Food and Agriculture Organization of the United Nations

VOLUME 5

RECARBONIZING GLOBAL SOILS

PRACTICES OVERVIEW

A technical manual of recommended management practices

FORESTRY, WETLANDS, URBAN SOILS

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INTERGOVERNMENTAL TECHNICAL PANEL ON SOILS

GLOBAL SOIL PARTNERSHIP

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Managed forests and silviculture

1. Harvest systems that limit soil disturbance and reduced impact logging

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1. Description of the practice

Conventional harvesting operations in forests are inevitably associated with the disturbance of the forest soil. The use of ground-based heavy machinery, including skidders, forwarders, or tractors, can severely damage forest soil (Photo 1). Machinery movement through the forest can cause soil mixing, compaction, and erosion with long-lasting consequences for biological, chemical, and physical soil properties (Horn *et al.*, 2007; Cambi *et al.*, 2015). On steep terrain, rut formation can result in drastic losses of humus and mineral soil (Labrière *et al.*, 2015; Naghdi *et al.*, 2016a). On flat terrain, soil compaction and rutting can lead to a decrease in soil aeration and water infiltration (Goutal, Renault and Ranger, 2013; Bonnaud *et al.*, 2019). Soil disturbance due to logging can be limited if alternative harvesting systems are applied, such as, for example, helicopter-, cable yard-, or animal logging (Bockheim, Ballard and Wellington, 1975; Miller and Sirois, 1986; Aust and Lea, 1992; Naghdi *et al.*, 2009). Adverse effects on soil properties may be reduced by changes in machinery configuration as well, such as reducing tire pressure, installing larger diameters tires, and increasing the number of axles (Solgi *et al.*, 2020). Soil disturbance can also be reduced if networks of logging and skidding roads are properly planned prior to operations (e.g. adequate density) and are planned permanently (e.g. for future harvests) (Ampoorter *et al.*, 2012; Picchio *et al.*, 2020), logging is conducted during seasons with frozen and/or dry soil conditions (Ampoorter *et al.*, 2012; Naghdi *et al.*, 2016b), or slash is placed over skid trails as a means to distribute the load of machinery (Agherkakli *et al.*, 2014). In tropical forests, improvements for forest harvesting operations have been proposed under the term “reduced impact logging” (RIL), which includes measures to reduce environmental impacts of harvesting operations (Putz and Pinard, 1993; Healey, Price and Tay, 2000; Putz *et al.*, 2008).

2. Range of applicability

Harvesting systems to limit soil disturbance and RIL can be applied globally. Most countries around the world are choosing a combination of spatial (permanent skidding trails or extraction tracks) and temporal (frozen or dry soils) limitations of forest traffic to permanently limit the surface disturbed. Alternative harvesting systems are rarely chosen due to higher costs.

