impacts stemming from soil constraints, the consequences, and accordingly the benefits, of healthy soils often have more direct and severe impacts for farming households (Stevens 2018). Social and economic impacts are already materialising in rural communities, and these impacts are expected to amplify as communities become more vulnerable under subsequent worsening of soil constraints and the increased

compounding pressures of climate change (Dahan 2018).

Managing soil constraints and improving soil quality largely relies upon changing landowner behaviour (Pannell and Claassen 2020). Thus, to make progress in addressing soil constraints, a thorough understanding of behaviour is required. When selecting farm-level management practices and accompanying policy or programs, program success is underpinned by farmers' perceptions of these practices and how they may react to different policy mechanisms or institutions (Pannell et al. 2006). However, farm-level management practices and farmers are extremely diverse, implying that encouraging adoption is a complicated task that differs substantially between contexts. Thus, it is important to consider economic, social, psychological, and cultural factors that might influence farmer behaviour to effectively reduce soil constraints.

Theories of how and why new practices are adopted were initially based in sociology and supplemented with epidemiological models (Pannell and Zilberman 2020). Pannell et al. (2006) tailored the Diffusion of Innovations Theory (Rogers 2003) to the agricultural context, suggesting that the rate and speed of adoption depends on how landowners perceive the relative advantage and trialability of the practice. Relative advantage refers to how favourable or desirable a new practice is to a landowner relative to their current practice, considering a range of financial, lifestyle, cultural, and environmental factors. Perceived relative advantage can be a complicated concept in the context of soil constraints depending on the time lags until soil and yield improvements can be realised; how obvious or observable these improvements are to the landowner; and how compatible the practice is with a landowner's operation, lifestyle, or culture. Trialability is a measure of how easily landowners can test a new practice on a smaller portion of their land to assess its costs and benefits relative to other options. Practices requiring large initial investments are more difficult to trial and therefore less likely to be adopted, as are those that are more complex or require considerable specialised training.

Financial or economic considerations are thought to be the primary drivers of adoption (Bartkowski and Bartke 2018; Baumgart-Getz, Prokopy, and Floress 2012; Prokopy et al. 2008). Producers will often adopt new practices that will increase their profitability or that provide a net benefit; however, depending on the level of net benefits provided, there can be substantial lag times until widespread adoption occurs (Pannell 2008), and agencies may have an interest in enhancing the speed of adoption. Income, access to credit, farm size, and other factors relating to a farmer's means to make the often large initial investments are also positively linked to adoption across many regions and contexts. However, for smallholder

farmers, subsistence or food production for family consumption can take priority over income or profitability in the adoption decision (Llewellyn and Brown 2020). While profitability and yield tend to be the most important determinants of farmer behaviour, a myriad of other factors also substantially influence adoption decisions and can explain why, at times, yield and profit improving practices may not be widely adopted (Ghimire et al. 2021).

Factors relating to culture, norms, and identity impact landowners' perceptions of soil management practices, explaining why some farmers might adopt environmentally beneficial practices at the expense of income, or why others might avoid environmentally beneficial practices even though it could improve their earnings (Ghimire et al. 2021; Davidson et al. 2019). Farmers may reject practices on cultural or intergenerational grounds, feeling that adoption would go against work undertaken by their ancestors (e.g., Neumann et al. 2007). Group norms can also present challenges, as a fear of behaving out of line with other group members can present a barrier to adoption (Ghimire et al. 2021; Mills et al. 2017). Thus, community-level extension or education programs are at times more effective than strategies targeting individual farmers in creating positive environmental norms and increasing personal feelings of a responsibility to act (van Dijk et al. 2015). On the other hand, notions of an individual's social identity may be harnessed to enhance adoption, drawing on ideas that "the good farmer" acts as a steward of the land (McGuire, Morton, and Cast 2013).

Attitudes can help explain how farmers perceive the benefits or costs of new technologies or practices, which in turn can explain why financial or economic implications of a new practice do not fully explain adoption. Those who are more environmentally concerned or feel normative pressure to make environmental improvements, are more likely to adopt environmentally-beneficial practices because they gain more personal, non-financial benefits, or satisfaction from taking action (Davidson et al. 2019). Risk attitudes and loss aversion can also explain why some farmers can be hesitant to adopt new practices that should increase profitability (Streletskaia et al. 2020). Some farmers tend to give more weight to small probabilities of loss associated with adopting a new practice, while others are averse to the uncertainty associated with how the practice might impact their yields and income, resulting in the delayed adoption of profit maximising technology (e.g., Liu 2013). Impatience or preferences for maximising earnings in the short-term at the expense of maximising returns over a longer horizon may also prevent the investments in soil quality that economists predict farmers would make (Duflo, Kremer, and Robinson 2011). Many of these attitudes are not easily shifted, and successfully encouraging adoption will require relatively long periods of interaction with

farmers to build trust and credibility (Mills et al. 2017).

Gender can be an important determinant of adoption, especially in lower-income countries and among smallholder farms. Rural women's rate of technology adoption that can improve livelihoods and soil quality often lags that of male farmers (Huyer 2016), in part because they face high levels of social oppression and economic inequalities in farming (World Bank, FAO, and IFAD 2009). Lower technology adoption rates by women can be attributed to sociocultural norms, longstanding beliefs and expectations regarding gender roles, a lack of agency in decision-making, and a lack of access to resources (Rola-Rubzen et al. 2020). Rural women have been found to have less access to financial and knowledge resources, in addition to less access to advanced agricultural technologies, leaving them reliant on traditional farming methods (Radovic-Markovic et al. 2020). Where access is available, women face additional constraints with the design of many technologies (e.g., heavy machinery designed for men), or are less likely to have access to equipment training opportunities (Kawarazuka et al. 2018). Finally, social and cultural norms can limit women's adoption of new technologies. Women typically have less agency to contribute to on-farm decision-making and must overcome beliefs that they are less technology savvy and more suited to the role of wife rather than farmer (Rola-Rubzen et al. 2020). With the rise of women's involvement in agriculture in developing economies (Paris and Rola-Rubzen 2019; Lastarria-Cornhiel 2006) it becomes increasingly important to develop gendered policy initiatives to encourage women's adoption of new technologies.

Choosing policies or approaches to address soil constraints

Pannell (2008) suggests that an appropriate or effective policy response to encourage adoption of environmentally beneficial on-farm practices depends on the degree of private costs or benefits the practice provides to the landowner, as well as the public benefits or costs it imposes on others. Many practices that improve soil quality are primarily associated with private benefits to landowners. However, many of these practices also provide public benefits in the form of greenhouse gas emission reductions (Smith et al. 2008), or benefits to many other ecosystem services (Bartkowski et al. 2020).

When the public benefits from practices that improve soil quality are very high and the private benefits for the landowner are near to zero, Pannell (2008) suggests that positive incentives, such as financial or material subsidies, are preferred. If society will largely benefit from adoption, yet the landowner is indifferent or unwilling to adopt, it is worthwhile to provide a

small incentive to change behaviour. If society will benefit from adoption and the farmer will also incur a small benefit, extension or education can be a cost-effective and efficient means to encourage adoption. Negative incentives, such as taxes or regulations, are effective in discouraging the use of practices that provide a small benefit to landowners but incur a large cost to society. When private net costs are at a similar level to public net benefits, then technological development is suggested to find ways of achieving the environmental goal at a lesser cost to the landowner.

The framework proposed by Pannell (2008) provides a starting point in formulating a policy or program to improve soil constraints through influencing landowner behaviour, but the many social, cultural, political, and psychological nuances must also be considered. Negative incentives can, at times, be politically unpalatable, or may not be feasible to implement on smallholders farms whose incomes are already very low. Budgets or positive incentives are not always large enough to effectively encourage improved soil practices, so governments and agencies may fall short of their environmental goals (Boxall 2018). When a new practice is financially advantageous compared to the current practice, initial financial support or aid for trial periods can be necessary for smallholders, in addition to extension efforts (Gathala et al. 2021). Other relatively subtle factors, such as procrastination, can result in farmers making suboptimal production and soil management decisions (Duflo, Kremer, and Robinson 2011). However, by identifying these factors, they can be overcome with small interventions, such as offering free fertiliser delivery for a short period of time early in the growing season. For these reasons, more complex policy approaches, beyond standalone subsidies or education programs, are often required to reduce soil constraints.

