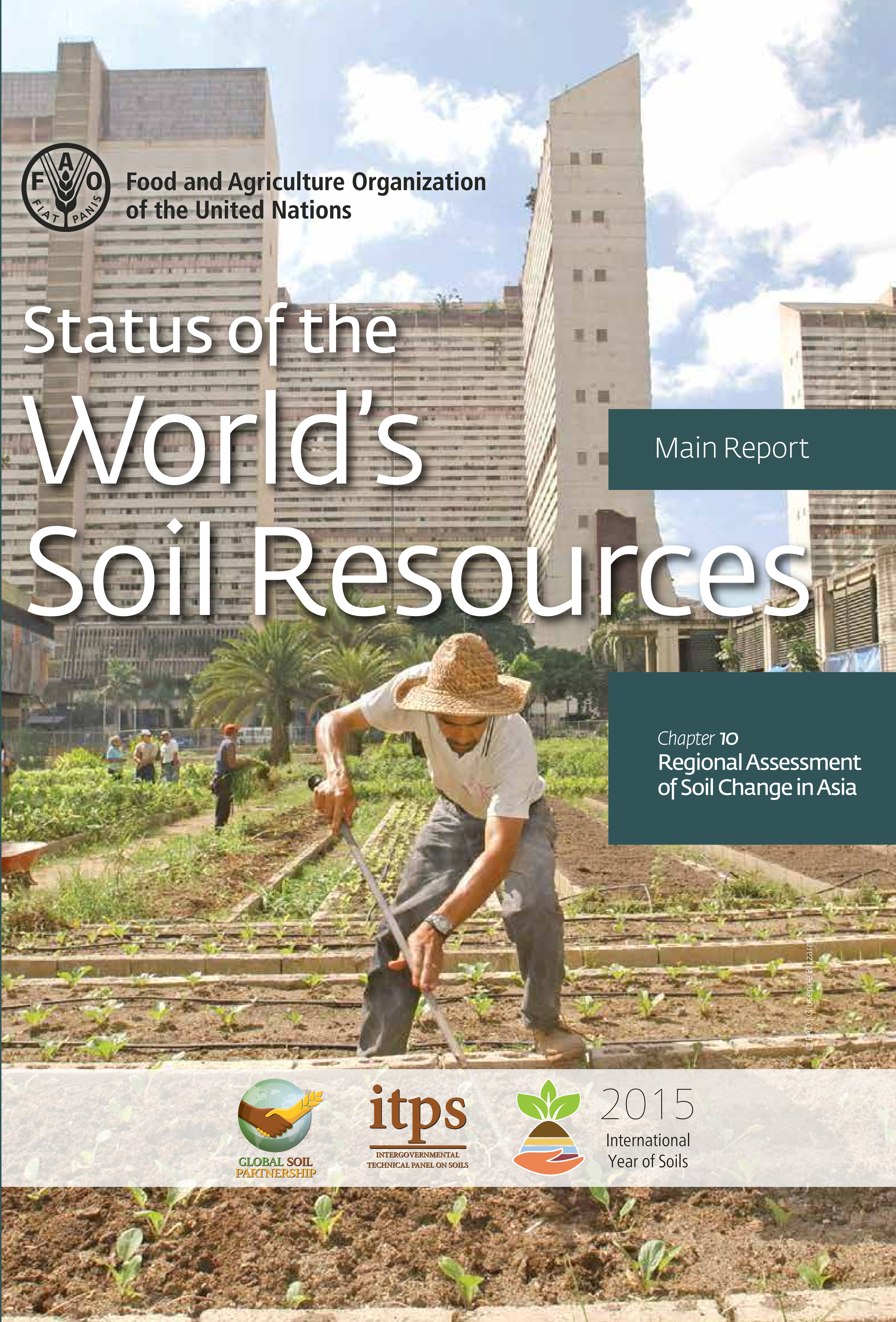

Food and Agriculture Organization of the United Nations

# Status of the World's Soil Resources

Main Report

*Chapter* 10
## Regional Assessment of Soil Change in Asia

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GLOBAL SOIL PARTNERSHIP

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INTERGOVERNMENTAL TECHNICAL PANEL ON SOILS

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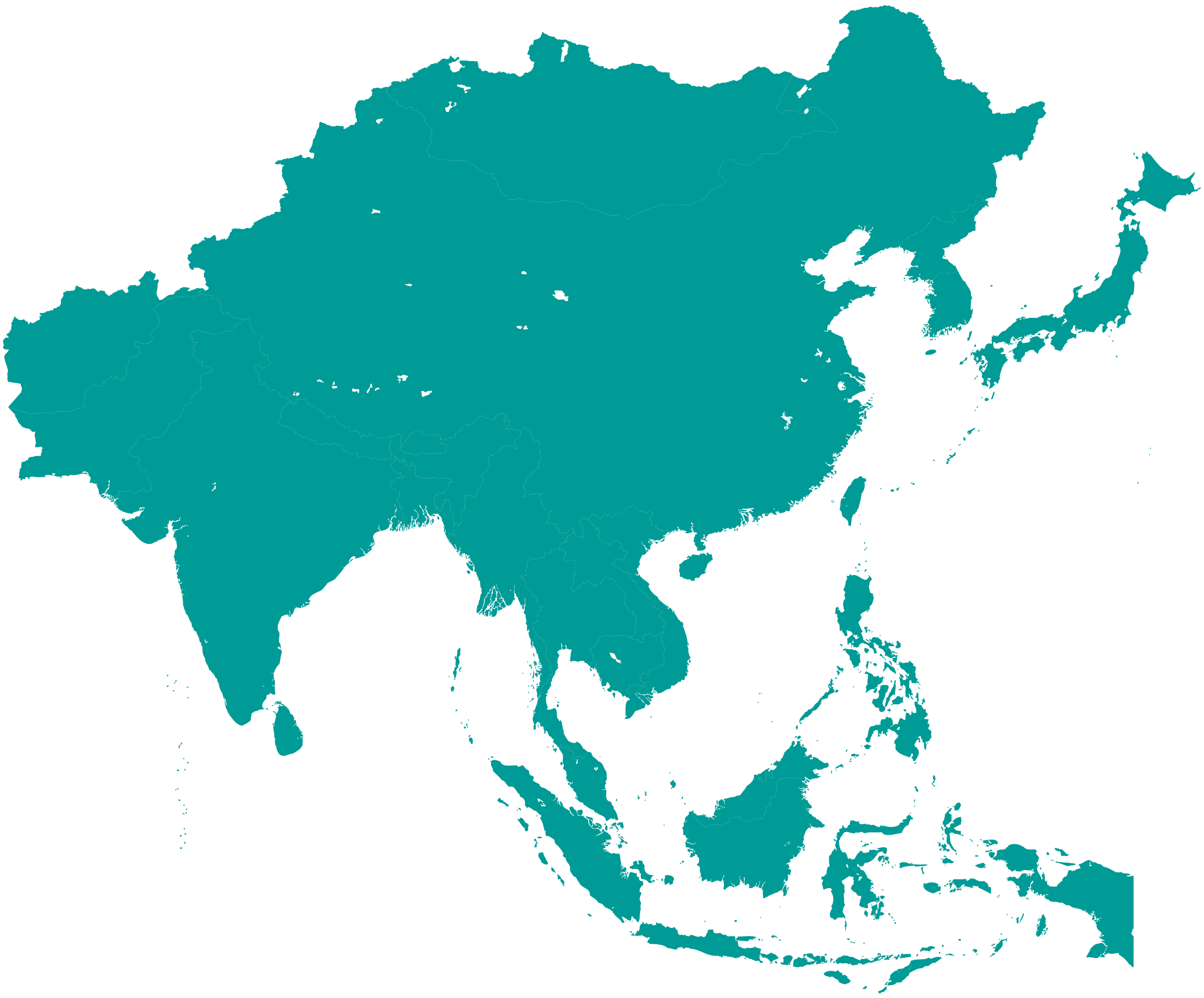
# 10 | Regional Assessment of Soil Change in Asia



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## 10.1 | Introduction

This chapter describes the status of soil resources in the member countries of the Asian Soil Partnership (ASP), which includes East Asia (five countries: China, Democratic People's Republic of Korea, Japan, Mongolia, and Republic of Korea), Southeast Asia (11 countries: Brunei Darussalam, Cambodia, Indonesia, Lao People's Democratic Republic, Malaysia, Myanmar, Philippines, Singapore, Thailand, Timor Leste, and Viet Nam), and South Asia (eight countries: Afghanistan, Bangladesh, Bhutan, India, Maldives, Nepal, Pakistan, and Sri Lanka).

Asia, the Earth's largest and most populous continent, is located largely in the eastern and northern hemispheres. The region, which consists of the above-mentioned ASP countries, covers 4.1 percent of the Earth's total surface area and comprises 16 percent of its land area. With approximately 3.9 billion people, Asia hosts 55 percent of the world's population. Population density is high – averaging 1.87 persons $ha^{-1}$ – compared to the world average of 0.54 person $ha^{-1}$. Like most areas of the world, Asia has experienced rapid population growth rate in the modern era. In the twentieth century, Asia's population nearly quadrupled, as did the world population.

In general, Asia enjoys a warm and seasonally humid climate and is well-endowed with natural resources for agriculture. The unique combination of the monsoon climate and the exceptionally large lowland area has made Asia the rice basket of the world (Kyuma, 2004). Sustained high levels of staple food production have enabled Asian countries to support a large population within a limited area of arable land. However, recently, Asian countries have faced rapid changes in both socio-economic and natural factors and these have had major impacts on agro-environments in the region. In particular, rapid economic development and urbanization are changing land management systems in many countries, and climate change has emerged as a significant source of risks. These changes are having major impacts on the status of soil resources in the region.

## 10.2. Stratification of the region

### 10.2.1 | Climate and agro-ecology

The map of Asia shows many vast rivers with large alluvial plains and deltas. Major rivers include the Yellow, Yangtze, Mekong, Chao Phraya, Irrawaddy, Ganges, Brahmaputra, and Indus. The Himalayan mountain range runs for more than 2 400 km, separating the Indian subcontinent from the rest of Asia. Many of Asia's major rivers have their source in the Himalayas and adjacent plateaus. These rivers have created vast areas of fertile land suitable for farming. On the east and southeast shores of the continent lie a number of islands chains or island arcs, many characterized by mountainous landscape with volcanic activities.

Most areas of the Asian region are strongly influenced by the monsoon, a seasonal reversing wind accompanied by corresponding changes in precipitation. For this reason, the region is often called 'Monsoon Asia'. The Asian monsoon is a highly significant component of the global climate system. It has a huge influence on how people live and on their livelihoods, and it provides water resources throughout the region (Salinger *et al.*, 2014). The East Asian monsoon carries moist air from the Pacific Ocean and the Indian Ocean to East and Southeast Asia in summer. In winter, it reverses and carries cold, dry air from the Eurasian Continent. In south Asia, winds blow from June to September from a south-westerly direction from the Indian Ocean onto the Indian landmass, bringing rain to most parts of the subcontinent. Subsequently, from around October, the winds reverse direction and start blowing from a north-easterly direction, from the subcontinent onto the Indian Ocean. These winds carry less moisture and bring rain to only limited parts of India. Dry areas predominate in parts of the north-western interior where the influence of oceanic winds is less. This wide diversity of climate is a major factor in the stratification of the region into different agro-ecological zones (Figure 10.1)

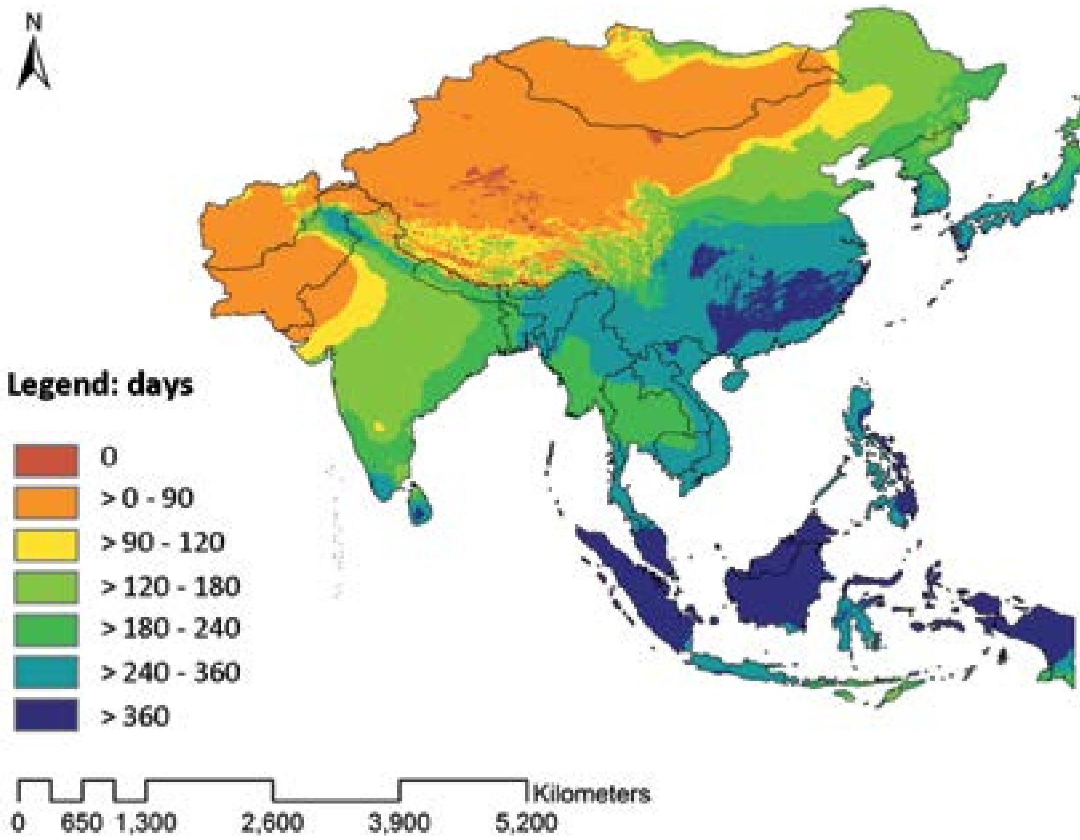


Figure 10.1 | Length of the available growing period in Asia (in days yr[-1]).
Source: Fischer et al., 2012.

A distinctive feature of monsoon Asia is its very high share of the world's area and production of rice (Kyuma, 2004). Some 90 percent of the global acreage and output of rice are concentrated in the region, earning it the title of 'rice granary' of the world. This dominance of rice cultivation is due to several factors, notably high precipitation and temperatures and the existence of extensive lowlands suitabe for paddy prduction. The vast expanse of lowlands is a unique feature of the region, resulting from a combination of geological instability and the high precipitation. Rice cultivation originally emerged as an adaptation to extensively inundated lowlands, but with time it was expanded to areas that could support rice only with irrigation. High productivity and high sustainability are the outstanding advantages of rice cultivation. By contrast, upland cultivation of dryland crops in Monsoon Asia is handicapped by low soil fertility and high susceptibility to soil erosion.