3. Impact on soil organic carbon stocks

Most peer-reviewed literature and meta-analyses have focused on the effects of intensive harvesting practices such as whole tree harvest and intensive biomass removal on soil organic carbon (SOC). The effects of alternative harvest systems and RIL systems on SOC losses and -stocks have only rarely been investigated and results are still inconclusive. For a dipterocarp forest in Sabah, Malaysia, RIL reduced SOC loss after logging by 4 tC/ha when compared to conventional logging (Putz and Pinard, 1993). In contrast, no differences in SOC stocks could be shown for conventional logging and RIL in a rain forest in Southern Cameroon (Tchiofo Lontsi *et al.*, 2019). Likewise, no differences in soil organic matter content where found for helicopter and skidder logged stands in Alabama, United States of America (Aust and Lea, 1991). Other studies compared harvesting systems regarding on-site soil surface damage and soil erosion losses. Particularly soil erosion can be associated with local SOC loss (translocation) from the harvest units (Berhe *et al.*, 2018). In a pine forests of Mississippi, United States of America, for instance, Miller and Sirois (1986) report that cable yard logging damaged 16 percent of the soil surface, while ground skidding damaged 31 percent. In boreal forest stands of British Columbia, Canada, tractor logging resulted in 71 percent exposed mineral soil (i.e. organic layer displaced), while helicopter logging resulted only in 5 percent exposed mineral soil (Bockheim, Ballard and Wellington, 1975); highest percentages of exposed mineral soil were determined for steep slopes and shale-rich soils. In a pine forest of Honduras with slopes greater than 30 percent, cable yard- and animal logging led to six to ten times less erosion than ground-based harvesting (tractor) during the rainy season (Rivera, Kershner and Dobrowolski, 2010). In a tropical forest of Malaysia, 17 percent of the area of a logged stand was covered by roads and skid trails, while under RIL guidelines only 6 percent of the area was similarly disturbed (Pinard, Barker and Tay, 2000); additionally, skid trails with disturbance of subsoil were less than 50 percent compared to regularly logged stands. Similarly, RIL resulted in 4 to 6 percent lower soil disturbance than conventional logging in a Brazilian rain forest (Pereira *et al.*, 2002). Worrell, Bolding and Aust (2011) report higher potential soil erosion rates (0.5 t/ha/yr) in steep Appalachian hardwood forests when comparing conventional skidding to cable yarding operations. In Hyrcanian forests of Iran, skidding on steep slopes (> 20 percent) resulted in higher soil disturbance and forest floor mass loss than skidding on less steep slopes (Naghdi *et al.*, 2016a).

4. Other benefits of the practice

4.1. Improvement of soil properties

Conventional harvesting systems using ground-based machinery increase soil bulk density, soil CO_2 concentration, waterlogging and runoff, and decrease soil porosity, water infiltration and permeability, air permeability, oxygen supply, and root and tree growth (Cambi *et al.*, 2015). Compared to conventional systems, alternative harvesting systems have be shown to cause smaller changes in chemical and physical soil properties, including saturated hydraulic conductivity, acidity, oxygen concentration, redox potential (Aust and Lea, 1992). More specifically, forwarder movement was shown to result in water logging and to decrease soil aeration in comparison to cable yarding systems (Goutal, Renault and Ranger, 2013; as a consequence, soil CO_2 efflux (Goutal *et al.*, 2012b), CH_4 absorption capacity (Epron *et al.*, 2016), earthworms abundance and diversity (Bottinelli, Capowiez and Ranger, 2014), tree regeneration, growth and rooting depth (Goutal-Pousse, Boc and Ranger, 2014) were reduced.

4.2 Minimization of threats to soil functions

Table 1. Soils threats

Soil threats	
Soil erosion	In general, harvesting systems that reduce soil disturbance may mitigate soil erosion. For example, cable yarding reduced potential soil erosion by 0.5 t/ha/yr (Worrell, Bolding and Aust, 2011) or by a factor 6 to 10 Rivera, Kershner and Dobrowolski, 2010).
Nutrient imbalance and cycles	In general, harvesting systems that reduce soil erosion are beneficial for nutrient balance and cycles. For example, reduced soil erosion rates due to cable yarding were shown to also maintain soil nutrient stocks (Worrell, Bolding and Aust, 2011). Reduced waterlogging maintain soil nutrients cycling (biological activity, rooting intensity and roots activity, redox potential) (Bonnaud *et al.*, 2019).
Soil biodiversity loss	Conventional harvesting systems have been shown to reduce and alter abundance of the microbiota, decrease macrofauna diversity, and increase bacterial diversity (Bottinelli, Capowiez and Ranger, 2014; Hartmann *et al.*, 2014).
Soil compaction	Cable yarding system and RIL can decrease compaction (Goutal *et al.*, 2012a; Tchiofo Lontsi *et al.*, 2019). Animal logging has been shown to decrease compaction and rutting (Horn *et al.*, 2007). The extent of compaction within a stand during both conventional and alternative harvesting systems may be reduced by operating during dry or frozen

Soil threats	
	conditions, limiting the number of passes, limiting activity to skid trails, and proper training and supervision of the operator (Piccio *et al.* 2020).
Waterlogging	Compared to skidder logging, alternative harvest systems can reduce changes in saturated hydraulic conductivity (Aust and Lea, 1992). On sandy soils, harvesting associated compaction can increase water holding capacity and increase productivity over at least one decade (Powers *et al.*, 2005).