Policy mixes can be useful in addressing the complexity of adoption across different practices and heterogeneous farmers (Prager et al. 2011). Some governments have cross-compliance requirements, in which farmers are required to adopt certain beneficial practices or abide by certain regulations in order to access other financial or material supports (Bartkowski and Bartke 2018; Rude and Weersink 2018). Extension programs are often paired with financial or material incentives to improve the adoption rates of farm-level practices that enhance soil quality (Gathala et al. 2021; Rollins, Simpson, and Boxall 2018). Other programs have focused on engagement, extension, and positive incentives with all landowners in a region in order to effectively shift group norms and ensure widespread adoption of new practices (Boxall 2018).

To effectively improve soil quality and address soil constraints, multi-faceted social science research is recommended to understand the economic,

social, psychological, and cultural factors that might act as barriers to adopting soil-enhancing practices. This research is necessary to understand whether information or education alone will ensure adoption in cases where on-farm net benefits are positive, or whether subsidies or other assistance will be enough to compensate for on-farm losses or uncertainty when public benefits are especially high. By conducting surveys and interviews that focus on how farmers perceive new technologies, with an emphasis on how their cultural, economic, social, and psychological traits influence their perception, potential barriers to adoption can be identified and strategies to address them evaluated. Combining this research with field trials is especially useful, so that researchers may better understand how different types of people interact with and perceive new agricultural technologies, and ultimately how and why people decide whether to adopt soil enhancing practices.

Areas requiring future research

The effects of soil constraints on farmers' livelihoods, the environment, and communities are becoming more severe, especially in developing countries where institutions are weak and the unintended negative effects of technical progress are more pronounced (Ananda and Herath 2003). While these impacts are well established in the literature (Lahmar 2010; Orton et al. 2018; Stevens 2018; Wani et al. 2017), much of these studies are discipline-focused and there is a need to shift towards a cross-disciplinary and issue-driven collaborative research approach (Adewopo et al. 2014). Achieving zero hunger, food security and nutrition is an important sustainable development goal linked to soil constraints, since land food production is the major source of calorie intake (Pimentel 2006). Hence, there is a need to understand how to achieve adequate food and better nutrition while preventing soil degradation, especially with an increasing population and changing climate.

In terms of ecosystem roles, soil constraints also affect supporting, provisioning, regulating, and cultural services (Dominati et al. 2014; Haygarth and Ritz 2009). More specifically, soil constraints affect nutrient cycling, water purification, soil organic matter and structure, carbon sequestration and greenhouse gas emissions (Breure et al. 2012) as well as soil fertility, adaptability and resilience, buffer and reaction function, biodiversity and habitat provision, disease suppression and pest resistance, and physical structure (Faber and Wensem 2012). However, there is a need to apply an economic valuation of these services beyond the farm level, which can inform the public and policy makers of the crucial role of soil

ecosystem and agro-ecosystem services to the community and society in general.

Lastly, soil management practices can be improved further to become more gender-sensitive, as women's decisions to adopt these practices are constrained by their tenure status and access to off-farm income (Quisumbing and Pandolfelli 2010), and their knowledge is influenced by the gendered division of labour (Christie et al. 2016). Thus, adoption studies of soil management practices disaggregated by gender are important when designing more gender-sensitive farm management practices to combat soil degradation. These areas of future research provide direction for ongoing discussions of the critical socio-economic factors that are heavily impacted by soil constraints.

Conclusion

This chapter has highlighted the poverty, food security, and social consequences of soil constraints, and their micro and macroeconomic consequences at farm-level, regional, national, and multinational scales. Examples of the different methods employed to capture these impacts have been provided, along with snapshots of specific findings and figures representing the range of extent of these impacts.

Improvements in yield and cropping intensity will be required to meet the growing demand for food. This will mostly have to come from developing countries, but unless new approaches are found that improve soil attributes and overcome soil constraints without degrading soils, these solutions will not be effective. More work is required on crop tolerances and yield responses to multiple soil constraints to find solutions and assess the economic and social effects of soil quality changes.

Soils provide a wide range of ecosystem goods and services, and soil constraints and soil degradation can severely limit efforts to reduce poverty and malnutrition and improve food security. Reductions in soil quality can have negative impacts on rural viability, leading to increased outmigration and conflict. Assessing the economic consequences of soil constraints and degradation is complicated and difficult. However, they have been found to cause major reductions in regional or national GDP and in land values.

Increased pressure to increase yields and cropping intensity has the potential to compound these effects, with climate change adding to the difficulties of managing and addressing soil constraints. Efforts to change behaviour will need to use approaches that involve targeted communities in the development of technological solutions to their problems to ensure solutions are consistent with cultural norms, resources, and capital. The use

of gendered approaches when undertaking these developments will be important.

Policy responses should also consider the relative levels of public and private benefit and the costs of proposed changes and use a combination of mechanisms. Multi-faceted social science research is also required to understand the economic, social, psychological, and cultural factors that might influence the efficiency and effectiveness of these policy mechanisms. This research needs to use transdisciplinary and participatory methodologies that address the technical, social, economic, and environmental constraints to expanding production without leading to a collapse of soil resources.

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CHAPTER 24

SOIL CONSTRAINTS TO CROPPING: ENVIRONMENTAL CONSEQUENCES

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Abstract

When examining the impact of constraints in cropping soils, we often consider only how they influence crop growth, with this influencing yield and profitability. However, it is imperative to also consider whether soil constraints within cropping systems have adverse effects on the broader environment, with this being the focus of the present Chapter. First, we give consideration to erosion, which not only constrains production in cropping systems, but also causes adverse environmental outcomes due to sedimentation (physical pollution of the environment) and eutrophication (chemical pollution). Next, we examine the loss of soil organic matter, which in cropping systems causes a decrease in soil fertility, but more broadly results in the release of profound quantities of carbon dioxide as a greenhouse gas. The toxicity of pollutants within cropping soils is also examined, including the overapplication of N and P fertilisers to cropping soils, which causes eutrophication, and also the application of pesticides, which often accumulate in soils and cause degradation of the surrounding landscape. Finally, we consider soil acidity, which when ameliorated using lime or dolomite, results in the release of substantial carbon dioxide to the atmosphere as a greenhouse gas. It is hoped that the discussion of how soil constraints in cropping systems also adversely impact upon Earth's environment will encourage the inclusion of such impacts in decision-making processes to ensure better outcomes for all stakeholders.

Keywords: Acidity, environmental degradation, erosion, soil organic matter, pollution.

Introduction

Soil constraints to cropping can result from on-site sources, such as the over-use of N fertilisers causing *in situ* acidification of soil, but they can also result from off-site sources, such as the deposition of acidity from acid rain caused by the release of nitrogen and sulfur oxides into the atmosphere from industrial activities. In a similar manner, some of the adverse effects of soil constraints are observed on-site, but other adverse effects are observed in the broader environment.

In earlier chapters of this book, the main focus was to understand how the soil constraints found within in cropping systems impact those specific systems themselves, with a clear focus on their impact on agricultural (crop) production. However, in this chapter, we focus more broadly on how soil constraints within cropping systems can also sometimes impact upon the broader environment. Often, such consequences are overlooked, but it is increasingly clear that we need to explicitly account for such impacts (Kopittke et al. 2021). Specifically, in the present chapter we give consideration to (i) erosion, which results in adverse environmental outcomes through sedimentation and eutrophication; (ii) the loss of soil organic matter and the decrease in soil fertility, which is a major contributor to global greenhouse gas emissions; (iii) soil pollutants, which cause toxic effects within the environment; and (iv) soil acidity, the amelioration of which using lime results in the release of greenhouse gases. Within this chapter, it is not possible to provide a detailed mechanistic description of how these soil constraints impact upon the environment. Rather, we aim to introduce the environmental issues that arise from soils in cropping systems, with the reader referred to more detailed literature reviews for further information.

Erosion

It is estimated that global soil loss due to water erosion is 20-30 Gt per year (FAO and ITPS 2015). Indeed, erosion rates from conventionally tilled agricultural fields are 10-100 times higher than the rate at which soil forms (Montgomery 2007). As discussed in Chapter 5, erosion results in a clear, long-term economic cost, not only due to lost productivity but also due to the loss of nutrients that must eventually be replaced through fertilisers, being equivalent to the economic loss of US\$33-60 billion for N and US\$77-140 billion for P (FAO and ITPS 2015). However, it is also clear

that erosion from cropping systems causes substantial offsite environmental harm, with this being the present focus. We focus on erosion (as a constraint to cropping) that also contributes to the physical pollution of the environment (sediment) as well as to the chemical pollution of the environment (eutrophication).