### 10.2.2 | Previous regional soil assessments

Based on the report by Oldeman (1991), the GLASOD project estimated that human-induced soil degradation in Asia region (including non-ASP west Asian countries) accounted for 31 percent of the inhabited land area, the highest share of any of the global regions. Soils in Asia were found to have been degraded by several factors: water erosion (59 percent), wind erosion (30 percent), chemical degradation (10 percent) and physical degradation (2 percent). Deforestation was identified as the most dominant causative factor for soil degradation, followed by agricultural activities and overgrazing. In most Asian countries the mining of soil nutrients is causing decline of average crop yields. Fertile soil is washed away by the erosive forces of water, or blown away by wind. This so-called first generation of environmental problems is leading not only to negative nutrient balances but also to habitat destruction and loss of biodiversity (Oldeman, 2000).

Following GLASOD, the need for more detailed and more country-specific degradation assessments became apparent. In 1993, the members of the Asian Network on Problem Soils recommended the preparation of a qualitative assessment for South and Southeast Asia at a scale of 1:5 million. This recommendation was acknowledged by FAO and UNEP. FAO assigned ISRIC to prepare a new physiographic map and database at 1:5 million scale. UNEP prepared and implemented the Assessment of the Status of Human-Induced Soil Degradation in South and Southeast Asia (ASSOD). Sixteen national institutions for natural resources in the region collaborated on the project under the coordination of ISRIC. The ASSOD project published revised sub-regional Guidelines for General Assessment of the Status of Human-Induced Soil Degradation and produced regional maps on the status of human-induced soil degradation at a scale of 1:5M together with digitized version of the map (van Lynden and Oldeman, 1997).

The different soil degradation types inventoried by ASSOD are described below and shown in Figure 10.2:

- Water erosion: Water erosion covers 21 percent of the total land area in the region (or 46 percent of the total degraded area). It is predominant in large parts of China (>180 million ha) except for the northern parts, on the Indian subcontinent (>90 million ha) and in the sloping parts of Indochina (40 million ha), the Philippines (10 million ha) and Indonesia (22.5 million ha). In relative terms, as a percentage of the total country area, moderate to extreme water erosion is particularly important in India (10 percent), the Philippines (38 percent), Pakistan (12.5 percent), Thailand (15 percent) and Vietnam (10 percent).
- Wind erosion: Wind erosion (9 percent of the total area, 20 percent of all degradation) is concentrated mainly in the most western and northern arid and semi-arid desert regions of Pakistan (>9 million ha on-site and >2 million ha off-site), India (20 million ha on-site, 3.6 million ha offsite) and China (>70 million ha on-site, >8.5 million ha off-site). Although large parts of these regions may be considered deserts, some human-induced wind erosion was also reported.
- Chemical deterioration: Chemical deterioration is distributed in patches, probably also partly due to different perceptions of this type of degradation. About 11 percent of the total area (or 24 percent of the degraded area) is affected by some kind of chemical deterioration. High relative extents of chemical deterioration (>30 percent of total country area) can be observed in Bangladesh, Cambodia, Sri Lanka, Malaysia, Pakistan and Thailand, generally with negligible to light impact.
- Physical deterioration: Occurrence of physical deterioration (affecting about 4 percent of the total area or 9 percent of the total degraded area) is even more dispersed and infrequent than chemical deterioration. Waterlogging and aridification are the main subtypes, in particular in Bangladesh, China, India and Pakistan. Compaction or crusting/sealing is relatively unimportant except in Thailand and the Philippines, although they occur in most countries. Waterlogging and compaction as a result of paddy cultivation are not considered as degradation. Loss of productive function as a result of urbanisation, industrialisation and infrastructure has been indicated for only a few countries although this phenomenon is on the rise.

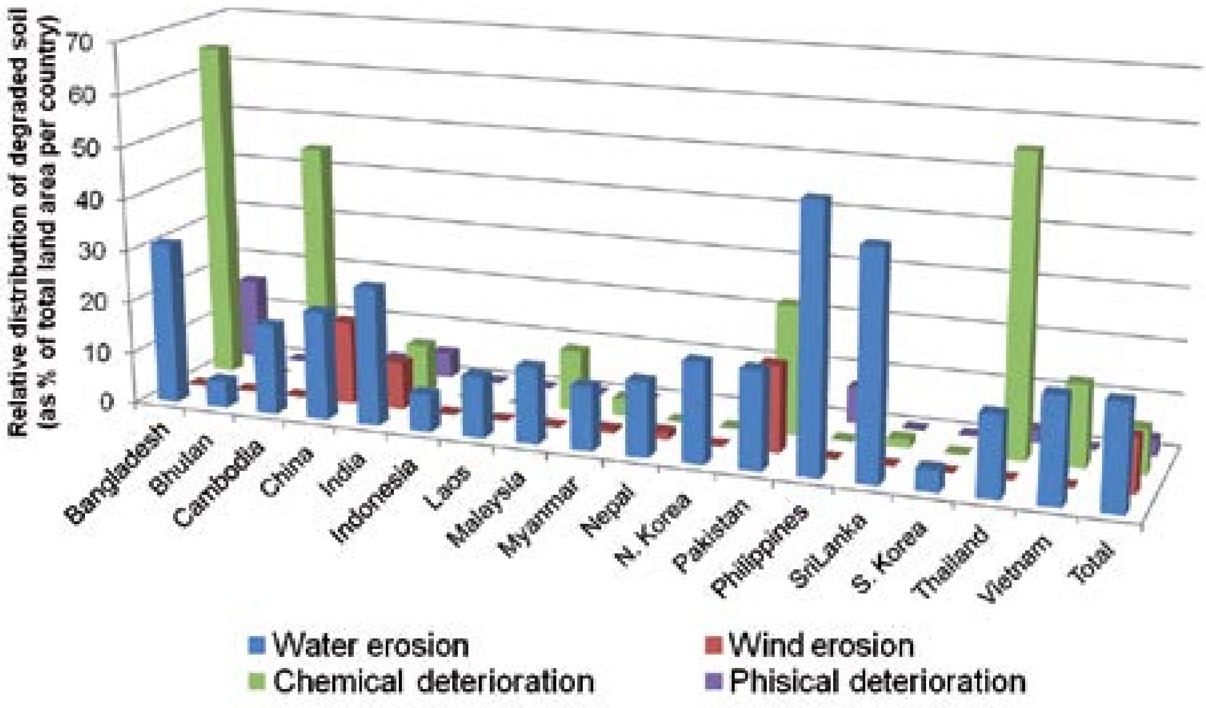


Figure 10.2 | Threats to soils in the Asia region by country.

## 10.3 | General threats to soils in the region

### 10.3.1 | Erosion by wind and water

Soil erosion is the most important threat to soil in Asia. Water erosion by rainfall and surface water flow is dominant in humid regions with torrential rains, as in South and East Asia. In drier or desert areas, wind is the driving force inducing soil erosion. This threat is discussed below in Section 10.4.1 in detail.

### 10.3.2 | Soil organic carbon change

Data for evaluating soil organic carbon (SOC) change in Asian countries are limited because countries do not generally monitor SOC stock and changes. However, data from available literature show that where there are increases in crop yield in croplands of East and Southeast Asia, SOC is retained. SOC has also been shown to accumulate in forest areas. However, in South Asia SOC is decreasing. This is because crop residues are widely used as fuel and fodder and are not returned to the soil. In Indonesia, three anthropogenic activities – deforestation, poor land management, and intensive cropping – contribute to SOC change in mineral and peat soils (Section 10.5.2). Throughout the region, the degradation of grassland has generally caused great losses of SOC stock. This threat is discussed in detail below in Section 10.4.2.

### 10.3.3 | Soil contamination

Sources of contamination of arable land in most Asian countries include (I) parent material, (II) mining, (III) smelting, (IV) agrochemicals and sewage sludge applications, and (V) livestock manure uses (Luo *et al.*, 2009). There is an urgent need to reduce hazardous chemical concentrations of Cd, As, Pb, Cu and Zn, especially in paddy soil and rice grains. In many regions of Southeast Asia (Bangladesh, India, China, Vietnam, Taiwan - Province of China, Thailand and Nepal), arsenic is naturally present in groundwater and represents a threat to sustainable agriculture (Smedley, 2003; Brammer and Ravenscroft, 2009). This enrichment is magnified considerably when mining takes place, as it is the case of the Ron Phibun district, in southern Thailand (Williams *et al.*, 1996). Arsenic contamination particularly affects rice, as this crop not only requires large amounts of water but is also grown under the anaerobic conditions favoured in rice fields where As is mainly present in its trivalent form which is readily available to plants (Brammer and Ravenscroft, 2009). The agricultural rice soils of the Guandu Plain in Taiwan, Province of China are seriously contaminated with As and Pb (Zhuang *et al.*, 2009; Chang *et al.*, 1999, 2007). Cadmium pollution of paddy fields has been found downstream of a Zn mineralized area in Thailand (Simmons *et al.*, 2005). China contributes around 28 percent of global Hg emissions, with India, Japan and the Korea Democratic Republic also among the ten countries that contribute the most to global Hg emissions (Li *et al.*, 2009).

According to the Agricultural Land Soil Pollution Prevention Law in Japan, the maximum allowable limit of Cd in paddy fields is set in terms of the Cd concentration in rice grains produced in the field, not the soil Cd concentration of the field. This is because the amount of bio-available Cd in soil is affected dramatically by water management practices used for rice cultivation (Asami, 1981).

With rapid industrialization, urbanization and intensive use of farmland, China is now facing serious soil pollution (Ministry of Environmental Protection, the People's Republic of China, 2014). About 19.4 percent of farmland has high levels of Cd, Ni and As pollution. Soil contamination has been estimated to cause a reduction of more than 107 tonnes of food supply annually (Wei and Chen, 2001). In 2006-2010, China's Ministry of Environmental Protection and Ministry of Land and Resources jointly launched a nationwide investigation of soil pollution status, covering an area of 6.3 million square kilometers. In 2014, a *Communiqué on Nationwide Soil Contamination* was issued by the two ministries indicating that the overall situation of the soil environment in China is not encouraging. Some regions are heavily polluted. There is concern over the quality of farmland soils and over other soil environmental issues also caused by anthropogenic activities, e.g. those related to mining and industrial activities which cause atmospheric deposition, and to the use of livestock manures (Luo *et al.*, 2009). Trace elements are pollutants of major concern, especially in the southern area of China. Currently, the evaluation of soil pollution in China is primarily based on the *Environmental Quality Standard for Soils* which was promulgated and implemented in 1995. The Standard needs further improvement because of its stringent limitations. At present, China's Ministry of Agriculture is working on an *Implementation Plan on Prevention and Control of Heavy Metal Pollution in Agricultural Products*.

Historic and current rates of intensive pesticide and fertilizer use in agricultural land and also industrial development have caused the accumulation of organic pollutants and heavy metals in soils of Indonesia. Earlier research showed high concentration of organo-chlorines in vegetables (Dibiyantoro, 1998). However, residue levels in foodstuffs have gradually reduced to within acceptable daily intake levels as established by WHO (Shoiful *et al.*, 2012; Rahmawati *et al.*, 2013). The tapioca industry in Java is now recognized as a contributor to cyanide levels which have risen above background levels in river water (Indrayatie *et al.*, 2013). Mining plays an important role in the Indonesian economy but this mining, particularly artisanal and small-scale mining (ASGM), can have a major impact on the environment (Limbong *et al.*, 2003; Prasetyo *et al.*, 2010). During ASGM, Hg is used to recover Au from the ore during grinding. The process is inefficient and releases a significant amount of Hg to soil, water and the atmosphere (Limbong *et al.*, 2003; Edinger *et al.*, 2008). Tailings from Hg amalgamation are then leached with cyanide. Ultimately, the final waste, contaminated with metals and cyanide, is released into the environment (Veiga *et al.*, 2009). Many ASGM operations also release As and Sb to the environment, although this depends on the composition of the host ore (Edinger *et al.*, 2008). These operations are unlicenced and illegal. Indonesia has now signed the Minamata Convention on Hg and has decentralized control of ASGM operations to provincial governments. Environmental protection has become a primary objective for government regulation.

## 10.3.4 | Soil acidification

There is large area of acid soils distributed across the tropical and subtropical regions of Asia, mainly in Southeast Asia, parts of East Asia and parts of South Asia. Acid sulphate soils are widely distributed in the coastal plains of Southeast Asia and southern China. The total area of acid sulphate soils in Southeast Asia is 7.5 million ha and there are about 112 thousand ha of these soils in China (Shamshuddin *et al.*, 2014). The soils in these regions are also sensitive to external acid such as acid deposition (Hicks *et al.*, 2008).

In Vietnam, ferralitic, basaltic, and grey degraded soils, which cover about one third of the country, have strong potential for acidification and degradation because of their nature (NISF, 2012). A leading contributor to soil acidification in Vietnam is the unbalanced and unsuitable application of chemical fertilizers. Data from the International Fertilizers Association (IFA) show that from 1961 to 2012, total NPK chemical fertilizer use in Vietnam increased 31 times (IFA, 2012). The increase in fertilizer application and unbalanced use lead to inadequate presentations of acidic factors in soil solutions. Some types of fertilizers (including organic fertilizers) can make the soil more acidic (Nguyen, 2014). Another reason for the acidification process in Vietnam is the presence of sulphate soil. The area of sulphate soil in the Red River delta has increased by 7 000 ha and the content of S in this soil has also strongly increased. In the Mekong river delta, the area has reduced by 261 000 ha and the content of total exchangeable cations and total dissolved salts has also reduced. Increase in temperature, change of precipitation, and sea level rise due to climate change may also be having negative effects on this process in Vietnam.

The total area of Indonesian acid upland soils (pH <5 and <50 percent base saturation) is about 102.8 million ha, of which more than half (55.8 million ha) are suitable for various types of agriculture and plantation (Puslitbangtanak, 2000). Relatively low cost technologies are widely available for rectifying soil acidity problems. These include liming, organic matter application, balanced fertilization and selection of acid tolerant crops. However, farming practices, such as application of acidifying fertilizers like ammonium sulphate, aggravate the problem (Kamprath, 1984).

In China, acid soils are mainly distributed in tropical and subtropical regions south of the Yangtze River. The total area of acid soils is about 204 million ha, 22.7 percent of the total land area in the country. In the past three decades, soil acidification has accelerated in southern China due to serious acid deposition and the heavy application of ammonium-based fertilizers. In subtropical regions of the country, soil pH decreased on average by 0.23 units for cereal cropland and 0.30 units for cash cropland between the 1980s and 2000s (Guo *et al.*, 2010). Soil acidification in cropland in China was accelerated mainly by ammonium-based fertilizers, while acid deposition was mainly responsible for acidification in forest soils. Although the emission of $SO_2$ was reduced by 20 percent in China in the past ten years, acid deposition is still a serious problem and its impact on soil acidification is unlikely be eliminated in the near future (Zhao *et al.*, 2009).

Tea is cultivated extensively in subtropical regions of China, Japan, Southeast Asia and parts of South Asia. There is serious acidification of soils in tea gardens (Wang *et al.*, 2010). Excess application of ammonium-based fertilizers and lack of leaching were responsible for acidification and salinization of soils in vegetable greenhouses (Guo *et al.*, 2010).

## 10.3.5 | Soil salinization and sodification

The threat of salinization/sodification in the region takes varying forms. In the semiarid and arid zones of central and west Asia, salt-affected soils are widely distributed. On the other hand, salt-affected soils are also developing in certain coastal areas in monsoon zones, caused mainly by salt water intrusion in South and Southeast Asia and by coastal tideland reclamation. Although the coastal area affected is relatively small, this could become a serious problem for lowland rice production. This threat is discussed in detail below in Section 10.4.3.

### 10.3.6 | Loss of soil biodiversity

Potentially the greatest contributor to soil biodiversity loss in Asia is land use change. In China, for example, an assessment of land-use change across the country indicated that the largest changes were associated with the conversion of productive cultivated land to urban areas, thereby removing fertile land from agricultural production. Conversion of cultivated areas, forest and grasslands between 1996 and 2008 has been estimated to be 1 475 × 104 ha, 269 ×104 ha, and 536 ×104 ha, respectively (Wang *et al.*, 2012). There have been attempts at land restoration to maintain soil biodiversity, such as in the coastal lands in the Jiangsu Province of China. Generally, there is a higher diversity of soil macrofauna in uncultivated land and forests compared to less diverse wheat farms and bulrush land (Baoming *et al.*, 2014). Soil faunal diversity is significantly impacted by land use with a strong relationship between vegetation and macrofauna distribution and composition (Baoming *et al.*, 2014). In Thailand, the forested land area has decreased by more than 50 percent in the past 40 years (Fisher and Hirsch, 2008). Although reforestation measures can be used to restore degraded lands, difficulties with establishment of native trees in these areas may lead to the use of substitute trees, resulting in long lasting effects on soil biodiversity. For example, a study of soil microbial communities of an *Acacai* tree plantation established on degraded land in Thailand found reduced microbial activities compared to natural evergreen forest, suggesting that key microbes had been lost (Doi and Ranamukhaarachchi, 2013).

Studies in India have also indicated an effect of land use change on soil organisms. The Nilgiri biosphere reserve in the Western Ghats of India, a global hotspot for aboveground biodiversity, has been under pressure because of high population density in the surrounding region (Mujeeb Rahman, Mujeeb, Varma and Sileshi, 2012). In this area, land management had a significant effect on soil macrofauna, with larger densities and diversity found in the forest sites and a clear response of macro-invertebrates to land use (Rossi and Blanchart, 2005). Interestingly, there was a high similarity in macrofauna between primary forest and disturbed forest plots, which indicated the potential of land restoration. A separate morphological analysis of soil invertebrate density at 15 different land use sites (from intensively managed agricultural systems to pristine forests) identified a wide range of soil faunal groups including earthworms, termites, ants, grasshoppers, crickets, mole crickets, bugs, coccids, cicadas, woodlice, centipedes, millipedes, and spiders (Mujeeb Rahman, Mujeeb, Varma and Sileshi, 2012). The natural forests had significantly higher taxonomic richness at the family level than soils from the annual cropping system and a significantly higher total number of individuals than annual crops, agroforestry and plantations. The highest richness (identified to family level) was found in the sites with the least anthropogenic disturbance, and the greatest diversity of earthworms, ants and termites (determined to species level) was found in the more complex forest ecosystems. An earlier study in the region recorded almost double the number of earthworm species in soils from forest compared to pasture (Blanchart and Julka, 1997). These results indicate a decrease in earthworm biodiversity associated with forest loss and a lower diversity of soil macrofauna with more intensive land use.

In addition to anthropogenic pressures on soil biodiversity, natural disturbances such as tsunamis can affect soil biodiversity. Tsunami-affected areas in the Phang Nga province of Thailand had a higher proportion of prokaryotes (archaea and bacteria; 83.25 percent) compared to non-affected areas (72.5 percent), whereas the non-affected areas were more hospitable to eukaryotes (animals, plants, fungi and protists) (Somboonna *et al.*, 2014). Increased occurrences of tsunamis and other natural disasters may result in losses to above and belowground biodiversity.

There has not been a comprehensive analysis of threats to soil biodiversity in Asia completed to date. Scientific evaluation of soil biodiversity over large regions has been extremely difficult due to: (1) size of organisms, (2) large abundance and diversity of organisms, and (3) lack of research and gaps in the information collected. In Japan, the fusion of complex systems theory and computer technology has made it possible to employ the tools of statistical physics to develop a totally new technology for assessment of farmland soils (Yokoyama, 1993). The technology is used primarily for a health check on farmland soil, to measure the environmental affinity of production. Commercial analysis services have been started to add value to the products (Sakuramot, Yokoyama and Iekushi, 2010; Yokoyama and Taguchi, 2013).

In paddy rice cultivation and upland farming with sustainable agriculture, crop rotation and organic amendments have generally maintained good soil microbial diversity. However, loss of significant microbial diversity is notable in large-scale and intensive vegetable production areas due to: (I) continuous monoculture cropping, (II) over-cultivation, (III) intensive use of chemical fertilizers, (IV) chemical fumigation of soils to prevent soil-borne diseases, (V) soil contamination and non-target effects of chemical pesticides such as fungicides, nematicides, herbicides and insecticides, and (VI) over-application of herbicides for weed control. The loss of diversity in the soil is conducive to further diseases and other cropping issues from processes (II) to (VI).

Recently, the effect of poor management practices on soil biodiversity has led to concerns. As a result, environmental and safety-oriented consumers have tried to promote organic farming and to accelerate the branding of sustainably produced agricultural products based on scientific evidence. The idea is to create incentives for farmers to consider soil biodiversity in their farming practices. The private sector has undertaken evaluation of microbial diversity, demonstrating high soil microbial diversity in organically managed paddy fields in Taiwan - Province of China and in the Philippines, as well as in upland crops in southern China. However, in soils of the semi-arid areas of northern China, the loss of microbial diversity is severe. Here the processes mentioned –in points (I) to (VI) above may lead to soil drying, increases in salt concentrations and a loss of stable soil surface. These changes could potentially result in loss of soil organisms, although research exploring the loss of ecosystem functions and services from these soils has yet to be conducted.

## 10.3.7 | Waterlogging

Two types of waterlogging may occur – permanent waterlogging in natural swampland, and occasional waterlogging in flood prone areas along the flat coastal regions and flood plains of main rivers. Constructed wetlands such as paddy fields are intentionally flooded as part of the management system. Waterlogging can have other anthropogenic causes such as poor drainage systems in settlements, industrial and urban development, or deforestation in upstream areas, all of which may increase the threat of water logging in flood prone areas.

In the GLASOD estimate, waterlogging affects 4.6 million ha in Asia, largely in the irrigated areas of India and Pakistan. Waterlogging is closely linked with salinization. Since the start of large scale irrigation schemes in the 1930s, the progressive rise in the water table beneath irrigation areas on the Indo-Gangetic plains has been monitored (e.g. Ahmad and Kutcher, 1992). For India, monitoring results suggest a waterlogged area more than twice the GLASOD estimate. For Pakistan, four sources give total areas affected by waterlogging of between 1.6 and 3.7 million ha, compared with the GLASOD value of 0.96 million ha. Since the Pakistan country data come from at least two independent surveys, show good agreement and are believed to result from detailed field surveys, these country estimates are likely to be more accurate than the much lower GLASOD estimates.

## 10.3.8 | Nutrient imbalance

Harvested crops remove nitrogen, phosphorus, and other nutrients from agricultural soils. Hence, sustaining agricultural production requires replacement of those nutrients, whether through biological processes like nitrogen fixation or through the addition of animal wastes or mineral fertilizer to fields (Vitousek *et al.*, 2009). Balanced nutrient supply is essential for achieving high crop yields, but excessive and/or imbalanced nutrient input may pose risks to the environment, human health and ecosystems. Nutrient inputs in Asia vary considerably amongst countries, areas and farming systems.

In some countries or regions in Asia, removal of nutrients from the soil in crop harvest appears substantially to exceed inputs through natural replacement or fertilizer application. For example, negative soil nutrient balances have been reported for each of the 15 agro-climatic regions of India (Biswas and Tewatia, 1991; Tandon, 1992).

Problems are also caused by imbalances in fertilizer application. Fertilizer use in the region is often dominated by nitrogen (N), relative to phosphorous (P) and potassium (K). This trend originated in the early years of the 'green revolution'. When fertilizers are first applied to a soil, a high response is frequently obtained from nitrogen. The improved crop growth depletes the soil of other nutrients; "In such systems, nitrogen is simply used as a shovel to mine the soil of other nutrients" (Tandon, 1992). Long-term experiments in India show depletion of soil P and K is higher for plots with N fertilizer, and depletion of K is higher still with N+P fertilizer (Tandon, 1992). The use of N fertilizer in Indonesia has been continually increasing since the late 1960s, reaching 2.4 million Mg $yr^{-1}$ in 2012. However, the consumption of P and K has not followed the same trend (IFADATA, 2007). This is partly due to the fact that N fertilizers are cheaper and more accessible, and also to the rapid crop response to N fertilization. A study on nutrient balances in rice fields under intensive cultivation found a positive balance for N, P, Ca, Mg and Na, but a negative balance for K and Si (Husnain, Masunaga and Wakatsuki, 2010).

For secondary nutrients and micronutrients, an increasing incidence of sulphur and zinc deficiency is occurring in the region. Sulphur deficiency has been reported for India, Pakistan and Sri Lanka, and zinc deficiency for India and Pakistan (FAO/RAPA, 1992). For Bangladesh, 3.9 million ha are reported to be deficient in sulphur and 1.75 million ha in zinc, including areas of continuous swamp rice cultivation (Bangladesh, 1992; Shaheed, 1992). Because of its generally alkaline soils, Pakistan is particularly liable to micronutrient deficiencies (Twyford, 1994).

On the other hand, nutrient additions to many fields in some Asian countries far exceed those in the United States and Northern Europe, and much of the excess fertilizer is lost to the environment, degrading both air and water quality. This important threat to soil is described more in detail in Section 10.4.4.

### 10.3.9 | Compaction

Slightly compacted soil conditions are conducive to soil productivity as they reduce soil erosion and maintain soil structure. Highly compacted soil is, however, a physically deteriorated condition affecting plant productivity under various land uses. Decrease in soil porosity that affects water content, hydraulic conductivity and gas permeability is a major disadvantage brought about by soil compaction. Loading of heavy equipment strongly compresses surface soil and/or subsoil in cropland, grassland and timber forests.

Mechanization of cultivation and harvest in Asian countries has increased, resulting in soil erosion and soil compaction due to tractor loading (Zhang *et al.*, 2006). Some studies of soils in rice/wheat cropping areas of India showed increases in compactness of subsurface soils as indicated by increased bulk density as high as 1.80 g $cm^{-3}$. This was due to the use of heavy machinery in conjunction with puddling activities (Sidhu *et al.*, 2014; Singh, Jalota and Sharma, 2009; Aggarwal *et al.*, 1995; Kukal and Aggarwal, 2003). Heavy machines for harvesting and skidding logs also compress soils considerably, and may be accompanied by rutting on the ground and by removal of organic matter (Kozlowski, 1999; Hattori *et al.*, 2013). The increase in the area of plantation forests in Asia, which has been especially rapid in China in recent decades (FAO, 2006), has led to compaction of soils through the use of heavy equipment for management. Livestock trampling is also a major cause of surface soil compaction in grassland and hilly grazing areas (Drewry and Paton, 2000). Soil compaction due to heavy grazing has led to severe land degradation in the extensive pastoral steppe regions of Mongolia and Inner Mongolia of China (Kruemmelbein, Peth and Horn, 2008). Where soils have been compacted in agricultural and forested areas, water surface runoff followed by soil erosion is a major threat. Soils in urban green areas are commonly compressed by human traffic and by vehicles for park management. This results in damage to plant roots and reduces productivity (Jim, 1998a) because in urban park vegetation over 50 percent of root density is concentrated in the top 50cm of the soil depth (Millward, Paudel and Briggs, 2011). Surface runoff on compacted soil generates flooding and delivers contaminants such as heavy metals and persistent organic pollutants into receiving water environments.

### 10.3.10 | Sealing and capping

Sealing and capping on the soil surface is mainly required in urban areas to construct roads and buildings. Although the impervious surface area (ISA) covers only 0.43 percent of the global land area at present (Schneider, Friedl and Potere, 2009), ISA is constantly on the increase, as shown by satellite image data and by measures of the constant increase in urban population (Elvidge *et al.*, 2007). More than half of the global population is now concentrated into cities. The Asia region has the largest ISA ratio in the world (Schneider, Friedl and Potere, 2009). China has the largest ISA in Asia followed by India, Indonesia, Japan and Bangladesh (Elvidge *et al.*, 2007). Increase in the ISA causes environmental issues such as the formation of urban heat islands (Changnon, 1992), increases in surface water runoff (Booth, 1991), and reduction of carbon sequestration due to reduction of the forested area (Milesi *et al.*, 2003).

Inter-regional differences in properties of sealed soils are not well documented. However, in general, construction processes of sealing affect soils physically and chemically due to disturbance of surface and subsoil by excavation and filling, and by addition of construction materials. In Japan, very large volumes of soil are excavated every year for land levelling, and much of this soil is carried away to other sites. Soils sealed by construction are compressed to enhance their physical strength, making them hugely compacted structures. Additives such as lime, which is used to enhance the sub-base strength of a road, make soils alkaline (Jim, 1998b). Pervious asphalt paving and inter-locking covers for light traffic roads are now recommended to drain rainwater by infiltration to the sub-soil. However, this infiltration treatment can make soil solution and drainage water alkaline. Cracks on a paved road surface due to heavy traffic may allow rainwater to infiltrate into the subsoil, reducing the strength of the roadbed and making the sub-soil alkaline.

## 10.4 | Major threats to soils in the region

### 10.4.1 | Erosion

Soil erosion is one of the major threats to soil quality in Asia. Soil erosion is the action of exogenic processes such as water flow and wind to move soil from its location. The processes inducing soil erosion vary with climate. Asia can be divided into several climate zones: tropical and subtropical in South Asia, humid subtropical and temperate in East Asia, semiarid in China, and arid in Mongolia and East Asia. Most regions of Asia are affected by the Asian-Australian monsoon which causes dry and wet seasons. Water erosion is the major type of erosion in the regions of South and East Asia with alternating dry and wet seasons. On the other hand, wind is the crucial driving force inducing soil erosion in the drier and desert areas.

Soil erosion by rainfall and surface water flow is generally affected by five factors: rainfall erosivity, soil erodibility, topography, surface coverage, and support practices. In humid regions, soil erosion is of little concern in well-established forests and in paddy fields. However, bare lands such as logged forests, construction areas and upland crop fields on slopes are exposed to a high risk of soil erosion. Annual soil loss in paddy fields has been generally reported in case studies to be lower than 1 tonnes $ha^{-1}$ (Chen, Liu and Chen, 2012; Choi *et al.*, 2012; Kim *et al.*, 2013). By contrast, soil loss from upland crop areas on slopes is much greater – for example, 38 million tonnes $ha^{-1}$ from fields in South Korea where no conservation practices were applied (Jung *et al.*, 2005). In semiarid regions, soil erosion is also of concern especially for slope areas with scant vegetation. In these areas, several hundred mm of rainfall in the rainy season can result in massive gully erosion. For these reasons, soil erosion is regarded as the most important threat to soil in Asia, especially for poorly-covered lands and bare soil.