Sedimentation – Physical pollution from soil erosion

Soil sediments have two broad types of negative physical effects on receiving waters in the broader environment, being (i) an increase in turbidity, and (ii) a physical disruption of the hydraulic characteristics of the water body. Firstly, the increase in turbidity that is caused by the sediment limits the penetration of sunlight into the water column. This reduces primary productivity in the water body by inhibiting the growth of aquatic plant species and algae, with these decreases in primary productivity resulting in a negative cascade of effects due to depleted availability of food for consumers, resulting in mortality, reduced physiological function, and avoidance (Henley et al. 2000). A particularly pronounced example of the adverse effects of high turbidity is its impacts on coral reef systems. Within the Great Barrier Reef Lagoon (Australia), whilst climate change poses the most significant threat, both N pollution and the discharge of sediments are also major threats, with the suspended solids reducing light for seagrasses and corals (Great Barrier Reef Marine Park Authority 2019). As a result of these threats, the conservation outlook for the Great Barrier Reef was listed as being “critical” in the most recent (2020) assessment by the International Union for Conservation of Nature (IUCN), which is an advisory body to the World Heritage Committee.

Sediments can also cause a physical disruption of the hydraulic characteristics of the water body, resulting in a range of socioeconomic impacts, such as disruption of river transport, siltation of water reservoirs, and increased flooding of human settlements.

Currently, it is estimated that mean rates of continental denudation (river sediment fluxes) are 21 Gt y^{-1} (Wilkinson and McElroy 2007). Of this, the highest 10% of the Earth’s surface accounts for 83% of the total global river sediment flux (Wilkinson and McElroy 2007). Thus, it is clear that erosion from the highest portions of the Earth’s surface still dominate global river sediment fluxes. However, it must be noted that human activities have resulted in a considerable increase in continental erosion in the lower, flatter areas of land due to agriculture. For example, global analyses have identified that land cover change (which is almost entirely due to agriculture) is now the main driver of soil erosion in at least 70% of watersheds, with a significant

shift to human-driven soil erosion in the last 400 years (Jenny et al. 2019; Walling 2006). Indeed, 83% of cropland erosion occurs in the lower 65% of the Earth's surface (Wilkinson and McElroy 2007). Thus, whilst river sediment fluxes are heavily dominated by erosion from the highest portions of the Earth's surface, agriculture (with associated land use change) now dominates erosion from the lower lands. This erosion from agricultural lands not only reduces agricultural productivity, but it also causes substantial offsite effects due to the movement of sediment into environments that do not naturally receive high sediment loads, such as coral reef systems.

Eutrophication – Chemical pollution from soil erosion

Eutrophication is the nutrient enrichment of water bodies resulting in excessive growth, with nitrogen (N) and phosphorus (P) being the two nutrients of most concern (Smith and Schindler 2009). Eutrophication of surface waters is of concern not only for freshwater systems, but also because there is clear evidence that when critical thresholds for P inflow into the oceans are transgressed, a global anoxic marine event can occur, resulting in mass extinctions of marine life (Rockstrom et al. 2009). Indeed, eutrophication of both freshwater and marine systems have a wide range of adverse effects, including reductions in species diversity, increased incidence of fish kills, loss of coral reef communities, and shifts to bloom-forming algal species (Smith and Schindler 2009). In the USA alone, it is estimated that eutrophication causes \$2.2 billion in damage annually (Dodds et al. 2009).

The nutrients that move into waterbodies originate from a diverse range of sources. Our focus here is on the role of soil constraints in cropping systems in contributing to eutrophication within the broader environment. In this regard, the erosion of soil from cropping systems is of critical importance. Indeed, once eroded soil is deposited in water bodies, the nutrients that are transported within the soil are a critical contributor to eutrophication. This is especially important for P, which binds strongly to soil particles and is transported during erosion. In contrast, N is highly mobile within soils and tends to move into water bodies due to leaching (c.f. erosion). Thus, for the purposes of the present discussion, which considers soil constraints, we focus on the contribution of erosion to eutrophication. Although soils are a major contributor to N eutrophication due to the inefficient use of N fertilisers, this is beyond the scope of this discussion focussing on soil constraints.

For P, total global anthropogenic contribution to freshwater systems from agriculture is estimated to be 558 Gg y⁻¹, with this largely being due

to erosion (Mekonnen and Hoekstra 2018). This compares to the total global anthropogenic inputs from all systems of ca. 1.4 tg y^{-1} (Mekonnen and Hoekstra 2018). Hence, the contribution of erosion of agricultural soils to total anthropogenic P release to freshwater systems is ca. 38%. Thus, the erosion of soils is not only an important constraint for agricultural production, but it is also an important cause of the movement of nutrients (especially P) into waterbodies where it contributes to eutrophication. Furthermore, given that industrial and domestic sources of P are generally point sources whilst agriculture (primarily soil erosion) is a diffuse source, reducing the magnitude of the contribution from agricultural production is generally more difficult than reducing contributions from point sources (for example, through tertiary treatment of wastewater).

Loss of soil organic matter and declining soil fertility

It is well known that soil organic matter (SOM) plays a critical role in maintaining the chemical, biological, and physical fertility of soils. For example, SOM has numerous functions, such as increasing the cation exchange capacity (CEC), serving as a slow-release form of nutrients (particularly for N, P, and S), increasing soil structure and porosity, as well as improving the water holding characteristics of soil. Not only is SOM critical for soil functioning, but it is also a major global reservoir of C given that SOM contains an average of 58% C. Indeed, soils store ca. 2344 Pg of organic C (OC) within the surface 3 m and ca. 1500 Pg C in the surface 1 m (Scharlemann et al. 2014), and hence soil OC stocks exceed the amount of C stored in the atmosphere (875 Pg C in 2019) and vegetation (600 Pg C) combined.

Loss of soil organic matter and greenhouse gas emissions

Although the soil is a major global reservoir for C, its use for productive agriculture results in a pronounced loss of this SOM. Long term cropping, for example, decreases soil OC stocks by 30-60% – in two global meta-analyses, Kopittke et al. (2017) reported a median decrease of 43%, whilst Guo and Gifford (2002) reported a mean value of 42% with 95% confidence intervals of 34-49%. As discussed in previous chapters, this represents a serious constraint for agricultural production, with the loss of OC having important effects on soil fertility by decreasing soil nutrient concentrations, reducing water holding capacity, and increasing bulk density (decreasing porosity). However, this loss of soil OC also has a profound impact in the broader environment, with this being the presant focus.

The release of CO₂ from soils due to the mineralisation of OM following land use change has resulted in the release of an estimated 133 Pg of C, with this primarily occurring in the last 200 y (Sanderman, Hengl, and Fiske 2017). In contrast, it is estimated that the total global cumulative release of CO₂ due to fossil fuel consumption, flaring, cement production, and land-use change between 1750 and 2010 was 540 Pg of C (IPCC 2014). Thus, it is estimated that the mineralisation of soil OC due to land use change has contributed ca. 25% of the total CO₂ emissions over the last 200 y (Kopittke et al. 2021). It is also possible to estimate the contribution of CO₂ derived from soils to the total radiative forcing caused by the emissions of all greenhouse gases. Specifically, the total increase in radiative forcing due to greenhouse gas emissions is +2.8 W m⁻², with CO₂ accounting for +1.8 W m⁻² of this total. Thus, given that the release of CO₂ from soils due to land use change accounts for 25% of total CO₂ emissions, it is estimated that the release of CO₂ from soils alone accounts for 16% of the total cumulative increase in radiative forcing due to anthropogenic emissions of all greenhouse gases.

Soil pollution

Soil pollution can be defined as the “presence of a chemical or substance out of place and/or present at higher than normal concentration that has adverse effects on any non-targeted organism” (Eugenio, McLaughlin, and Pennock 2018). Many pollutants in soil result entirely from anthropogenic processes, although some [such as copper (Cu), for example] occur naturally in soils but accumulate to elevated levels due to human activities. Broadly, pollutants originate from either point or diffuse sources. Point-sources include mining and smelting operations, industrial sites, and landfill. At such sites, the source of the pollution tends to be quite concentrated, and the source is generally easily identified. In contrast, diffuse pollution results in pollution across a broader portion of the landscape, with the source of the pollution sometimes more difficult to identify. A range of activities can cause diffuse pollution, including agriculture (pesticides, fertilisers, and animal manure) and the land-application of biosolids and waste products. Especially for diffuse pollution, the effects on the broader environment can be substantial, but are more difficult to quantify and manage.