Major threats of soil erosion by water are found in the hilly and mountainous landscapes of Indonesia. Natural conditions, anthropogenic influences on land cover and intensive land use make the steep and densely populated islands of Java, Sumatra and Sulawesi the most threatened areas. Approaches to coping with erosion problems range from engineering measures such as terracing, sediment pit construction and waterways improvement to vegetative measures including agroforestry approaches, contour strips and cover crops (Agus and Widianto, 2004).

In the Philippines, a monsoon country with high rainfall, erosion by water is one of the major causes of land degradation. In the 2013 Land Degradation Assessment final report by the Bureau of Soils and Water Management, the estimated annual soil loss for agricultural land for the whole country based on the 2003 Land Use System Map is about 61.8 tonnes $ha^{-1}$ $yr^{-1}$. Paningbatan (1987) estimates that soil loss of about 10 tonnes $ha^{-1}$ can be considered tolerable for Philippines conditions. In the 1950s, USDA (quoted by Schertz, 1983) established soil loss tolerance values for the Philippines at about 5 to 12 tonnes $ha^{-1}$ $yr^{-1}$ at soil bulk density of 1 200 kg $m^{-3}$. An analysis made by the Department of Environment and Natural Resources (DENR) on the state of the Philippine environment showed that, overall, 75 percent of total croplands are vulnerable to erosion of various degrees. To counter high rates of erosion, sustainable land management practices are being promoted. These include the application of various soil conservation and management strategies in highland agriculture as well as other technologies like agroforestry and multiple-storey cropping. The Philippines is a member of the UN Convention to Combat Desertification (UNCCD), and the Department of Agriculture and other government agencies, academia and non-government organizations aggressively pursue various programs. These approaches seek to engage farmers as partners in development rather than treat them as the cause of the problem.

In South Korea, deviation of annual precipitation was 251 mm $yr^{-1}$ between 1981 and 2010 and rainfall erosivity ranged from 2 264 MJ mm $ha^{-1}$ $yr^{-1}$ $hr^{-1}$ to 6 856 MJ mm $ha^{-1}$ $yr^{-1}$ $hr^{-1}$ in the same period (Park et al, 2011). The national average rainfall erosivity was 4 276 mm $ha^{-1}$ $yr^{-1}$ with EI 30 data between 1973 and 1996 (Jung *et al.*, 2004) and was 4 147 MJ mm $ha^{-1}$ $yr^{-1}$ $hr^{-1}$ with EI 60 data between 1981 and 2010 (Park *et al.*, 2011). The variation of rainfall is expected to increase with climate change based on the RCP scenario (CCIC, 2014), which implies that the probability of extreme rainfall erosivity would also increase in future. Soil erodibility in South Korea was 0.027 million tonnes hr $MJ^{-1}$ $mm^{-1}$ and ranged from 0.001 million tonnes hr $MJ^{-1}$ $mm^{-1}$ to 0.102 million tonnes hr $MJ^{-1}$ $mm^{-1}$ with soil series. The soil erodibility of paddy fields is the greatest at 0.036 million tonnes hr $MJ^{-1}$ $mm^{-1}$ followed by upland crop fields (0.026 million tonnes hr $MJ^{-1}$ $mm^{-1}$) and forests (0.020 million tonnes hr $MJ^{-1}$ $mm^{-1}$) (Jung *et al.*, 2004).

Differences in soil erodibility between land use types are affected by geological characteristics. Forests are located in mountains and upland crop fields are generally placed on lower slopes below the mountains. This means that forests and upland crop fields have been chronically exposed to past overland flow and soil erosion, which has resulted in their current status with less easily-eroded particles such as silt and very fine sand. By contrast, paddy fields in the bottomlands are formed from sediments with greater soil erodibility. However, actual annual soil loss in paddy fields has been generally reported to be lower than 1 million tonnes $ha^{-1}$ in case studies because paddy is protected from rainfall by the water already ponded in the field and by the ridges which store water inside paddy fields (Chen, Liu and Chen, 2012; Choi *et al.*; 2012, Kim *et al.*, 2013). Levels of soil erosion are tolerable in well-managed forests and grasslands (Kitahara *et al.*, 2000; Lee, 1994). Where grass was sown in spring, soil loss on grasslands was found to reach only half of that from bare soil in the first year. Losses decreased abruptly after the second year, to only 3 percent or less of those on bare soil (Jung, 1998; Jung and Oh, 1993).

Soil loss from upland crop fields is much greater than from paddy fields and forests. The national average of soil loss in upland crop fields was 38 million tonnes $ha^{-1}$ in South Korea (Jung *et al.*, 2005). A variety of conservation practices have been applied to reduce soil erosion including agronomic and engineering practices. On low slopes, agronomic practices such as mulching and grass strips can reduce soil loss by 80-90 percent. Engineering practices such as terraces, channels and drop spillways can reduce soil loss on steeper slopes (Jo *et al.*, 2009). Based on the Law of Water Quality Conservation and the Law of Soil Environment Conservation, soil erosion in South Korea has been monitored in severely eroded areas and best management practices have been proposed to regional governments and farmers.

### 10.4.2 | Soil organic carbon change

ASSOD describes soil organic carbon change as "a net decrease of available nutrients and organic matter in the soil". Not all Asian countries monitor the soil organic carbon (SOC) stock and its change. Even where soil properties are monitored, some data may not be appropriate for aggregation at the country scale as they lack certain parameters such as bulk density, or the number of observations may be insufficient. However, data compiled from the literature are adequate to draw a rough sketch of SOC change in the region. Piao *et al.* (2012) compared three methods to estimate SOC change 1990-2009 in five East Asian countries. Using an inventory-remote sensing model approach, they concluded that the sub-region was a net ecosystem carbon sink (+0.293 Pg C $yr^{-1}$). They found an estimated net SOC increase in the forest (+0.014 Pg C $yr^{-1}$), shrub land (+0.022) and cropland (+0.022) and SOC decrease in the grassland (-0.003).

China occupies the largest land area of the region (46 percent of the total). China reported SOC changes in the range -0.143 Pg C $yr^{-1}$ to +0.094 Pg C $yr^{-1}$ during 1980-2000. India occupies the second largest area (14 percent of the total). In India, it is estimated that forest accumulated SOC at the rate of +0.041 Pg C $yr^{-1}$ over the one hundred year period 1880-1981. The relevant data are listed in Table 10.1. Overall, there is a tendency towards SOC accumulation in forested areas and towards a decrease in grassland areas. To confirm these findings and allow analysis to guide future decisions, more detailed and comparable datasets for both SOC stock and change should be compiled in the region.

**Table 10.1 | Soil organic carbon change in selected countries in Asia**

| Sub-region/Country | | Area M ha | SOC stock Pg C | SOC stock Mg C $ha^{-1}$ | SOC change Pg C $yr^{-1}$ | Reference |
|---|---|---|---|---|---|---|
| East Asia sub-region | Total 1990 - 2010 | 1156 | | | 0.055 | Piao *et al.* (2012) |
| China | Total 1980 - 2000 | 871 | 89.6 | 102.9 | | Xie *et al.* (2007) |
| | | | 86.8 | | -0.143 | |
| | Total 1981 - 2000 | 760 | 85.8 | 112.9 | | Tian *et al.* (2011) |
| | | | 87.6 | | 0.094 | |
| | Total 2000 | | 101.9 | | 0.029 | Houghton and Hackler (2003) |
| | Total | | | | 0.075 | Piao *et al.* (2009) |
| India | Forest 1880 - 1981 | 64.2 | 6.8 | 106.1 | | Chhabra *et al.* (2003) |
| | | | 10.9 | | 0.041 | |
| | Total | | 47.5 | | | Velayuthum *et al.* (2000) |
| | Total | 329 | 24.0 | 73.1 | | Bhattacharyya *et al.* (2008) |
| Indonesia | Total | 183 | 20.8 | 113.4 | | Shofiyati *et al.* (2010) |
| Japan | Arable land 1979 - 1998 | 5.4 | 0.44 | 81.8 | | Takata (2010) |
| | | 5.0 | 0.45 | 89.6 | 0.0001 | |
| | Forest 1950s - 1990s | 24.3 | 2.18 | 90.0 | | Morisada *et al.* (2004) |
| | Total | 29.3 | 2.63 | 89.9 | | Takata (2010) and Morisada *et al.* (2004) |

China is a vast country – 6 percent of the global land area and 46 percent of the area of the region. Estimates in China of the totoal SOC storage in the 0-100 cm soil profile show high variability, ranging from 50 to 183 Pg (Xie *et al.*, 2007). The reasons for this variability include uncertainty about the area of croplands, the quantity of soil profiles and the methods applied for scaling up from soil profiles to a national level. Using data from the Second National Soil Survey carried out in the 1980s, Xie *et al.* (2014) estimated areas and SOC stocks as follows: paddy lands 29.87 million ha and 2.91 Pg C; uplands 125.89 million ha and 10.07 Pg C; forest 249.32 milion ha and 34.23 Pg C; and grassland soils 278.51 million ha and 37.71 Pg C.

Farmland and forests were found to have SOC sequestration rates of 23.62 and 11.72 Tg C yr –1, respectively, resulting in 0.472 and 0.234 Pg SOC respectively being accumulated in these soils during the period 1980 to 2000. However, degradation of grassland depleted 3.56 Pg C over the same period. Thus during twenty years, a net amount of 2.86 Pg C was lost, approximately 3.4 percent of total SOC storage in China.

With an artificial neural network model to link SOC change to six parameters - latitude, longitude, elevation, soil type, land use type, and original SOC in early 1980s - Yu *et al.* (2009) estimated an increase of 260 Tg C occurred in Chinese cropland in the period 1980 to 2000 in the topsoils (0-20 cm). The increase of SOC content is mainly attributed to the large increase in crop yields and the increased residues retained in fields. By contrast, SOC storage in grassland is dwindling, especially in the northwest and southwest parts of China, mainly due to the degradation of grassland. Nationally, an area of 5.29 million ha grassland had degraded 1986-1999 (Han and Gao, 2005). Unlike the grassland ecosystem, SOC storage in forestland has increased (see also above). On the forested area of 249.32 million ha, estimates of carbon sequestration range from 234 to 304 Tg SOC in the period of 1980 to 2000, mainly attributed to forest expansion and regrowth (Zhou *et al.*, 2006; Xie *et al.*, 2007).

Other studies show conflicting results, with some showing recent Chinese soils as a net carbon sink. Tian *et al.* (2011) estimated SOC change using two process-based terrestrial ecosystem models with considering factors including climate change and land use change. They concluded that biomass and soils in China accumulated organic carbon at a rate of 0.121 and 0.094 Pg C $yr^{-1}$ respectively from 1981 to 2000. Piao *et al.* (2009) adopted a bottom-up approach and showed consistent results (0.105 and 0.075 Pg C $yr^{-1}$, respectively) in the same period, while Houghton and Hackler (2003) reported net C loss from Chinese ecosystems (-0.008 PgC $yr^{-1}$) in 1990s, including loss in biomass (-0.028 Pg C $yr^{-1}$) and net C sink in soils (0.029 Pg C $yr^{-1}$).

Velayutham and Bhattacharyya (2000) estimated soil organic C stock in different soil orders and different agro-ecological regions in India (Sehgal and Abrol, 1992). For the top 1 m depth, they estimated soil organic C stock as 47.5 PgC, which is double the previous estimates of Dadhwal and Nayak (1993) and Gupta and Rao (1994). The trend may, however, be negative as crop residues are widely used as fuel and fodder and not returned to the soil, which would result in a decrease in soil organic content. In Bangladesh, the average organic content is said to have declined by half, from 2 percent to 1 percent, over the past 20 years (Bangladesh, 1992). For the Indian State of Haryana, soil test reports over 15 years show a decrease in soil carbon (Chaudhary and Aneja, 1991). Decreased organic content leads to: (I) degradation of soil physical properties, including water holding capacity, as has developed in India (Indian Council of Agricultural Research, personal communication); (II) reduced nutrient retention capacity; and (III) longer release of nutrients, including micronutrients, from mineralization of organic master. As a consequence of all these effects, there may be longer responses to fertilizer.

The threat to soil organic carbon (SOC) change in Indonesia's mineral and peat soils is mainly caused by deforestation, poor land management or intensive cropping, or by a combination of these factors (Hartanto *et al.*, 2003; Lal, 2004). In lowland rice grown on mineral soils, SOC tends to be maintained or increased because of anoxic condition (Kyuma, 2004). Land use types greatly influence soil loss. Annual crop based systems where soil conservation measures are not practiced are associated with high erosion

(Valentin *et al.*, 2008). For peatland, the threat to SOC occurs when peat forest is cleared and drained (Page, Rieley and Banks, 2011). The change from saturated to unsaturated conditions leads to the enhancement of aerobic microbial activities. This is the main factor in the SOC loss which occurs through the resulting accelerated microbial decomposition (Hooijer *et al.*, 2014; Agus and Subiksa, 2008). Peat fire is another cause of SOC loss. Maintaining a high water table is the key to reducing SOC losses through both these pathways. More details on the loss of soil carbon in Indonesia are included in the country case study section (10.5.2).

Permanent meadows and pastures area occupy 73 percent of the land area of Mongolia, while forest and arable land area account for 7 percent and 0.4 percent, respectively (FAO, 2012). During the 1990s and 2000s, the forest area decreased (loss of 8.19×102 $km^2$ $yr^{-1}$), and as a result forest biomass in Mongolia has decreased at a rate of 0.004 PgC $yr^{-1}$ (Piao *et al.*, 2009). Li *et al.* (2005) reported net ecosystem exchange in a Mongolian steppe under grazing using the eddy covariance technique and suggested that the steppe was almost carbon neutral.

Forest occupies 69 percent of the land area of Japan, while arable land and permanent meadows and pastures area account for 12 percent and 2 percent, respectively (FAO, 2012). Total SOC stock in forest area was estimated separately by Morisada, Ono and Kanomata, (2004) and Ugawa *et al.* (2012) using a bottom-up approach but with different data sets. Morisada *et al.* (2004) used 3 391 soil profile data sampled from the 1950s to the 1970s and calculated the weighted average SOC stock as 90 Mg C $ha^{-1}$ (0-30cm) and 188 Mg C $ha^{-1}$ (0-100cm). Ugawa *et al.* (2012) compiled 4 km mesh (around 3 000 profiles) data sampled from 1999 to 2003 and estimated average SOC stock as 69.4 Mg C $ha^{-1}$ (0-30cm). Since the Japanese forest area has changed little during the last 40 years, and has even increased slightly, the difference of SOC values between the two studies could be attributed to the difference in the methodology of sampling. Assuming the average SOC stock represented the total forest area (25.0×106 ha), total SOC stock in forest area would be in the range of 1.7 to 2.2 Pg C. Takata (2010) compiled soil survey data (1979-1998) to calculate SOC stock in Japanese arable soils, and reported that 0.44 Pg C in 1979 increased very slightly to 0.45 Pg C, mainly due to an increase of both area and stock per unit area in grassland.

### 10.4.3 | Soil salinization and sodification

As outlined above section (10.3.5), the threat of salinization/sodification in the region takes varying forms. In the semiarid and arid zones of central and west Asia, salt-affected soils are widely distributed. On the other hand, salt-affected soils are also developing in certain coastal areas in monsoon zones, caused mainly by salt water intrusion in South and Southeast Asia and by coastal tideland reclamation. Although the coastal area affected is relatively small, this could become a serious problem for lowland rice production.

In the GLASOD study, the region is estimated to have 42 million ha affected by salinization, nearly all of which is located in the dry zone. Of this salinized area in the drylands, there are estimated to be approximately 4 million ha in both India and Pakistan. Salinization is also a major problem on irrigated land: GLASOD estimates that 10 percent of irrigated lands in India are affected, 23 percent in Pakistan and 9 percent in Sri Lanka, although these percentages are probably overstated since some of the salinization results from saline intrusion into unirrigated land.

GLASOD provides estimates of areas subject to strong salinization. These numbers are important, as by definition they refer to land abandoned and taken out of cultivation. However, there is sometimes a wide difference between GLASOD figures for strongly saline soils and country estimates, although it should be noted that some of these include naturally occurring saline soils. For India, country estimates range between 7 and 26 million ha, all higher than the GLASOD value of 4 million ha. For Pakistan, there is better agreement; leaving aside three estimates of 9-16 million ha, the GLASOD and six country estimates lie in the range 4-8 million ha. Two apparently independent surveys, by the Soil Survey of Pakistan and the Water and Power Development Authority, show relative agreement at 5.3 and 4.2 million ha, respectively.

In Bangladesh, an extension inland of coastal soil salinity has been noted in recent years. Lower river flows, reduced by upstream abstraction for irrigation, have proved insufficient to dilute and displace sea water. In Sri Lanka, small areas of light salinization have appeared on irrigated lands of the Mahaweli scheme; the problem has not yet reached serious proportions, but needs to be monitored.

Estimates of the extent of saline soils need to be associated with the dates of survey. Through successful reclamation, the extent of saline soils has been reduced in some areas, particularly as a consequence of the series of Salinity Control and Reclamation Projects (SCARP) in Pakistan. For example in the Pakistan Punjab the area of waterlogged and saline soils, which had risen from 61 000 ha in 1960 to 68 000 in 1966, had been reduced to 23 000 ha by 1985 (Chopra, 1989).

Tideland reclamation projects have been carried out for centuries on the western coast of Korea. Records show that tideland reclamation in Korea began at the Ganghwa island in Gyeonggi-Do in 1235 (the 22nd year of King Gojong, Goryeo Dynasty). Reclamation in the early years was on a small scale, but has expanded over the years. Large scale modern reclamation projects started in 1960s as part of the national development program. Some of these projects have been as substitutes for the more than 20 000 ha of farmland which have been converted each year into industrial estates or other urban purposes. Since 1945, 75 738 ha of tideland have been reclaimed for paddy fields in 185 project areas by the Korean government and private companies (Park, 2001).

The main constraints to crop production on reclaimed tideland are soil salinity, a high water table with poor drainage, and an unfavorable soil chemical composition. Resalinization of the surface soil is caused by evapotranspiration during the dry season and by capillary rise of saline water from groundwater resources. High soil salinity in reclaimed tidelands needs to be managed by controlling the amount and quality of irrigation water (Jung, Joo and Yoon, 2002). Soil characteristics on reclaimed land change continuously as desalinization progresses. However, in general, newly reclaimed saline soils have poor chemical properties and weak physical soil characteristics of soil thickness, soil structure and water logging, so that it is difficult to grow crops economically (Park, 1991).

### 10.4.4 | Nitrogen imbalance

Balanced nutrient supply is essential for achieving high crop yields, but excessive or unbalanced nutrient inputs may pose risks to the environment, human health and ecosystems. Nutrient losses may occur via emission to the air or discharge to the water through runoff, leaching and erosion.

Nutrient inputs in Asia vary considerably amongst countries, districts and farming systems. Nutrient inputs tend to be higher than in other regions, and the trend may continue. FAO has predicted that for the coming four decades 60 percent of the world population increase will be in Asia (FAO, 2014). Feeding this rapidly growing population while minimizing harm to the environment is of great importance for both Asia and the world. Innovations in policy, science and farming practice are urgently needed to achieve this goal.

Nutrient cycles link agricultural systems to their societies and surroundings and create the need for decisions on tradeoffs. Inputs of nitrogen and other nutrients are essential for high crop yields, but downstream and downwind losses of these same nutrients diminish environmental quality and human well-being (Vitousek *et al.*, 2009). In this section, the issue of nitrogen imbalances in agriculture in Asia is highlighted as an important threat to soil.

One study found that nutrient balances differ among Asian countries, varying generally with the level of economic development (Vitousek *et al.*, 2009). The study examined six countries representing developing (China and India), developed (Japan and South Korea) and least developed countries (Laos and North Korea). China had the highest input of nitrogen (505 kg N $ha^{-1}$ of arable land), which is nearly ten times levels applied in Laos (59 kg N $ha^{-1}$). On the other hand, the mean N output was 108 kg N $ha^{-1}$ in China, while North Korea achieved half that level (44 kg N $ha^{-1}$). The N input was dominated by fertilizer application, especially in the booming economies of China and India, where the fertilizer N inputs accounted for 76 percent and 62 percent respectively of the total N inputs. In the least developed countries, such as Laos and North Korea, the N input mainly consists of animal manure, crop straw and biological N fixation. For developed countries, such as Japan and South Korea, the N inputs were more evenly contributed by fertilizer and manure. However, because of relatively high N input, higher N losses were observed in China, Japan and South Korea, compared with the least developed countries. The nitrogen use efficiency (NUE) was relatively high in Laos and North Korea, at the expense of depleting the soil N. The average soil N depletion rate ranged between 10–30 kg N $ha^{-1}$ $yr^{-1}$ in Laos and North Korea, while in China soil N accumulated by 26 kg N $ha^{-1}$ $yr^{-1}$.

In the least developed countries, manures were considered as precious nutrient resource for crop production. This was also the case in China before the 1980s, when artificial fertilizer was not subsidized. However, from the 1980s on, fertilizer was more widely used than manure in China. The amount of fertilizer applied in China far exceeds rates in other countries in Asia, and even exceeds rates in the United States and EU (Ju *et al.*, 2009; Vitousek *et al.*, 2009). In China in 2005 lack of regulation led to more than half of manure being discharged untreated to water bodies (Ma *et al.*, 2010). By contrast, the environment is highly protected in Japan. More than 80 percent of manure is treated before being applied to crop land (Mishima, 2012). The N inputs and outputs also showed large variations between different crops. The highest nutrient inputs and accumulation in cash crop production were observed in China (Yan *et al.*, 2013). As both livestock and cash crop production are expected to increase considerably in Asia in the future as demand rises from a fast growing and increasingly urbanized population (FAO, 2014), N use is likely to increase but there may be a more intelligent and sustainable approach to N inputs and outputs.

Nitrogen use efficiency (NUE) is defined by the N output as crop production divided by the N inputs that are added through fertilizer, biological N fixation and N deposition (Ma *et al.*, 2010). In a study across Asia and the Middle East, national average NUE ranged from 131 percent in Laos to 19 percent in the United Arab Emirates.

The NUE was more than 100 percent in Laos, Nepal and Myanmar, because these countries relied to a great extent on recycled N inputs such as manure and crop straw. The NUE showed a reverse trend of N surplus among countries, decreasing from high N depletion countries such as Nepal and Laos to the high N surplus countries such as the United Arab Emirates and China. None of the crop production systems were sustainable in high N depletion or surplus countries. Although lower N losses and higher NUE were observed in the least developed countries, the continuous N depletion will limit crop yields and food production. In N surplus countries, the high levels of N accumulation may lead to higher N losses and consequently to serious environmental problems (Guo *et al.*, 2010; Liu *et al.*, 2013).

The average N losses varied among countries, from 23 kg N $ha^{-1}$ $yr^{-1}$ in Afghanistan to 327 kg N $ha^{-1}$ $yr^{-1}$ in China (Figure 10.3). This was due to the differences in N input rate, the crop production structure and the area of arable land. For a variety of reasons, large areas of arable land are not presently cultivated in Afghanistan, Iraq, Mongolia and Syria. The contribution of the different N loss pathways varied among countries, for example, ammonia emission contributed to 50 percent of the N losses in Philippines, but to only 22 percent of the N losses in Laos.

In conclusion, N inputs and outputs vary considerably amongst countries in Asia, with variations mainly attributable to levels of development and to policies. The nutrient imbalance in Asia could have a large impact on crop production and on the environment. These impacts may increase in the future with the further rapid development of livestock and cash crop production. Both the N imbalance and the N losses can be improved greatly without any sacrifice of crop yield. For example, balanced N fertilizer application and maximum manure N application standards have already proved effective in the EU (Velthof *et al.*, 2009). Recent studies show that crop yields in China can be increased by 20-30 percent with no increase in input of fertilizer simply by using ISSM technologies (Chen *et al.*, 2014). However, in the least developed countries increased use of fertilizer would greatly boost crop yields, as has happened in China during the past 30 years. For the developed countries, promoting mixed crop and livestock production systems could be the best choice to mitigate nutrient losses

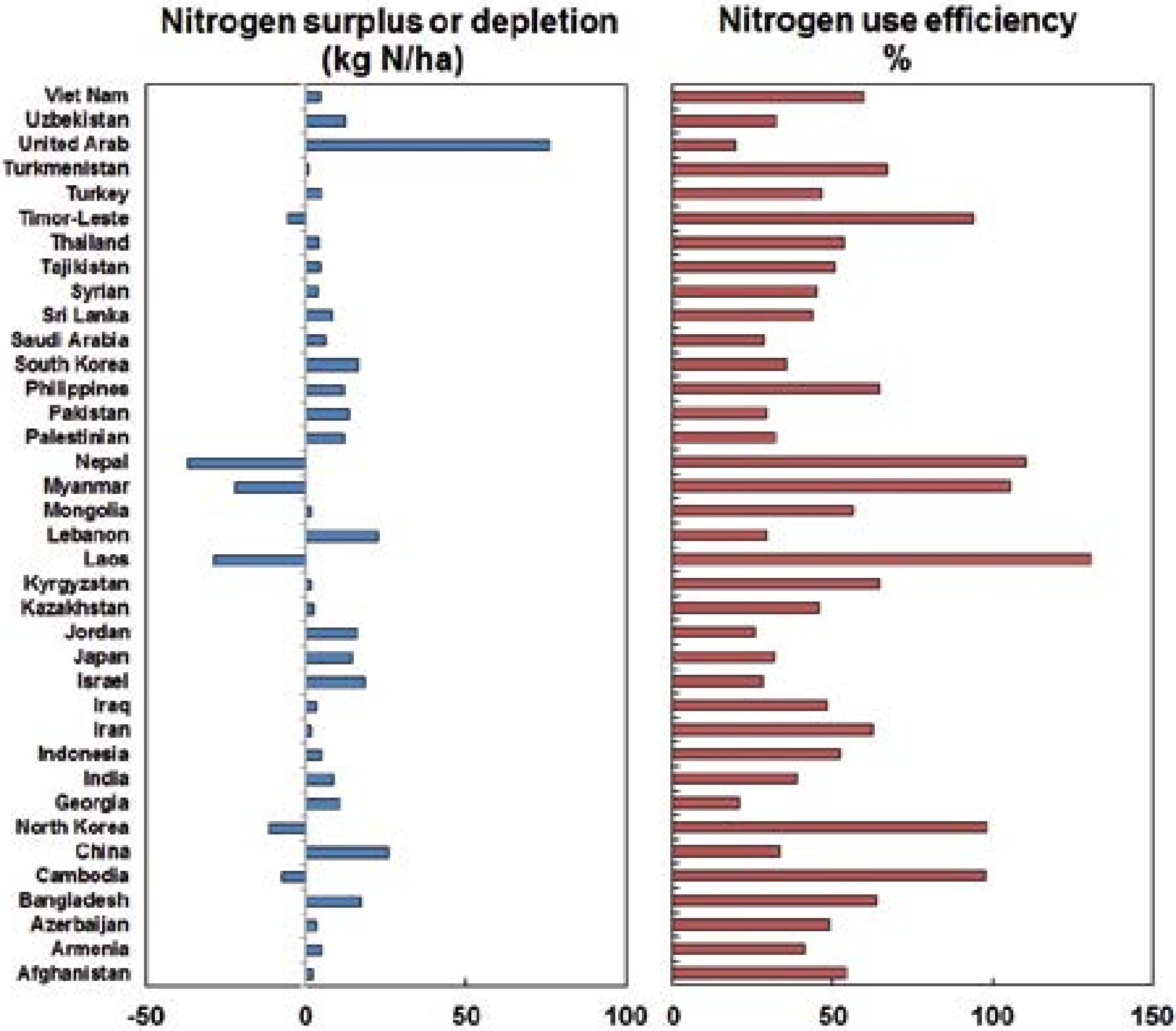


Figure 10.3 | Nitrogen surplus or depletion, and nutrient use efficiency in crop production in Asia and the Middle East in 2010.

## 10.5 | Case studies

### 10.5.1 | Case study for India

The area of India is 328.2 million ha, of which 141 million ha is under cultivation. The country is bordered by the Arabian Sea in the west, the Bay of Bengal in the east, and the Himalayas in the north. Physiographically, India is divided into four broad divisions: (I) Himalayan Range (Northern Mountains); (II) Hill regions, Indian Peninsula and Eastern Plateau; (III) the great Indo-Gangetic Plains and Coastal Plains; and (IV) the Islands (Singh, 1971). India is endowed with diverse climates and there are three distinct main seasons (rainy, winter and summer). The country is influenced by monsoonal type climate and rainfall. Annual rainfall varies from less than 100 mm in the cold desert area of Jammu & Kashmir, Lahaul Spiti and the Thar Desert of Rajasthan to over 11 000 mm in Cherrapunji in Meghalaya. Mean annual temperatures in the country vary from 8 to 28°C. Variation in the mean summer and mean winter temperature in the northern region is <10°C and in the south <5°C. In all, 20 Agro-Ecological Regions (AERs) have been identified, subdivided into 60 Agro-Ecological

Sub-Regions. Inceptisols are the dominant soils covering 39.75 percent of the total area, followed by Entisols (28.08 percent), Alfisols (13.55 percent), Vertisols (8.52 percent), Aridisols (4.28 percent), Ultisols (2.51 percent), Mollisols (0.4 percent) and others (2.92 percent) (Bhattacharyya *et al.*, 2013).