Pollutants can be classified in a range of ways, but generally, are classified as being either inorganic or organic. Inorganic contaminants include both metals / metalloids [elements such as lead (Pb), Cu, mercury (Hg), cadmium (Cd), and arsenic (As)] and non-metals (elements such as N and P) (Swartjes 2011). Many of these inorganic pollutants are also essential nutrients for

plant and animal growth [such as N, P, Cu, Zinc (Zn), and Nickel (Ni)]; however, their accumulation to elevated levels is of concern due to their adverse effects both within the cropping system itself, but also more broadly within the environment. Organic pollutants are also highly diverse but can be classified broadly as being chlorinated compounds (such as DDT, polycyclic aromatic hydrocarbons, polychlorinated biphenyls, and dioxins) or non-chlorinated compounds (such as benzene).

Within cropping systems, pollutants generally result from agricultural production itself, with these being the systems that we focus on presently. These pollutants not only potentially reduce food production (see Chapter 10), but also can result in movement of the pollutant through the food chain thereby threatening human and environmental health, with the latter being the focus of the present chapter.

Inorganic pollutants

For the inorganic pollutants in cropping systems, the main concerns are N and P (non-metals) as well as Cd, Cu, As, Pb, and Hg (metals / metalloids). These inorganic pollutants result from a diverse range of sources, such as the use of fertilisers, pesticides, the addition of solid wastes, irrigation with contaminated waters, and the application of livestock manures.

Globally, the largest environmental damage from pollutants in cropping systems results from over-application of N and P (Steffen et al. 2015; Kopittke et al. 2021). For N, the movement of excess concentrations of this nutrient into the environment causes eutrophication of water bodies, it is an important contributor to global climate change through the release of N_2O as a greenhouse gas, and also contributes to the pollution of drinking waters due to the leaching of NO_3^- . Agricultural systems are now heavily reliant on inorganic N fertilisers, with 109 Tg N produced for N fertilisers in 2020 (FAO 2019). Whilst these N fertilisers are needed to feed approximately half of the world's human population (Erisman et al. 2008), their use is currently highly inefficient – only 47% of the N applied is used by plants, with the remainder lost to the broader environment (Lassaletta et al. 2014). Thus, whilst it is not possible to stop using N fertilisers whilst still feeding the human population, their efficiency of their usage could be increased markedly, thereby reducing adverse off-site environmental impacts. In addition, the application of animal manures to soils can contribute to the movement of N and P into the surrounding environment.

Excess N and P are both major contributors to eutrophication of water bodies, as discussed earlier for P. Whilst the movement of P into water

bodies is largely by erosion, because N is highly mobile within the soil matrix, the N largely moves into water bodies via leaching. However, not only does N contribute to eutrophication, but the conversion of soluble N from fertilisers to N_2O contributes to global warming. In this regard, it is known that soils account for approximately 60% of total global N_2O emissions (Tian et al. 2019), with total N_2O emissions accounting for 6.1% of the total anthropogenic increase in radiative forcing due to global greenhouse gas emissions (Kopittke et al. 2021). Thus, the use of N fertilisers, including overuse and subsequent pollution, is also an important contributor to climate change.

Although other inorganic pollutants (in addition to N and P) can also accumulate within cropping soils, they tend to have adverse impacts directly within those agricultural systems, or for human health, rather than having broader impacts within the environment. For example, Cu can accumulate in cropping soils due to its use in fungicides, with this Cu having potentially adverse effects at high concentrations on plant root growth in such soils (Wightwick et al. 2008) given that Cu is comparatively immobile in soils and is toxic to roots. Similarly, contaminants such as Cd can accumulate in cropping soils through application of phosphatic fertilisers (McLaughlin et al. 1996) or through irrigation with contaminated waters (Wang et al. 2019). However, given that Cd is readily translocated from the roots to the edible portions of the plants, the accumulation of Cd in cropping soils is of concern due to the movement of this toxic element into human foods.

Organic pollutants

Agricultural production is an important source of organic pollutants globally, especially due to the application of pesticides. Indeed, the intentional release of pesticides into cropping systems, particularly since the Green Revolution, has resulted in widespread pollution and broad environmental harm. There are a wide variety of synthetic organic compounds used in pesticides, which differ in chemical structure and properties, including organochlorine compounds (such as DDT), organophosphorus compounds (such as parathion and chlorpyrifos), and chlorphenoxy compounds (such as 2,4-D) (Eugenio, McLaughlin, and Pennock 2018). Although a detailed discussion of organic pesticides and their environmental impacts is beyond the scope of the present chapter, the topic is briefly introduced here.

Of the 2 million t of pesticide applied globally each year (Sharma et al. 2019), a substantial proportion accumulates in the soil. From the soil, these pesticides can have a wide range of impacts on non-target organisms, not only within the soil (such as earthworms or the soil microbial community),

but also more broadly in the environment due to the movement of pesticides into other environmental compartments, such as by leaching to groundwater, root uptake by plants and movement into the food chain, loss to the atmosphere, and transport to surface waters.

Once within the environment, the behaviour of any pesticide depends upon a range of properties, including its solubility in water (which influences leaching and mobility in the environment), its partition coefficient (which influences its potential to bioaccumulate), and its reactivity (which influences its rate of degradation). Pesticides in the environment can have a wide range of impacts, including on soil microorganisms, plants, aquatic organisms, insects, birds, and mammals (including humans). The most well-known example of the adverse effects of pesticides in the environment is of DDT, where the movement of DDT into the environment caused eggshell thinning across a broad range of species, including the bald eagle in the USA. However, many other examples are available, including fish kills in the USA where millions of fish are lost (Cottam 1965).

Thus, whilst our production of food is heavily reliant on the use of pesticides, these compounds have substantial adverse effects on the environment resulting from their initial accumulation in cropping soils. In order to reduce these adverse effects, new approaches are needed to reduce our reliance on pesticides as well as to develop compounds that are less harmful to the environment.

Soil acidity

Soil acidity is an important constraint to crop production – it is estimated that up to 30% of the ice-free topsoils worldwide are acid, and up to 75% of subsoils (FAO and ITPS 2015). Within cropping soils, the adverse effects of soil acidity are generally restricted to the crop being grown (such as Al toxicity reducing root growth), with only limited effects on the broader environment. Although highly acid soils (for example, acid sulfate soils) can result in the mobilisation of toxic concentrations of metals into the environment, this situation is not observed in cropping systems due to the greater sensitivity of plant growth – amelioration of soil acidity for crop growth is required at pH values higher than that required to allow mobilisation of high concentrations of these metals.

Nevertheless, there is one aspect of soil acidity that does contribute broadly to environmental degradation. Specifically, the addition of liming materials, including lime (CaCO_3) and dolomite ($\text{MgCa}(\text{CO}_3)_2$), to ameliorate soil acidity results in the release of CO_2 as a greenhouse gas. The emissions of CO_2 from the soil-application of lime vary depending upon soil

properties and climate (Shoghi Kalkhoran et al. 2019). In the USA, for example, it is estimated that the addition of lime and dolomite results in the release of 9 Tg of CO₂ annually, contributing 1.7% of total agricultural greenhouse gas emissions and 0.13% of total greenhouse gas emissions in the USA (West and McBride 2005).

In addition to the release of CO₂ from liming, the increase in soil pH caused by liming can also influence the release of other greenhouse gases including N₂O and CH₄ by altering biological processes in the soil (Page et al. 2009). However, changes in N₂O and CH₄ caused by increases in pH from liming appear to be varied, depending upon site-specific conditions (Page et al. 2009).

Thus, although soil-application of lime is important to ameliorate acidity and improve crop yield, it also results in emissions of CO₂. In this regard, optimising N fertiliser applications is essential given that over-application of N fertilisers not only causes acidification (requiring addition of lime and causing emissions of greenhouse gases), but the addition of N fertilisers also results in the release of N₂O, which is itself a greenhouse gas.

Conclusions

The aim of this chapter was to change the focus from examining how soil constraints impact upon plant productivity to consider how these same constraints also have broader environmental impacts beyond the cropping system of interest. In this regard, it is clear that erosion, loss of organic matter, pollution, and soil acidity all alter environmental functioning, including through the release of greenhouse gases, eutrophication of water bodies, exertion of toxic effects in unintended organisms, and causing physical pollution of water bodies. It is hoped that the discussion in the present Chapter will broaden future discussions regarding the management of soil constraints to focus not only on the cropping system of interest, but also to give consideration to the surrounding environment.