### Degraded and wastelands of India

Velayutham and Bhattacharyya (2000) reported that the total area subject to soil degradation in India is 45.9 percent. Of this, 37.0 percent is affected by water erosion, followed by wind erosion (4.0 percent), salinization (2.2 percent), loss of nutrients (1.1 percent) and waterlogging (1.6 percent). Land not fit for agriculture (ice-caps, salt-flats, arid mountain and rock outcrops) constitutes 5.5 percent of the total area. About 27.5 percent of soils have no degradation problem and the remaining 9.8 percent of the area is classed as 'stable terrain', e.g. natural conditions. Recently the National Academy of Agricultural Sciences (NAAS) inventoried the soil degradation status of the country based on reconciliation of databases gathered from different organizations (Figure 10.4 and Table 10.2). This confirmed that soil erosion by water is the most serious threat, affecting 82.57 percent of the total area, followed by wind erosion (12.40 percent), acidic soils (17.94 percent), salt-affected soils (6.74 percent), waterlogged soils (0.88 percent) and mining and industrial waste (0.19 percent).

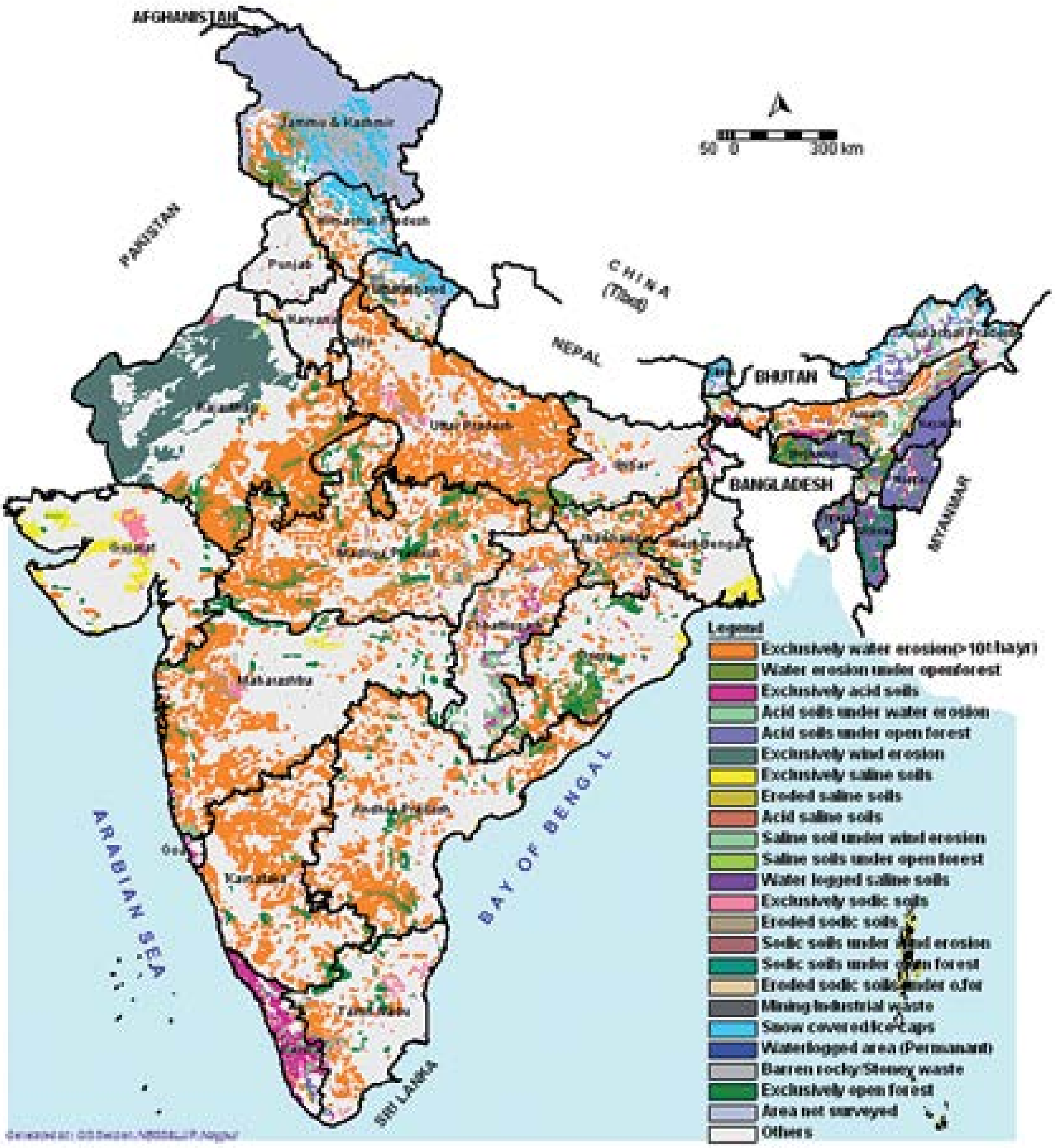


Figure 10.4 | Degradation and wastelands map of India.
Source: ICAR and NAAS, 2010.

**Table 10.2 | Harmonized area statistics of degraded and wastelands of India. Source: ICAR and NAAS, 2010.**

*Notes: FSI (1999) was used to exclude degraded land under dense forest; Unculturable Wastelands: Barren rocky/stony waste: 6 M ha, are the source for runoff water and building material; Snow covered/Ice-caps: 6 M ha, are best source of water and are not treated as wastelands*
*#For acid soils, areas under paddy growing and plantation crops were also included in the total acid soils*
*[\$]Sub-surface waterlogging not considered.*

| Degradation type | Arable land (million ha) | Open forest (<40 percent canopy) (million ha) | Data source |
|---|---|---|---|
| Erosion | | | |
| Water erosion (>10 tonnes $ha^{-1}$ $yr^{-1}$) | 73.27 | 9.30 | Soil Loss Map of India–CSWCR&TI |
| Wind erosion (Aeolian) | 12.40 | – | Wind Erosion Map of India–CAZRI |
| **Sub-total** | **85.67** | **9.30** | |
| Chemical degradation | | | |
| Exclusively salt- affected soils [#] | 5.44 | | Salt-Affected Soils Map of India, CSSRI, NBSS&LUP, NRSA and others |
| Salt-affected and water eroded soils | 1.20 | 0.10 | |
| Exclusively acidic soils (pH< 5.5) [#] | 5.09 | - | Acid Soil Map of India NBSS&LUP Acidic (pH < 5.5) and water |
| Acidic soils (pH< 5.5) and eroded soils | 5.72 | 7.13 | |
| **Sub-total** | **17.45** | **7.23** | |
| Physical degradation | | | |
| Mining and industrial waste | 0.19 | | Wasteland Map of India–NRSA |
| Waterlogging (permanent surface inundation) [\$] | 0.88 | | |
| **Sub total** | **1.07** | | |
| **Total** | **104.19** | **16.53** | |
| **Grand total** (arable land and open forest) | **120.72** | | |

## 10.5.2 | Case study for Indonesia

Topography, climate, soil parent materials and anthropogenic factors determine the various types and degree of soil threats in Indonesia. These threats include soil carbon depletion both for mineral and organic soils, erosion by water, soil contamination, soil acidification and nutrient imbalance. Soil organic carbon depletion in peat is the most significant threat.

Soil erosion threats are found in almost all hilly and mountainous landscapes of the Indonesian archipelago due to the very high (>2000 mm) annual rainfall over most of the area (83 percent). Most other sloping areas with lower rainfall are also affected by erosion due to the high intensity of the monsoonal rainfall during the rainy season (Agus, Amien and Sutono, 2002). Besides causing erosion, the high rainfall also leads to leaching of basic cations and hence to soil acidification. The main problems associated with managing acid soil are low pH, P-fixation, low basic cation concentration, low cation exchange capacity (CEC) and toxicity of soluble Al and Fe.

The total area of Indonesian acid upland soils (pH <5 and <50 percent base saturation) is about 102.8 million ha (Puslitbangtanak, 2000). Relatively low cost technologies are widely available for rectifying soil acidity problems. These include liming, organic matter application, balanced fertilization, and selection of acid tolerant crops (Kamprath, 1984; von Uexküll and Mutert, 1995). Nutrient imbalance is related to insufficient or unbalanced fertilizer use (Tan, Lal and Wiebe, 2005). The use of nitrogen fertilizer in various farming systems in Indonesia has been continually increasing since the late 1960s, reaching 2.4 Mg $yr^{-1}$ in 2012. However, the application of P and K has not followed the same trend (IFADATA, 2007). This is in part because N fertilizers have been easier to obtain, they are cheaper, and they give a more rapid crop response to the farmer. For intensive cultivation of vegetable crops, over-fertilization has led to accumulation of N, P, Ca, Mg and Na, but to negative balances of K and Si (Husnain, Masunaga and Wakatsuki, 2010; Widowati *et al.*, 2011). K deficiency was also reported in newly-developed rice fields (Sukristiyonubowo, Nugroho and Ritung, 2012). For low input systems, nutrient removal through harvest is seldom sufficiently replenished by proper fertilization.

### Soil organic carbon depletion

Depletion of soil organic carbon (SOC) is considered as one of the major threats. In extreme cases, it contributes to the physical, chemical and biological degradation of the soil. The rate of SOC depletion in mineral soils is relatively small. In peat soils, organic C is the main soil constituent and the rate of depletion is high for drained organic soils (IPCC, 2014; Agus, Hairiah and Mulyani, 2011).

#### *The loss of carbon from mineral soil*

Research findings differ on the trend of soil organic matter depletion in Indonesia (Agus *et al.*, 2013). One study (Murty *et al.*, 2002) suggested that SOC stock decreased by about 30 percent when forest is converted to continuous agricultural production; forest transformed to degraded land would lose 50 percent of its C stock, while forest converted to plantation would lose about 30 percent of its C stock. Another study found that SOC stocks in the 0–15 cm soil profile of Dipterocarp forest in Sumatra decreased by 48.1 Mg $ha^{-1}$ when the forest degraded to *Imperata grassland*. However, Imperata grasslands are not necessarily associated with low SOC. In some cases, this grassland may have similar C content to forest (Santoso *et al.*, 1997). For Sumatra, van Noordwijk *et al.* (1997) found some decrease in SOC when forest is converted to cropland. Tanaka *et al.* (2009), on the other hand, did not observe significant differences in SOC between secondary forests, 9 and 19 year old oil palm plantations and a 30 year old rubber plantation.

Management systems can restore SOC. Under intensive management, it was found that in the first and second cycles of oil palm, organic matter increased 32 percent and SOC 15 percent in the 0-45 cm soil layer, relative to secondary forest as the initial land use (Mathews, Tan and Chong, 2010). Regular application of palm fronds increased SOC, especially in the 20 percent of the plantation area receiving an equivalent of 4.8 Mg C $ha^{-1}$ $yr^{-1}$ from palm fronds (Haron *et al.*, 1998). If degraded land is converted to plantation, it was found to gain about 30 percent SOC stock (Murty *et al.*, 2002; Germer and Sauerborn, 2008). In East Kalimantan, natural regeneration from Imperata grassland to secondary forest increased soil carbon content by 14 percent, from 14.5 g $kg^{-1}$ to 16.5 g $kg^{-1}$ (van der Kamp, Yassir and Buurman, 2009). In Java, agricultural top soil with continuous rice cropping accumulated more than 1.7 Tg C per year over the period of 1990-2000 (Minasny *et al.*, 2012). Kyuma (2004) also suggested that SOC in newly established paddy rice soils tends to increase with time. Strategies to increase the soil carbon pool include soil restoration and woodland regeneration, no-till farming, cover crops, nutrient management, manuring and sludge application, improved grazing, water conservation and harvesting, and efficient irrigation (Lal, 2003).

***The loss of carbon from organic soils***

Indonesian peatland is estimated to cover about 14.9 million ha (Ritung *et al.*, 2011). It is a massive store of carbon, storing around 27 Pg C (Agus *et al.*, 2013). This huge C store is formed under saturated condition in concave areas with the annual rate of peat formation in the range of 0-3 mm thickness (Noor, 2001) or equivalent to zero to about 1.5 Mg C ha $yr^{-1}$ carbon accumulation (Agus and Subiksa, 2008; Parish *et al.*, 2007). This relatively slow process has led to the formation of peat domes that began between 6 800 and 4 200 years ago (Andriesse, 1994). Some formations may be as old as 26 000 years (Page *et al.*, 2002).

Carbon stock in peat ranges from 420 to 820 Mg C $ha^{-1}$. Peat depths range from 0.5m to over 10 m (Agus, Hairiah and Mulyani, 2011) and the peat C stock is strongly determined by peat thickness (Warren *et al.*, 2012). Under saturated natural conditions, peat C is slowly emitted in the forms of $CO_2$ and $CH_4$ due to anaerobic microbial activities. As peat forest is cleared and drained, the peats become unsaturated and $CO_2$ emission escalates at a pace far exceeding the rate of sequestration. Currently, from the 14.9 million ha classified as peatland, forested areas amount to just over half (52 percent), shrub cover is about 21.7 percent, and the rest is under agriculture or settlements. Peat soil C loss runs in parallel to the level of local development. Peatland areas of Sumatra and Kalimantan have been drained on a wide scale, and hence are fast losing carbon. On the other hand, Papua peatlands are mostly conserved (Figure 10.5).

The major processes of carbon loss from drained peatland are (I) peat decomposition under aerobic condition and (II) peat fire. Natural phenomena such as lengthy droughts aggravate peat fire risks (Page *et al.*, 2002; IPCC, 2014; Parish *et al.*, 2007; Agus and Subiksa, 2008; Wosten *et al.*, 2008).

The literature varies in its estimates of peat carbon loss through decomposition. One source (IPCC, 2014), based on research data from Malaysia and Indonesia, presented peat $CO_2$-C emission factors based on land cover classes. In this study, degraded forest is expected to emit as much as 5.3 Mg $CO_2$-C $ha^{-1}$ $yr^{-1}$, while various agricultural and forestry uses emit a wide range of $CO_2$-C as shown in Table 10.3. The 95 percent confidence interval was very wide for each land cover class, which suggests the need for site specific emission factors. A case study in Riau Province (Husnain *et al.*, 2014) showed insignificant differences in peat emissions under an oil palm plantation, an Acacia plantation, a secondary forest and a rubber plantation. The emission rates were 18.0 ± 6.8; 16.1 ± 5.3; 16.6 ± 6.8; 14.2 ± 4.6 Mg $CO_2$-C $ha^{-1}$ $yr^{-1}$, respectively. For bare land sites, the rates measured lay between 15.2 ± 8.2 and 18.2 ± 6.5 Mg $CO_2$-C $ha^{-1}$ $yr^{-1}$. The findings of other studies (Marwanto and Agus, 2014; Dariah, Marwanto and Agus, 2014) were similar to those of the IPCC (2014), with emissions from oil palm plantations on peat soils of about 10 to 11 Mg $CO_2$-C $ha^{-1}$ $yr^{-1}$.

A study in Central Kalimantan Province revealed a linear relationship between average water table depth and peat $CO_2$ emission. Degraded drained forest was found to emit about 2.7 Mg $CO_2$-C $ha^{-1}$ $yr^{-1}$ in areas with zero mean water table depth, and about 15 Mg $CO_2$-C $ha^{-1}$ $yr^{-1}$ for areas where average water table depth was 1 m (Hooijer *et al.*, 2014). Regardless of the variations found amongst the studies, the values recorded demonstrate rapid carbon depletion in drained peatland.

Peat fire is another important process that may cause a huge amount of carbon loss in a short period of time, and the literature is in agreement on the importance of peat fire as one of the main causes of peat carbon loss (Page *et al.*, 2002; Hooijer *et al.*, 2014). However, determination of peat C loss through peat fire is a future research challenge, especially with regards to gathering firm data, for example on the volume (area and depth) of burnt scars (IPCC, 2014).

**Table 10.3 | Emission factors of drained tropical peatland under different land uses and the 95 percent confidential interval. Source: IPCC, 2014.**

*1) Emission of primary peat forest is assumed to be zero.*
*2) Some confidence intervals contain negative values because calculation was based on error propagation of uncertainties. However, all underlying $CO_2$ fluxes were positive.*

| Land cover type[1] | Emission ($Mg\ CO_2$-$C\ ha^{-1}\ yr^{-1}$) | 95 percent Confidence level[2] ($Mg\ CO_2$-$C\ ha^{-1}\ yr^{-1}$) | |
|---|---|---|---|
| Drained forest land and cleared forest land (shrubland) | 5.3 | -0.7 | 9.5 |
| Plantations, drained, unknown or long rotations | 15 | 10 | 21 |
| Plantations, drained, short rotations, e.g. acacia | 20 | 16 | 24 |
| Plantations, drained, oil palm | 11 | 5.6 | 17 |
| Plantations, shallow-drained (typically less than 0.3 m), typically used for agriculture, e.g. sago palm | 1.5 | -2.3 | 5.4 |
| Cropland and fallow, drained | 14 | 6.6 | 26 |
| Cropland, drained, paddy rice | 9.4 | -0.2 | 20 |
| Grassland, drained | 9.6 | 4.5 | 17 |


Figure 10.5 | Indonesian peatland map overlaid with land cover map as of 2011.
Source: Wahyunto et al., 2014.

## 10.5.3 | Case study for Japan

The islands which make up Japan are located in the one of the most active parts of the Circum-Pacific Ring of Fire. The major islands are Hokkaido, Honshu, Shikoku and Kyushu. There are 110 active volcanoes in Japan (16 volcanoes in Hokkaido, 38 volcanoes in Honshu and 11 volcanoes in Kyushu). Large quantities of tephras from volcanoes have been deposited on the Pleistocene terraces, constituting a main parent material of Japanese soil (Andosols). Japan's total land area is about 378 000 km$^2$. About 72 percent of Japan's land area is mountainous, and rivers are characterized by their steep gradients and relatively short lengths. About two-thirds of the total land area consists of forest. The plains cover only about 28 percent of the total land area. Most plains are located along the seacoast. Arable lands account for 12.1 percent of the total land surface, mainly distributed in the plains. The climate of Japan is influenced by a monsoonal flow that carries moist air from the Indian Ocean and Pacific Ocean. In general, Japan has four distinct seasons: spring (March to May), summer (June to August), autumn (September to November) and winter (December to February).

Japan's arable land covers 4.5 million ha, with paddy fields accounting for just over half the area (54 percent). Grey lowland soils (Fluvic Hydragric Anthrosols or Gleyic Fluvisols (FAO/IUSS/ISRIC, 2006) comprise the largest cultivated soil area; followed by Gley soils (Gleyic Fluvisols), Andosols (Aluandic or Silandic Andosols), Brown forest soils (Haplic Cambisols), Brown lowland soils (Haplic Fluvisols), and Wet Andosols (Gleyic Andosols). Urban sprawl and other changes in land use (including abandonment of cultivation) led to a shrinking of the agricultural land area by about 1 million ha between 1973 and 2001. Urbanization and consequent soil sealing advanced into flat lowland areas, largely into paddy fields on Grey Lowlands soils and Gley soils. In addition, loss of Andosols to expanding urbanization was widely observed over the flat upland fields in the middle part of Hunshu islands. By contrast, upland fields on steep slopes in the western part of Japan, largely on distributed Brown Forest Soils, were simply abandoned (Takata *et al.*, 2011b).

### Soil organic carbon change

Spatio-temporal variations in soil organic carbon (SOC) content in arable land were evaluated by both model-based (Yagasaki and Shirato, 2014) and monitoring-based (Takata, 2010) approaches. In the model-based approach, SOC stock change was simulated using the original Rothamsted Carbon model (Coleman and Jenkinson, 1996) and two modified Rothamsted Carbon models (Shirato, Yagasaki and Nishida, 2011; Takata *et al.*, 2011a). The rate of change in the total SOC stock in Japanese agricultural lands evaluated with 10 year intervals was estimated to be -0.95 Tg C yr$^{-1}$ between 1980 and 1990. A greater loss of SOC, equal to -1.06 Tg C yr$^{-1}$, was found subsequently for the period from 1990 to 2000.

An agricultural soil monitoring project named 'Basic Soil-Environmental Monitoring in Japan' has been conducted since 1979. This soil monitoring project has taken readings at repeated five year intervals at about 20 000 fixed points. The rate of change in the total SOC stock evaluated by this monitoring was 2.3 Tg C $yr^{-1}$ from 1979 to 1989. The project found that SOC increased in arable land. However, the monitoring detected a greater loss of SOC for a later period (1989-1998), equal to -2.3 Tg C $yr^{-1}$. During this period, the agricultural land area decreased from 5.4 to 5.0 million ha. At the same time, soil carbon content in arable land gradually rose from 88 to 90 tonnes C $ha^{-1}$. These results indicated that the decline of SOC stock in Japanese arable land 1989-1998 was mainly influenced by the fluctuation in the arable land area.

Spatial variation of SOC content in forest soils has been monitored since 2005. The mean SOC content of forest soil at 0-30cm was 69.4 tonnes C $ha^{-1}$ (Ugawa *et al.*, 2012), lower than the value of 90 tonnes C $ha^{-1}$ in arable land.

### Heavy metal contamination

Rapid industrialization in Japan during the 1960s polluted arable soil with heavy metals such as cadmium (Cd), Copper (Cu), and Arsenic (As). There were four main pollution sources: mining activity, factories and incinerators, fertilizer, and precipitation and irrigation water (Makino, 2010). In 1970, the Japanese government enacted the Agricultural Land Soil Pollution Prevention Law to regulate heavy metal pollution. The law designated Cd, As, and Cu as hazardous substances to be regulated. The allowable limitation of Cd was set in terms of the Cd concentration in rice grains (1 mg $kg^{-1}$). The allowable limitations on As and Cu were set to 15 (1M HCl soluble) and 125 (0.1M HCl soluble) mg $kg^{-1}$ soil, respectively. The amount of bioavailable Cd in soil is affected by many factors, so setting an allowable concentration in terms of the soil Cd content is impractical (Asami, 1981). The area of polluted arable land was assessed as 7 592 ha (Cd, 7 050 ha; Cu, 1 405 ha; As, 391 ha). About 92 percent of the total polluted area has been remediated by uncontaminated soil dressing (MOE, 2014).

### Radioactive Cs contamination

As a result of the accident at the Fukushima Dai-ichi Nuclear Power Station (FDNPS) operated by the Tokyo Electric Power Company, radioactive cesium (Cs) was released into the surrounding environment. To determine the extent of decontamination required in arable land and to consider management options, the Ministry of Agriculture, Forestry and Fishery (MAFF) surveyed and measured soil Cs concentrations in 3 461 agricultural fields, and used these data to construct a distribution map of radioactive Cs concentration in agricultural soil in eastern Japan (Takata *et al.*, 2014). The distribution map of radioactive Cs concentration in agricultural soil is shown in Figure 10.6.

Contamination level 4 (>25 000 Bq $kg^{-1}$) was observed only in a 20 km evacuation area (EA) surrounding FDNPS (EA-20km) and the Deliberate Evacuation Area (DEA); 77 percent of the level 4 areas were observed in the EA-20km. Farmers of level 4 contaminated fields were advised to solidify their topsoil with a fixation

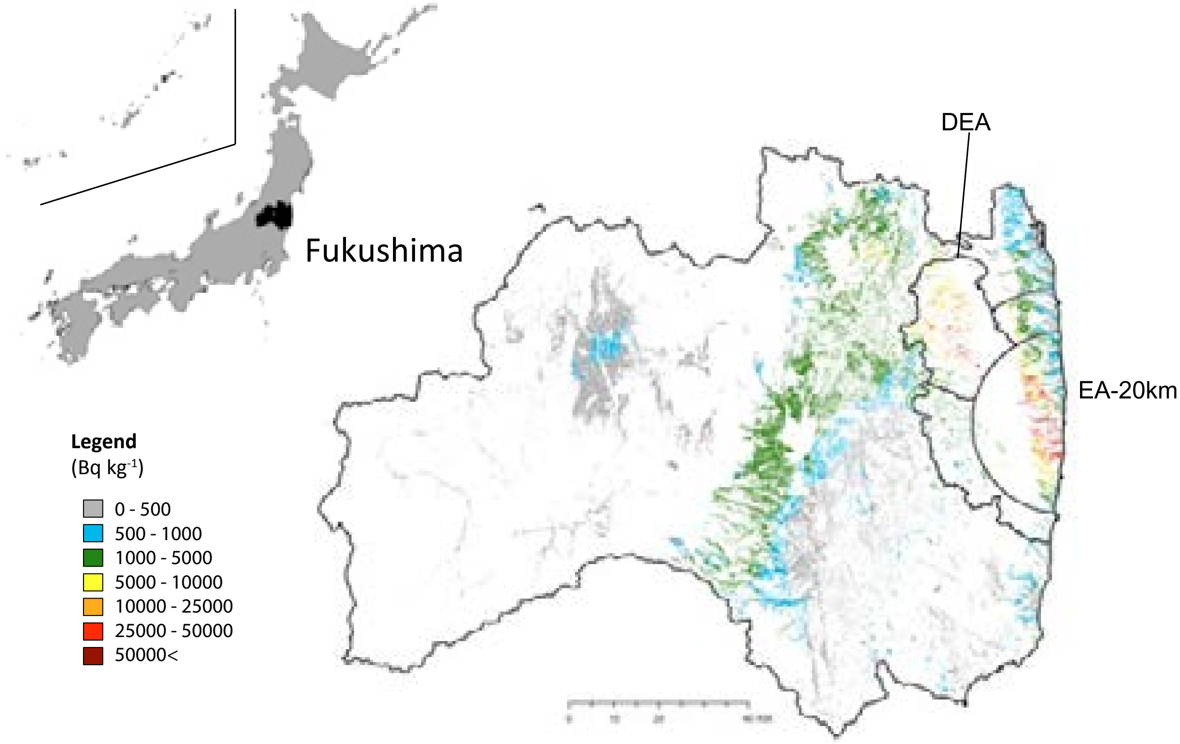


**Figure 10.6 | Distribution map of radioactive Cs concentration in soil in Fukushima prefecture (reference date of 5 November, 2011). Source: Takata et al., 2014.**

agent to prevent scattering of contaminated soil during a topsoil removal operation. Contamination level 3 (10 000–25 000 Bq $kg^{-1}$), which indicated a need to remove topsoil, was distributed in only the EA-20km and DEA, with more than half (55 percent) in the EA-20km. Eighty percent of fields with contamination level 2 (5 000–10 000 Bq $kg^{-1}$) were distributed in the evacuation zone, with the remaining 20 percent distributed in the non-evacuation zone in Fukushima Prefecture. Paddy fields with contamination level 2 (2 100 ha) have three options for decontamination: topsoil removal, fine-textured topsoil removal using water, and topsoil burying. Upland fields, orchards, and meadows that are at contamination level 2 (1 200 ha) have two options for decontamination: topsoil removal and topsoil burying.

## Nutrient imbalance

The soil surface nitrogen (N) and phosphate (P) balance in Japanese arable land has been improving (Mishima, Endo and Kohyama, 2010a; 2010b). These values serve as the index of the impact of arable land on the environment and of the farm-gate balance of nutrients. The soil surface N (and P) balance is defined as the total N (P) input (N kg $ha^{-1}$) minus the total N (P) output (N kg $ha^{-1}$). Chemical fertilizer application in Japan declined continuously during the period from 1985 to 2005. The application rate of livestock manure also peaked in 1990 and declined thereafter. Crop production, however, remained constant during this period. Between 1985 and 2005, the surplus N and P (positive value of soil surface N balance) declined from 89.9 to 49.3 kg N $ha^{-1}$ and from 153 to 105 kg P $ha^{-1}$ respectively (Mishima, Endo and Kohyama, 2010a, 2010b). However, this trend was not consistent at the regional level because organic amendment applications were largely related to the availability and movement of livestock excreta (Mishima, Endo and Kohyama, 2010a) and to soil type (Leon *et al.*, 2012). High surplus P and low crop P uptake compared with N, P input for crop production could be reduced. This limited negative environmental effects such as eutrophication of soil and water and conserved limited P resources.

The Basic Soil-Environmental Monitoring Project found excess soil Ca in paddy fields, upland fields and orchards during 1979-1998 (MAFF, 2008), but also a gradual increase in soil Mg deficit over the same period. Thus the balance of Ca and Mg has been deteriorating in Japanese arable land. In addition, the soil pH of paddy fields gradually decreased from 5.8 to 5.7 during 1979-1998 (MAFF, 2008).

### Soil erosion

Spatial estimation of soil loss from arable land at national scale was carried out using the Universal Soil Loss Equation (USLE) and environmental inventories (Kohyama *et al.*, 2012). Hourly rainfall and runoff factors (R: resolution 1 km) were calculated by the amount of precipitation analysed by radar-AMeDAS. Topographic factors (LS) are shown in Figure 10.7, calculated using a digital elevation model (resolution 50 m) and ALOS satellite imagery. The soil erodibility factor (K) of arable land was calculated using the physico-chemical soil properties (soil texture, soil organic matter content, etc.) as measured in the Basic Soil-Environmental Monitoring Project (Taniyama, 2003). The K factor of arable land was determined by soil series groups. The K factor was relatively higher in clayey lowland soil group than in humic Andosol groups. The cover and management factor (C) of each crop was determined by Taniyama (2003), and was delineated using the agro-environmental census data map (Kohyama *et al.*, 2003).