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CHAPTER 25

SOIL CONSTRAINTS TO CROP PRODUCTION: FUTURE DIRECTIONS

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Abstract

Increasing agricultural production to satisfy the demands of a growing population, while simultaneously dealing with climate change and increasing soil degradation, is a major challenge. To satisfy our appetite for cropping products without dramatically increasing the land area used for agriculture, it will be essential to intensify agricultural production and reduce crop yield gaps worldwide. An essential component for achieving this is to improve our management of soil constraints such as acidity, salinity, sodicity, erosion, compaction, and declining biological diversity. Because constraints vary across the landscape and down soil profiles, improvements in our ability to identify the location and nature of soil constraints are required to improve management efficiency. The creation of constraint identification tools using modern data analysis techniques to harness the large amount of information available from remote sensing platforms and other sources may be a particularly fruitful area of research. Once identified, soil constraints can be managed via a) amelioration to modify the soil and reduce or remove constraints; b) agronomic adaptation to increase productivity or profitability in the presence of constraints; and c) land use change, which may be required when constraints are so severe that it is not logistically or economically possible to sufficiently ameliorate them to improve production. While we have made large advances in our understanding of how to manage constraints, further refinement is required, particularly of how to manage multiple constraints, constraints occurring in

the subsoil, and how to develop integrated systems that are adapted to local climates, soil types, crop species, and communities and are thus economically and logistically attractive to farmers. To achieve this intensification will require a multidisciplinary approach that links the skills and knowledge of farmers with specialists in soil science, agronomy, genetics, ecology, engineering, computer science, data analysis, economics, and extension.

Keywords: Amelioration, agronomic adaptation, remote sensing, subsoil, sustainable intensification.

Introduction

By 2050, the global population is predicted to reach 9.7 billion (United Nations 2019), an increase of approximately 25% on that present in 2021. This population growth, combined with increasing demand for biofuel and meat/dairy products, is expected to significantly increase demand for cropping products over the next 30 years, particularly in Africa and South Asia where the greatest population growth is expected to occur (FAO 2017). To satisfy this demand, we have the option of either expanding the area of land used for agriculture, or we can increase production from existing land resources. As opportunities for expansion are both limited and undesirable if the world's remaining biodiversity is to be preserved, the majority of the production increase must come from the intensification of existing agricultural land. This represents a significant challenge for world food production and sustainable development.

Since the 1960s, improvements in crop genetics and the widespread use of inputs to reduce the environmental limitations to production (e.g., irrigation, fertilisers, pesticides) have led to large increases in yield and agricultural output while allowing the total area of land used for agriculture to remain relatively stable. For example, in 2018, world wheat production had increased by over 400% relative to 1961, despite only a ~4% expansion in the land area on which wheat was grown (FAO 2020). However, in some resource poor regions where farmers lack access to agricultural inputs, infrastructure, markets and extensions services, yield gains have been low e.g., southern Africa (Grassini, Eskridge, and Cassman 2013). In addition, declines in the rate of crop yield increase are now being observed in many regions worldwide (Grassini, Eskridge, and Cassman 2013; Brisson et al. 2010; Ray et al. 2012; Lin and Huybers 2012; Peltonen-Sainio, Jauhiainen, and Laurila 2009), and the further increases in productive capacity that will be required to achieve future food security are threatened by climate change, poor farming practices, and degradation of the soil resource (Hochman,

Gobbett, and Horan 2017; Brisson et al. 2010; Lobell, Schlenker, and Costa-Roberts 2011; Kopittke et al. 2019).

A combination of approaches will be required to lift yields in resource-poor regions and reverse the emerging trends in yield stagnation to sustainably increase global food production. Continued improvements in crop genetics and the optimisation of markets and economic signals to favour efficiencies in production are expected to play an important role. In addition, strategies to improve yield in underperforming locations will also be essential (OECD and FAO 2017; Ray et al. 2012). In many regions a gap exists between the water-limited yield potential of a crop and the actual yield achieved (Foley et al. 2011; George 2014). This gap can exist due to suboptimal crop management practices, the lack of access to resource inputs (e.g., fertiliser, equipment), and the presence of soil constraints to production, such as those illustrated in Figure 25.1. These factors not only decrease production and farmer profitability, but are also often responsible for external environmental impacts, such as erosion, increased greenhouse gas emission, ground and surface water salinisation, eutrophication, and chemical contamination (Rachel et al. 2010; Foley et al. 2011).

The yield gaps that exist due to soil constraints can be particularly important to address if increases in crop production are to be achieved. A soil constraint can be defined as any soil characteristic that restricts plant growth and lowers crop production. Soil constraints can be physical, chemical, or biological (Figure 25.1) and can be a naturally occurring soil characteristic (e.g., nutrient deficiencies due to low soil cation exchange capacity, naturally occurring salinity or sodicity) or develop due to poor management practices (e.g., erosion, soil fertility decline due to extractive agriculture, irrigation-induced salinisation/sodification, acidity due to poor N management). It has been estimated that ~76% of agricultural land globally is currently impacted by one or more soil constraints, and that the areas affected are likely to increase by five to seven million hectares annually (Bot, Nachtergaele, and Young 2000). The successful management, and/or prevention of soil constraints will thus be an important component of reducing yield gaps. Recognising when constraints are so severe that cropping is not viable is also important as the reversion of highly marginal land with very low levels of productivity back to native vegetation or more perennial farming systems (e.g., agroforestry) can often reduce environmental impacts (Cassman et al. 2003; Kopittke et al. 2019). Indeed, agricultural intensification on more productive areas can often remove the pressure to farm highly marginal areas and allow these to be retired (Foley et al. 2011). This chapter will examine the recent

advances in our understanding of soil constraint management and their potential to contribute to improvements in cropping systems. Those areas most needed for future research area also discussed.

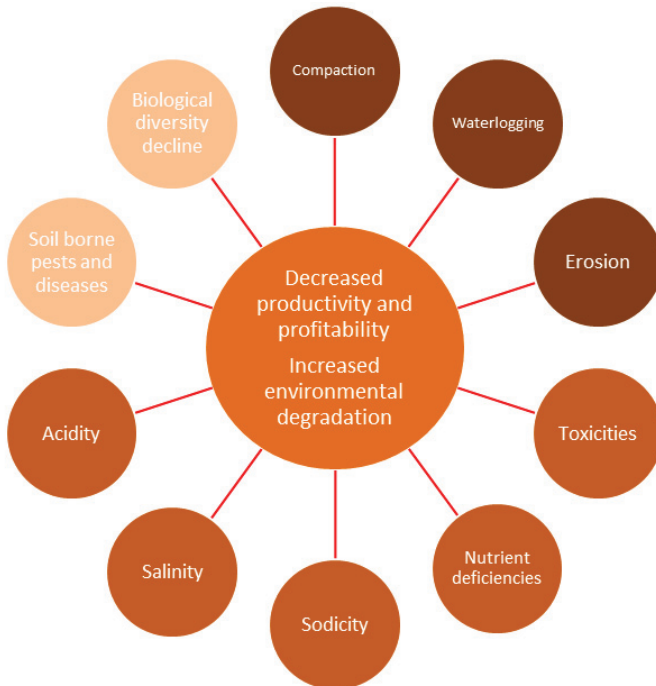


Figure 25.1 Examples of common physical (dark brown), chemical (light brown), and biological (pink) soil constraints and their impact

Identification of soil constraints

The first step towards successfully combating soil constraints is to identify the constraints present at a location. Soil constraints tend to occur heterogeneously across landscapes and down soil profiles (Figure 25.2), and accurate spatial identification is essential to increase the efficiency and effectiveness of any management approach selected. Multiple constraints may also occur at the same location, and it is important that all constraints are identified and treated as maximum yield increase cannot be expected before this occurs.