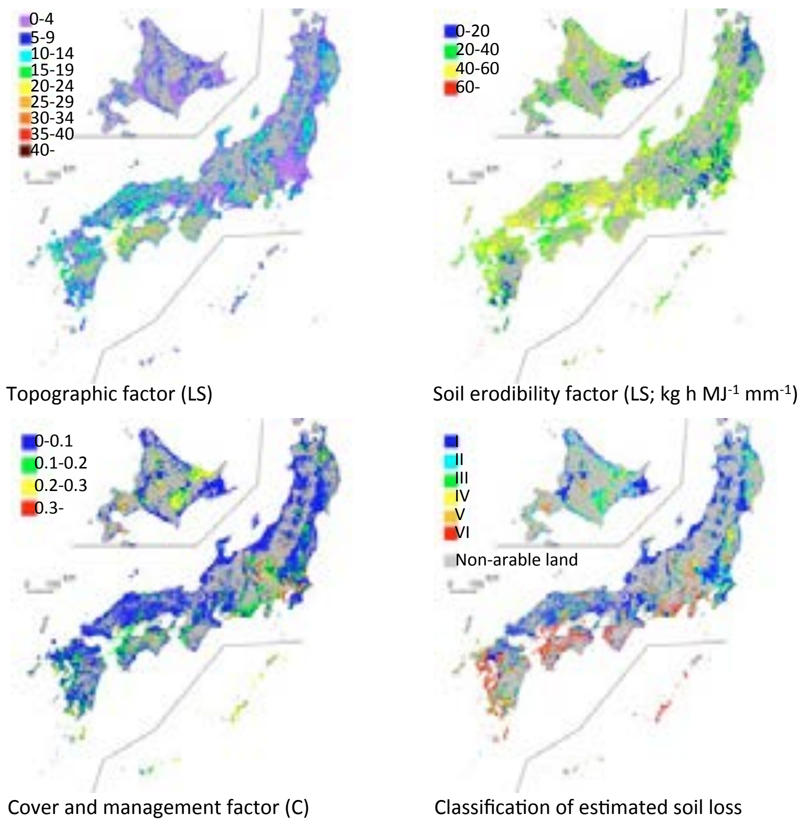


Figure 10.7 | Distribution map of the parameters of USLE and classification of estimated soil loss. Class I: less than 1 tonnes ha$^{-1}$ yr$^{-1}$; Class II: 1-5 tonnes ha$^{-1}$ yr$^{-1}$; Class III: 5-10 tonnes ha$^{-1}$ yr$^{-1}$; Class IV: 10-30 tonnes ha$^{-1}$ yr$^{-1}$; Class V: 30-50 tonnes ha$^{-1}$ yr$^{-1}$; Class VI: more than 50 tonnes ha$^{-1}$ yr$^{-1}$.
Source: Kohyama et al., 2012.

Loss of soil in Japanese arable land was categorized into six classes: Class I; less than 1 tonnes $ha^{-1}$ $yr^{-1}$, Class II; 1-5 tonnes $ha^{-1}$ $yr^{-1}$, Class III; 5-10 tonnes $ha^{-1}$ $yr^{-1}$, Class IV; 10-30 tonnes $ha^{-1}$ $yr^{-1}$, Class V; 30-50 tonnes $ha^{-1}$ $yr^{-1}$, Class VI; more than 50 tonnes $ha^{-1}$ $yr^{-1}$. The proportion of soils in these classes was: 43 percent in Class I; 18 percent in Class II; 9 percent in Class III; 12 percent in Class IV; 5 percent in Class V; and 14 percent in Class VI. Highly erodible zones were mainly distributed in areas of western Japan which are characterized by complex topography and heavy precipitation.

## 10.5.4 | Case study of greenhouse gas emissions from paddy fields

Rice is the staple crop for the majority of the world's population. In Asia, rice cultivation areas roughly account for 89 percent of the global total (Yan, Akimoto and Ohara, 2003). While rice production is thus vital for feeding the world's population, it is also an important source of greenhouse gas emissions, notably methane ($CH_4$) and nitrous oxide ($N_2O$). $CH_4$ is converted from substrate by methanogenic bacteria in strictly anaerobic environments, while $N_2O$ is an intermediate production of nitrification and denitrification. Both of these two gases possess considerably greater infrared absorbing capability than carbon dioxide ($CO_2$) on a mass basis: 25 times for $CH_4$ and 298 times for $N_2O$.

Using the tier 1 method described in the 2006 Intergovernmental Panel on Climate Change (IPCC) Guidelines for National Greenhouse Gas Inventories (IPCC, 2006), and country-specific estimates of rice harvest area and data on agricultural activities, Yan and colleagues estimated that global $CH_4$ emission for 2000 was 25.6 Tg $CH_4$ $yr^{-1}$, with a 95 percent uncertainty range of 14.8 to 41.7 Tg $CH_4$ $yr^{-1}$, considerably lower than earlier estimates (Yan *et al.*, 2009). Rice paddies in monsoon Asia countries contributed far and away the largest share of these emissions, estimated at 23.7 Tg $CH_4$ $yr^{-1}$. China, with an amount of 7.41 Tg $CH_4$ $yr^{-1}$, was estimated to be the largest $CH_4$ emission country, followed by India, Bangladesh, Indonesia, Vietnam, Myanmar and Thailand. The areas with the greatest emission intensity were the delta regions of large rivers in Bangladesh, Myanmar and Vietnam, the island of Java in Indonesia, central Thailand, southern China and the southwestern portion of the Korean peninsula (Figure 10.8).

$CH_4$ emission from rice fields is the net result of three processes: production, oxidation and transport. Three main factors affect one or more of these processes: organic amendment, the water regime during the rice-growing season, and water status in pre-season. Appropriate agricultural management of organic amendments and the water regime should therefore be promoted to mitigate $CH_4$ emissions. Techniques could include off-season straw incorporation and midseason drainage (Yan *et al.*, 2009).

Unlike $CH_4$, $N_2O$ emission from rice paddies has been found to be much lower than from upland crops, because under a flooded environment, nitrification is weak and denitrification proceeds to the end step with $N_2$ as the dominant product. The majority of $N_2O$ emissions from rice paddies usually occur shortly after the mid-season drainage and nitrogen fertilizer additions (Yan *et al.*, 2000). The availability of substrates and soil moisture condition are critical controlling factors for $N_2O$ emission since they affect the activity of nitrifiers and denitrifiers. The fertilizer-induced $N_2O$ emission factor for rice fields averages approximately 0.3 percent, probably fluctuating dependent on the water regime (Akiyama, Yagi and Yan, 2005). Mitigation of $N_2O$ emissions from paddy fields should also not be overlooked because of the excessive nitrogen fertilizer consumption by rice production. Since a tradeoff relationship between $CH_4$ and $N_2O$ emissions from rice paddies is frequently observed, any mitigation options pursued should take their comprehensive global warming potential fully into account.

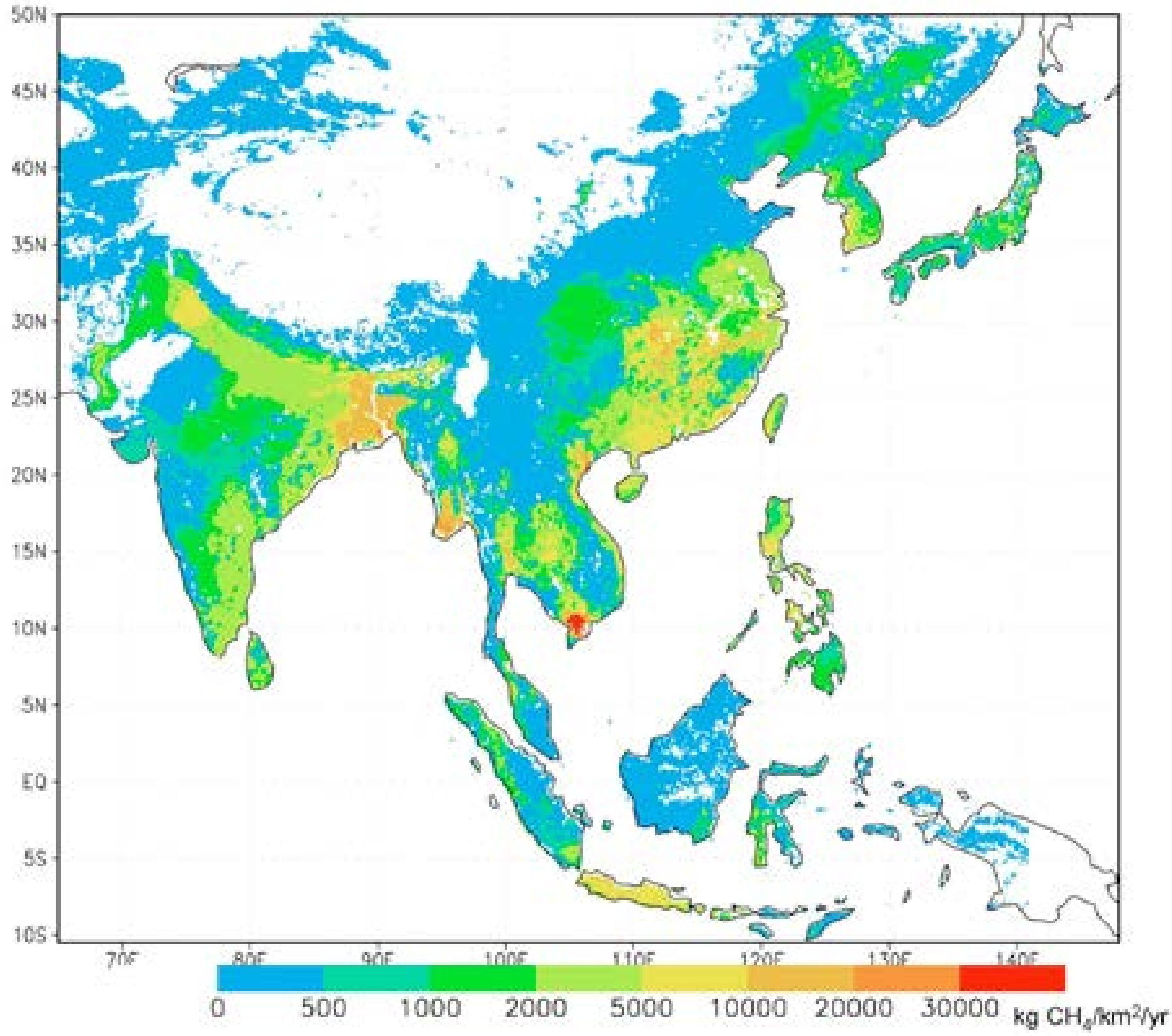


Figure 10.8 | Estimate $CH_4$ emission from rice paddy in Asia.
Source: Yan et al., 2009.

## 10.6 | Conclusion

In Asia, management of land and water resources has been identified as one of the priority ways to achieve sustainable food security by raising land productivity, reversing land degradation and water loss, and increasing biodiversity and the quality of the environment. Asian countries have also committed themselves to strengthening regional cooperation and national capacities to develop a more integrated approach to the management of natural resources. An integrated approach is needed to improve the ability of countries to plan and monitor the better use and management of their land resources to increase agricultural productivity while maintaining land and environmental quality.

However, since the GLASOD and ASSOD projects of the 1980s and 1990s, no extended assessment of the status of soil resources has been carried out in the region. There have been extensive scientific communications amongst experts in the region, including the activities of the East and Southeast Asia Federation of Soil Science Societies (ESAFS). Based on the above finding, a provisional assessment is made of the status and trend of the 10 soil threats in order of importance for the region. At the same time an indication is given of the reliability of these estimates (Table 10.4).

However, a number of the country reports that contributed to this chapter emphasized that rapid socio-economic change and resulting changes in land use and its management, as well as climate change, have had great impacts on the soil resources in Asian countries. Therefore, there is a need to conduct a new and extensive assessment of changes in soil resources in the region.

Responding to this need, a regional conference on soil information was held in Nanjing, China, on February 2012 to share the latest soil information and knowledge about advanced science and technologies on soil resources in Asian region. The conference recognized the benefits to be gained from further sharing of information and data on soil surveys, soil mapping and capacity development. The conference saw the establishment of the initial Asia Soil Partnership (ASP) and the signing of the Nanjing Communiqué (GSP, 2012), which put the following goals as the priorities:

1. sharing and transferring soil knowledge and new technology within and beyond the region
2. providing soil information to all those with an interest in the sustainable use of soil and land resources
3. building consistent and updated Asian soil information systems and starting to contribute to the global soil information system through initiatives such as GlobalSoilMap.net
4. training new generations of experts in soil science and land management

After the endorsement of the global Plans of Action in GSP, the next step in the Asian Region is the development of the regional implementation plan for sustainable soil management, which can translate the plans in the Nanjing Communiqué into practice.

Stable =

Variable ↗↙

Improving ↗

Deteriorating ↙

Evidence and consensus is **low**

Evidence and consensus are **limited**

**Adequate** high-quality evidence and **high level** of consensus

**Table 10.4 | Summary of Soil Threats Status, trends and uncertainties in Asia**

| Threat to soil function | Summary | Condition and Trend | | | | | Confidence | |
|---|---|---|---|---|---|---|---|---|
| | | Very poor | Poor | Fair | Good | Very good | In condition | In trend |
| Soil erosion | Serious water erosion occurs in regions with dry and wet seasons covering South Asia to East Asia, particularly in the hilly and mountainous landscapes. However, it is of little concern for well-established forests and paddy fields.<br>Wind erosion is concentrated mainly in the most western and northern arid and semi-arid regions of Afghanistan, Pakistan, India, and China. | | ↙ | | | | | |
| Organic carbon change | Increase in crop yield retains soil organic carbon (SOC) in croplands of East and Southeast Asia. Whereas, SOC is decreasing in South Asia, because crop residues are widely used as fuel and fodder, and not returned to the soil.<br>The degradation of grassland has caused great losses of SOC stock. | | ↗↙ | | | | | |
| Salinisation and sodification | The threat of salinisation/ sodification in the Asia region is widespread but variable. In semiarid and arid zones of central Asia, salt-affected soils are widely distributed. On the other hand, salt- affected soils are developed in certain coastal areas in monsoon zones, mainly by salt water intrusion in South and Southeast Asia. | | ↗↙ | | | | | |
| Nutrient imbalance | Negative soil nutrients balances have been reported for N, P, K and micronutrients in many…" South Asian countries. Whereas, large excess of nutrients, in particular N, causes serious environmental problems in other countries. | | ↙ | | | | | |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Contamination | Rapid urbanization, industrialization, and intensive farming causes contamination of heavy metals (Cd, Ni, As, Pb, Zn, etc.) and pesticides in various parts of Asia, which, in turn, poses a serious risk to human health. | | ↙ | | | | | |
| Soil sealing and land take | Rapid urbanization and development of mega-cites significantly increased the rate of impervious surface area (ISA). Asia region has the largest ISA within the global regions. | | ↙ | | | | | |
| Soil acidification | There is substantial area of acid soils distributed in tropical and subtropical regions of Asia, mainly in Southeast Asia, parts of East and South Asia. This is mainly caused by unbalanced and unsuitable application of chemical fertilizers. Distribution of acid sulphate soils in tropical Asia also limits crop production. | | ↙ | | | | | |
| Compaction | Mechanization of land management has increased compaction of surface soil and/or subsoil in cropland, grassland and timber forests. Increase in livestock trampling is also a major cause of surface soil compaction in grassland and hilly region. | | ↙ | | | | | |
| Waterlogging | Anthropogenic activates such as poor drainage system and deforestation in the upstream areas increase the threat to waterlogging in the flood prone areas. | | | ↙ | | | | |
| Loss of soil biodiversity | Limited information is available for soil biodiversity in Asia. Some reports show high microbial biodiversity in the soils of organic farming lands. | | | ↗↙ | | | | |

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