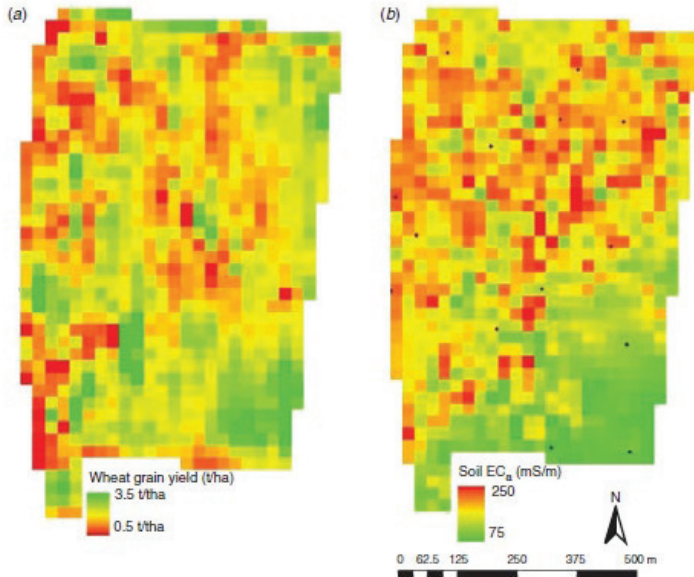


Figure 25.2 A wheat yield map (a) and EM38 survey (b) of a cereal cropping paddock in southern Queensland, Australia. Note the high degree of spatial variation in EC_a across the paddock and the correlation between EC_a and yield. The EM38 measurements are highly correlated with indicators of salinity, such as soil electrical conductivity, soil chloride, and exchangeable sodium percentage. Diagram modified from Dang et al. (2010)

The traditional identification of soil constraints relies on soil survey techniques that physically extract soil samples across a landscape and then use field or laboratory analysis to characterise the soil and identify yield limiting characteristics. However, while these techniques can be effective, they are also time consuming, labour intensive, and expensive to conduct, particularly at a high degree of spatial resolution over large areas, and particularly where constraints occur in the subsoil. Because of this, high quality soil survey measurements are often unavailable to identify soil constraints and the expense of conducting such surveys means they are beyond the reach of many farmers to conduct individually.

In recent years, developments in proximal and remote sensing have allowed significant advances in soil constraint identification and mapping to occur. These techniques allow the collection of larger amounts of spatial data using cheaper, simpler, and less laborious techniques. Proximal

techniques, for example, use field-based sensors to obtain signals from the soil that correspond to physical and chemical measures, which can then be related to soil properties (Grunwald, Vasques, and Rivero 2015; Viscarra-Rossel et al. 2011). Many devices can be used for proximal soil sensing, including, ion-sensitive field effect transistors to measure soil pH and soil nutrients, portable visible near-infrared spectrometers to measure organic carbon content and mineral composition, ground-penetrating radar to measure soil water content, and electromagnetic induction (EM) to measure apparent soil electrical conductivity (EC_a) (Viscarra-Rossel et al. 2011). Electromagnetic induction (particularly EM38 and Dual EM38) is a particularly useful proximal technique that has been successfully used to identify and map variations in soil salinity and sodicity at the paddock level, including when present in the subsoil (Dang et al. 2011; Corwin and Lesch 2003).

The use of remotely sensed data from satellites, aircraft, or unmanned aerial vehicles is also an emerging technique that can be used to identify and map soil properties and plant growth to assist in soil constraint identification (Grunwald, Vasques, and Rivero 2015; Weiss, Jacob, and Duveiller 2020). One of the most reliable ways of determining the location of constraints is to identify consistently low yielding locations (i.e., season after season). Remote sensing of visible and near-infrared wavelengths can be used to derive vegetation indices that are well correlated with plant growth and yield over a range of spatial scales (paddock to region) (Chlingaryan Sukkarieh, and Whelan 2018; Grunwald, Vasques, and Rivero 2015). This allows the user to characterise plant growth over time to identify consistently low yielding areas. This can then be combined with information from other remote sensors, proximal sensors, and with measured data from physical soil samples to relate the poor yielding areas with soil properties and identify the characteristic/s limiting yield. Recent advances in computing power, data storage capacity, and machine learning approaches (e.g., artificial neural networks, regression trees, random forests, Bayesian networks), are also increasingly allowing the processing of large amounts of data and the detection of non-linear relationships between yield and soil characteristics. This can improve the detection of relationships and aid the identification of the boundaries between low and high yielding areas (Sishodia, Ray, and Singh 2020; Chlingaryan, Sukkarieh, and Whelan. 2018; Grunwal, Vasques, and Rivero 2015).

Gathering and manipulating remote sensed imagery requires a high degree of specialist knowledge and computing power, which presents an impediment to the wide-spread use of this technique by individual landholders, particularly in resource poor regions. The recent development

of web-based tools that automate image analysis and allow the user to extract information for their location represents a significant advancement in the identification of constraints. One example of such a web-based tool is provided below via a case study of our development of “Constraint ID”. These tools can be utilised by landholders themselves, or where farmers do not have sufficient access to technology, by extension agencies, which can use them to help guide farmers in the identification of constraints.

Case Study: Constraint ID – a web-based platform for soil constraint identification

Better knowledge and understanding of the variation of past crop yields (both spatially across a paddock, and temporally from season to season) can help growers make better management decisions to improve yields in the future. Remote-sensing data compiled from earth-observing satellites provide a valuable information source about past crop yields, allowing us to look through multiple years of imagery for consistent patterns through time – that is, spatial patterns of yield variation that repeat season after season. Such consistent patterns (within a field that has been managed uniformly spatially) might imply the presence of a soil constraint limiting yields in parts of the field.

The Constraint ID software tool (<https://constraintid.com.au/>) has been developed to help growers identify constrained areas within fields. It has been developed for use in the Northern Region of Australia’s grain cropping area (Queensland and New South Wales), where winter wheat cropping is dominant. The tool allows the user to enter the boundary for a field of interest on an interactive map, and historical data from the Landsat series of satellites (data from Landsats 5, 7, and 8) for the field are then analysed. The Landsat data were selected for the Constraint ID tool for a number of reasons, but perhaps the most important were the long history of data (we included data from 1999 to the past year) and the fact that the data are freely available. When the user has entered their field boundary, the software tool performs a number of steps for each winter-wheat growing season from 1999 to the past year. It:

1. Forms a time series of field-median enhanced vegetation index (EVI) values and finds the date of the maximum value.
2. Calculates a number of heuristics from this time series to make an initial determination whether or not the field was cropped that year (the user has the opportunity to manually over-ride this initial determination).

3. Retrieves all EVI images from within ± 30 days of the date of maximum EVI with sufficient spatial coverage of the field.
4. Ranks [standardised to the range (0,100)] the pixels within each of these images and forms a composite of these ranked EVI images to give a map of what we will call the “crop yield index” for the season.

The crop yield index doesn't show whether any particular year was higher or lower yielding than any other year; rather it is designed to represent the spatial variation of yield within each season. By looking at the mean of the crop yield index over a number of growing seasons (and its standard error), the tool can indicate whether there is evidence that certain parts of the field show *consistently* lower yields (according to the index) than other parts.

Screenshots from Constraint ID (Figure 25.3) illustrate results for a paddock from a farm near Biloela, central Queensland. All that the user has to do to get such an output is to register and then enter their field boundary by clicking on an interactive map. There appear to be some consistent spatial patterns in the maps, with the west side of the field relatively high yielding compared with the east side. The two maps in the lower panel confirm this consistent variation, with the red zone showing all of the pixels that were consistently below the 50th percentile of the crop yield index.

To help growers interpret results from the analysis of remote-sensing data, the tool allows the user to upload soil data they have collected from the field. The tool then compares the soil data that were collected from the high- and low-yielding areas to provide an indication of what soil constraints might be driving this variation. The measured soil data are also compared with critical values for each soil constraint (Page et al. 2021) through a simple “traffic-light” system, whereby measured values falling in the “red” zone on the soil profile plots indicate that there is potential for severe yield loss. Further screenshots from Constraint ID (Figure 25.4) illustrate sample locations from the paddock and the measured indicators of salinity (ECe), sodicity (ESP), and acidity or alkalinity (pH). These data show that the ESP below 60 cm is higher in the low-yielding than in the high-yielding areas and suggest that subsoil sodicity might be a driver of the consistent spatial patterns of yield variation in this case.

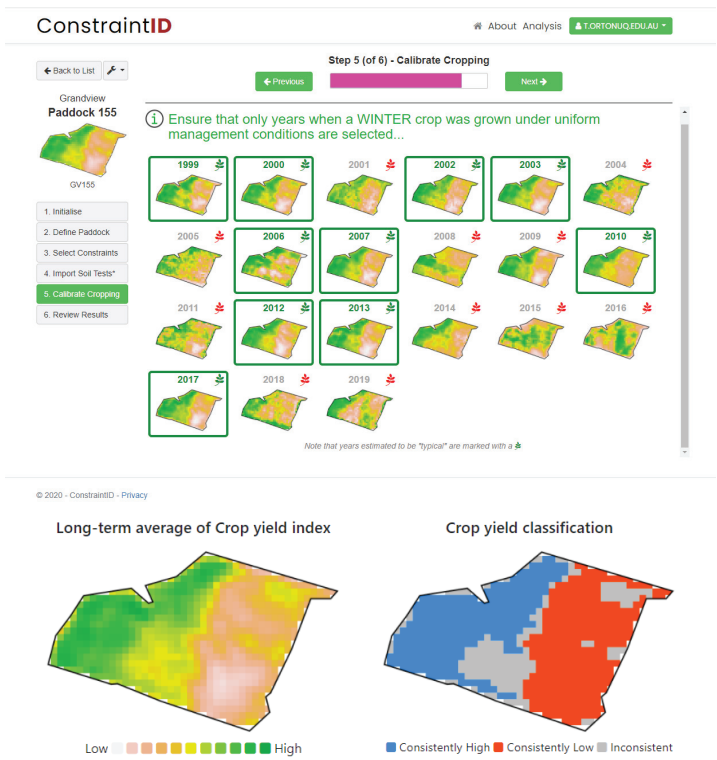


Figure 25.3 A screenshot from the Constraint ID web-based tool showing the crop yield index for the years 1999–2019 (green for high, brown and white for low). The years determined by the tool to be cropped are indicated by a green leaf. The maps in the lower panel show the long-term average of the crop yield index and a classification of pixels as consistently high or consistently low.

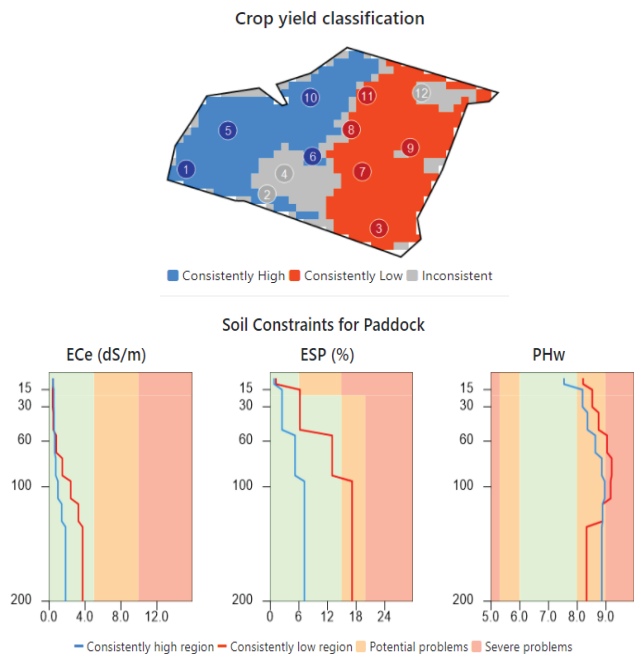


Figure 25.4 Screenshots from Constraint ID showing the locations of sampled soil locations with respect to the identified consistently high- and low-yielding areas, and the averages of measured soil constraints through the soil profile within these areas.

Soil Constraint Management

Once soil constraints have been successfully identified, it is then necessary to decide how to manage these to maximise agricultural production and farm productivity. There are three broad approaches that can be used to manage soil constraints: a) amelioration; b) agronomic adaptation; and c) land-use change.

Amelioration requires that some action or input be used to improve the soil condition to either remove or lessen a soil constraint. Examples of amelioration for common soil constraints are listed in Table 25.1 and can include:

- The addition of external inputs e.g., lime to neutralise acidity, fertiliser to reduce nutrient deficiency, organic matter to improve soil biological diversity and fertility.
- Physical manipulation of the soil environment e.g., cultivation to remove soil crusts or layers of high bulk density, drainage to improve waterlogging, bunds to reduce erosion.
- The growth of plant species with the ability to change the soil environment to alleviate constraints e.g., deep rooted perennial species to loosen compacted subsoils, salt tolerant plants to reduce soil salt concentrations, cover crops to reduce erosion and increase soil organic matter concentrations.

Agronomic adaptation involves modifying farming practices to improve plant growth or profitability in the presence of soil constraints and/or to reduce the risk of soil constraints developing or worsening. Examples of such practices are listed in Table 25.1 and can include:

- Growing plant species or cultivars that are more tolerant to constraints to maintain yield e.g., acid tolerant wheat cultivars in acidic soils, salinity tolerant species in saline/sodic environments.
- Matching productive inputs (e.g., fertiliser, pesticide) to expected yield to improve profitability and reduce environmental impacts. For example, reduced application of fertiliser in lower yielding areas of a paddock can reduce input costs and prevent excessive fertiliser losses due to sub-optimal plant uptake.
- Modifying crop management to give plants an advantage when growing under constrained conditions. For example, the manipulation of sowing times/seeding rates/row spacings, and ensuring adequate plant nutrition can often help plants cope better in constrained environments.
- Using practices that reduce the extractive nature of agriculture and prevent soil degradation. Examples of such practices can include no-tillage, stubble retention, cover cropping, crop rotations, good N cycle management.

Amelioration and adaptation practices can be used separately or together and there are many instances where they have been applied to improve crop production and reduce yield gaps. The greatest levels of success are likely to occur when they are applied in an integrated manner in order to both improve production and reduce off-site impacts (Chen et al. 2011; Robertson et al. 2016). For example, it is well understood that increasing N fertilisation will alleviate nutrient deficiencies and increase yield. However,

Table 25.1 Examples of the major management approaches for common physical, chemical, and biological soil constraints.

Soil Constraint	Common Management Approaches	
	Amelioration	Agronomic Adaptation
Physical		
Erosion	- Engineering structures e.g., bunds, contour banks - Increase soil cover via stubble retention, cover crops	- No-till to reduce soil disturbance and prevent organic matter loss
Waterlogging	- Drainage structures - Surface water management to divert water flows - Treatment of soil structural problems (dispersion/compaction)	- Raised beds - Higher seeding rates to dewater profiles - Tolerant crops/cultivars
Compaction	- Tillage to remove compacted layers - Growth of deep-rooted perennial species - Treatment of dispersion	- Controlled traffic and no-till - Improved timing of tillage - Minimise organic matter loss - Tolerant crops/cultivars
Chemical		
Acidity	- Lime - Gypsum - Organic matter	- Tolerant crops/cultivars - Improved N cycle management - Matching of inputs and yield potential
Sodicity	- Calcium (gypsum/lime) - Organic matter - Acidification to dissolve native Ca sources and reduce alkalinity - Appropriate irrigation management - Plant based amelioration	- Tolerant crops/cultivars - Cultivation to break up surface seals, compacted layers - Matching of inputs and yield potential - Manipulation of row spacings and sowing rates to optimise water availability

Salinity	- Improve profile leaching - Appropriate irrigation management - Use of tolerant plants to remove salt	- Tolerant crops/cultivars - Matching of inputs and yield potential
Nutrient deficiencies	- Fertiliser - Incorporation of legumes into crop rotations - Organic matter - Treatment of acidity/alkalinity	- No-till/ crop rotations to slow organic matter loss - Minimise erosion - Tolerant crops/cultivars
Toxicities	- Manage acidity/ alkalinity/ waterlogging - Organic matter - Bioremediation for contaminated soils	- Tolerant crops/ cultivars - Appropriate fertiliser and pesticide use

Biological

Soil borne pests and diseases	- Fungicides - Pesticides - Soil disturbance - Biological control - Soil solarisation - Steam sterilisation	- Crop rotation and diversification - Tolerant crops/ cultivars - Appropriate management of pest/ disease hosts (stubble, weeds)
Soil biological diversity decline	- Organic matter increase - Diversified crop rotations	- No-till and stubble retention to reduce soil disturbance and organic matter loss

overuse of N fertiliser can also cause significant environmental degradation (soil acidification, eutrophication of waterways, greenhouse gas emissions) (Zhang et al. 2015). It is only when increased fertiliser use is combined with agronomic practices that ensure high nitrogen use efficiency and decrease environmental losses that maximum improvements in yield, farmer profitability, and environmental sustainability are achieved (Chen et al. 2011; Zhang et al. 2015).

It is also important to recognise that in some circumstances it may not be possible to ameliorate constraints or to adapt agronomic management sufficiently to improve productivity and make cropping economically worthwhile. This may particularly be the case where constraints are very severe, present in the subsoil, or where access to resources limits farmer ability to improve constraint management. Under such circumstances the continuation of farming is likely to occur at very low levels of productivity

and be the source of negative environmental impacts (e.g., erosion, soil organic carbon loss) (Kopittke et al. 2019). Where this is the case the conversion of areas from cropping to other land uses is often the most economically and environmentally sustainable option (Kopittke et al. 2019). This can involve the conversion of areas back to native vegetation, or the introduction of alternative agricultural systems such as grazing or agroforestry (Dang et al. 2006). Where socioeconomic factors limit the ability of farmers to make such shifts in land use, government policies or economic incentives may be required to facilitate change.

Current gaps in soil constraint management

While significant research effort has been devoted to the identification of techniques to both ameliorate constrained soils and to better adapt farming systems, a lack of understanding in several major areas is currently limiting our ability to optimise production.

Multiple constraints

The first of these is an understanding of how to most efficiently manage areas where multiple constraints are present. The majority of research work regarding soil constraints has been conducted where the impact of management change is assessed in relation to singular constraints only. However, in many “real world” situations, multiple constraints occur at the one location and can interact in a variety of ways to influence yield. In these situations, it is necessary to manage all constraints concurrently if maximum yield increase is to be achieved.

Where multiple constraints are present, combinations of ameliorants can often be most effective at improving yield, particularly if ameliorants improve different aspects of problem soil behaviour or one amendment increases the effectiveness of the other (Page, Dang, Dalal et al. 2020; Valzano, Murphy, and Greene 2001; Baldock et al. 1994; Page et al. 2018). However, our ability to identify when combined ameliorant use is appropriate, what ameliorants to use, and at what rates these should be applied given the constraints present is often poor. Similarly, our understanding of appropriate agronomic adaption strategies in the face of multiple constraints is also underdeveloped. For example, the use of tolerant crops/cultivars is a major tool to maintain production in constrained areas, however, plant breeding programs or assessments of crop tolerance tend to focus on tolerance to individual constraints, and it is often unclear which species/cultivars will have tolerance to the range of constraints present at a site (Page et al. 2021). For example, on a saline-sodic site, plants may require tolerance to salinity

and an enhanced ability to growth through soil crusts. However, in most regions, farmers have very poor access to information that would allow them to identify such species

Subsoil constraints

Constraints present in the subsoil often have the greatest impact on crop growth in semi-arid rainfed systems where they can limit the crop's ability to access stored subsoil moisture and have a major impact on yield, particularly during lower rainfall seasons (Dang et al. 2006). Constraints occurring in the subsoil can be challenging to identify, particularly when relying on traditional soil survey, which often does not sample past the topsoil. There can also often be wide variations in yield in areas affected by subsoil constraints, where negative impacts may only be observed in drier years when the crop needs to rely on stored subsoil moisture for growth, further complicating identification. The identification of subsoil constraints using traditional techniques requires deep point sampling, which is typically difficult and expensive to conduct at high spatial resolution. Proximal techniques, such as EM surveys, are often more effective, however, these approaches require access to specialised equipment, which in resource poor regions, can be an impediment to use.

In addition to the issues with identification, the amelioration of subsoil constraints is also difficult due to the challenges (both logistical and economic) of moving ameliorants to depth and physically manipulating subsoil environments (Page, Dang, Dalal et al. 2020 ; Page et al. 2018; Dang et al. 2006). Where subsoil constraints are naturally occurring soil characteristics, rather than ones induced due to poor management, amelioration can be particularly challenging. For example, large areas of the Australian cereal cropping belt are affected by subsoil salinity and sodicity, which have developed naturally due to the build-up of salts under conditions of limited rainfall and profile leaching (Rengasamy 2002). Economically ameliorating such soils, which are predominantly cropped under rainfed conditions, is generally either extremely challenging or impossible.

In such circumstances the use of agronomic adaptations to better manage crop production in the presence of constraints, or switching to an alternative land use (pasture/ agroforestry/ native vegetation), is often the only economically feasible approach to management. The use of tolerant crops/ cultivars and better matching of inputs to match expected yield to improve profitability are often particularly effective strategies (Dang et al. 2010). However, our knowledge of those crops/ cultivars that are best adapted to the range of constraints present at a site is often poor (Page et al. 2021). Matching of inputs to expected yields also requires farmers to have access

to both yield mapping tools and knowledge of how to appropriately manage input rates.

Incorporation of socioeconomic considerations

In many instances, we may have a good understanding of how to manage soil constraints. For example, our understanding of how to deal with soil infertility via fertiliser management, or acidity via lime, is highly developed. However, a failure to adequately address socioeconomic factors often acts as a barrier to implementing these management practices. For example, the nutrient deficient soils common to many regions in southern Africa see significant improvements in productivity once adequate fertiliser is applied (Nhamo et al. 2014; Sánchez 2010). Such an amelioration strategy is relatively easy to achieve from an agronomic perspective, however, fertiliser use is often low due to its low availability and the high price of fertiliser relative to farmer income (Nhamo et al. 2014; Sánchez 2010; Grassini, Eskridge, and Cassman 2013). Similarly, it is well understood that increasing soil organic matter is a key component of sustainable intensification and will alleviate many soil constraints by providing nutrients, reducing certain toxicities, regulating pH, and improving soil structure and aeration (Oldfield, Bradford, and Wood 2019; Foley et al. 2011). However, techniques to increase soil organic matter (organic ameliorants, incorporation of ley phases into cropping rotations etc.) are generally difficult to incorporate economically.

The poor integration of management practices into local farming systems and communities can also be a barrier to effective soil constraint management. For example, the use of conservation agriculture (CA), which incorporates reduced tillage with residue retention and appropriate crop rotations, can often be highly successful at improving or preventing soil constraints by increasing soil organic matter, nutrient content, biological diversity, soil structure and crop water availability (Page, Dang, and Dalal 2020). However, the adaption of CA principles to local soils, climatic conditions, and crops, and ensuring that farmers have access to the necessary equipment and inputs (seeding equipment, fertiliser, seed varieties) is essential for system success. The provision of high-quality extension and economic support to help farmers adopt what can be a radically different farming practice compared to conventional techniques is also essential (Page, Dang, and Dalal 2020; Pannell, Llewellyn, and Corbeels 2014). Where management practices are introduced without adequately adapting the system to local circumstances limited success is often observed, which can lead to either a failure to adopt, or eventual abandonment the CA system (Page, Dang, and Dalal 2020; Giller et al. 2015; Pannell, Llewellyn, and Corbeels 2014). Similarly, strategies such as

matching inputs with expected yield are well demonstrated as a useful approach to improve profitability and decrease environmental degradation in constrained areas. However, the high-tech precision agriculture approaches that are used to achieve this in resource rich regions will not be transferable to smallholder farmers and these approaches will require significant adaptation.

Conclusions and future research

Sustainable agricultural intensification in the face of climate change and increasing degradation of the soil resource is one of the major challenges facing society over the next 50 years. To achieve this intensification will require a multidisciplinary approach that links the skills and knowledge of farmers with specialists in soil science, agronomy, genetics, ecology, engineering, computer science, data analysis, economics, and extension. While we have made significant advances in our ability to manage constraints, further research effort is required to continue to reduce yield gaps and improve environmental outcomes.

The identification of which soil constraints are limiting yield and where exactly they occur (across the landscape, within fields, and down the soil profile) is the first key element required for the effective management of soil constraints. Consequently, greater research to provide farmers with tools that improve their understanding of the constraints occurring across properties are likely to lead to significant improvements in their capacity to close yield gaps. Research effort to fully exploit the large amount of data that is now available from remote sensing platforms and harness advances in computing power and analysis techniques to create constraint identification tools are likely to be particularly fruitful areas of research.

Continued focus on the identification and optimisation of amelioration techniques and agronomic systems to best cope with soil constraints will also be required, and particularly the adaptation of these techniques to local climates, soil types, crops, and communities. Increased emphasis on how to achieve optimum management in the presence of multiple constraints and when constraints are present in subsoil environments will also be essential for many regions. Research with a focus on the improved integration of ameliorant use with adaptive agronomic management to improve overall system performance while at the same time reducing the environmental impact of agricultural systems is likely to meet with the greatest long-term success. Indeed, if soil constraints are tackled without considering the offsite impacts then they will inevitably fail to lead to overall improvements in human wellbeing over the longer term.

Research to identify high quality and locally adapted extension techniques to increase awareness of constraints and provide farmers with the information they need to identify logistically and economically viable management approaches are also essential to improve the adoption on the ground. It is not enough to simply identify methods of increasing yield and decreasing environmental degradation. These approaches must be tailored for individual socioeconomic environments, as unless management is viewed as logistically sensible and economically sustainable by farmers it will not be adopted, and yield gaps will persist.

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