4.3 Increases in production (e.g. food/fuel/feed/timber)

Reduced impact logging systems have positive effects on tree sapling density, canopy cover, plant species richness, and aboveground biomass stocks (Putz and Pinard, 1993; Pinard, Barker and Tay, 2000). Alternative harvesting systems have positive effects on forest regeneration (seedlings survival and growth) (Picchio *et al.*, 2020). Across 26 sites from the Long-term Soil Productivity (LTSP) experiment in North America (https://www.fs.fed.us/psw/topics/forest_mgmt/ltsp/), the effect of compaction on forest productivity was mediated by soil texture; a 40 percent increase in productivity was observed following severe compaction on sandy soils while a nearly 50 percent decline in productivity was observed following severe compaction on clayey soils (Powers *et al.*, 2005). However, these changes in productivity were attributed to changes in water holding capacity and gas exchange rather than SOC or nutrient cycling.

4.4 Mitigation of and adaptation to climate change

Reduced impact logging can significantly reduce overall logging emissions (Ellis *et al.*, 2019; Griscom *et al.*, 2019). Additionally, RIL could be shown to lower N_2O emissions from soil (Mori, Imai and Kitayama,2018). Skid trails impacted by ground-based harvesting displayed higher GHG fluxes by reducing CH_4 oxidation and enhancing NO_2 emissions (Warlo *et al.*, 2019).

4.5 Socio-economic benefits

Alternative harvesting systems often incur additional costs and require more trained operators than conventional ground-based harvesting systems. However, many alternative systems and RIL measures can be financially competitive if all the costs and benefits are considered or equalization payments for intangible benefits (e.g. biodiversity) are provided. Cable yarding, for example, allows wood to be harvested any time of the year (just-in-time wood extraction) whereas ground-based harvesting should be avoided during conditions of poor soil bearing capacity (e.g. when soil is wet). Moreover, the affected soil surface area of cable yarding systems is usually lower compared to ground-based harvesting systems (Photo 1, Figure 1), with positive effects on forest productivity and regeneration. RIL can be shown to decrease the amount of wasted wood in harvesting operations and can be competitive with conventional logging if the saved costs are taken into account (Boltz, Holmes and Carter, 2003).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

There is no available information on tradeoffs with other soil threats.

5.2 Increases in greenhouse gas emissions

For tropical forests, Ellis *et al.* (2019) estimated that selective logging emitted 834 Tg CO_2 in 2015, representing 6 percent of total tropical GHG emissions. A full implementation of RIL measures, including reduced wood waste, narrower haul roads, and lower impact skidding equipment, would reduce emissions from logging by 366 Tg CO_2/yr (= 44 percent reduction). Based on eddy covariance measurements in tropical rainforest in Brazil, Miller *et al.* (2011) showed RIL to have only minimal effects on total CO_2 efflux to the atmosphere.

6. Recommendations before implementation of the practice

No information available

7. Potential barriers to adoption

Table 2. Potential barriers to adoption

Barrier	YES/NO	
Social	Yes	In some countries, limited number of cable yarding companies (Magaud, 2020) constrains wider application of this practice. Moreover, many alternative harvesting systems are applicable only under specific circumstances.

Barrier	YES/NO	
Economic	Yes	RIL (Boltz, Holmes and Carter, 2003) and alternative harvesting systems such as cable yarding and animal skidding (Schweier and Ludowicy, 2020) can be less profitable than conventional ground-based harvesting .
Knowledge	Yes	Training and knowledge of individual contractors and operators is imperative to reduce forest damage (Picchio, Mederski and Tavankar, 2020), for proper implementation of RIL (Putz *et al.*, 2008), and for reducing soil disturbance during conventional cable yarding or conventional harvest (Chase *et al.*, 2019).

Photos of the practice

Photo 1. Soil disturbance (rut formation and soil compaction) from conventional skidder logging in Austria (top left) and France (top right). Cable yarding operation in the Austrian Alps (bottom)

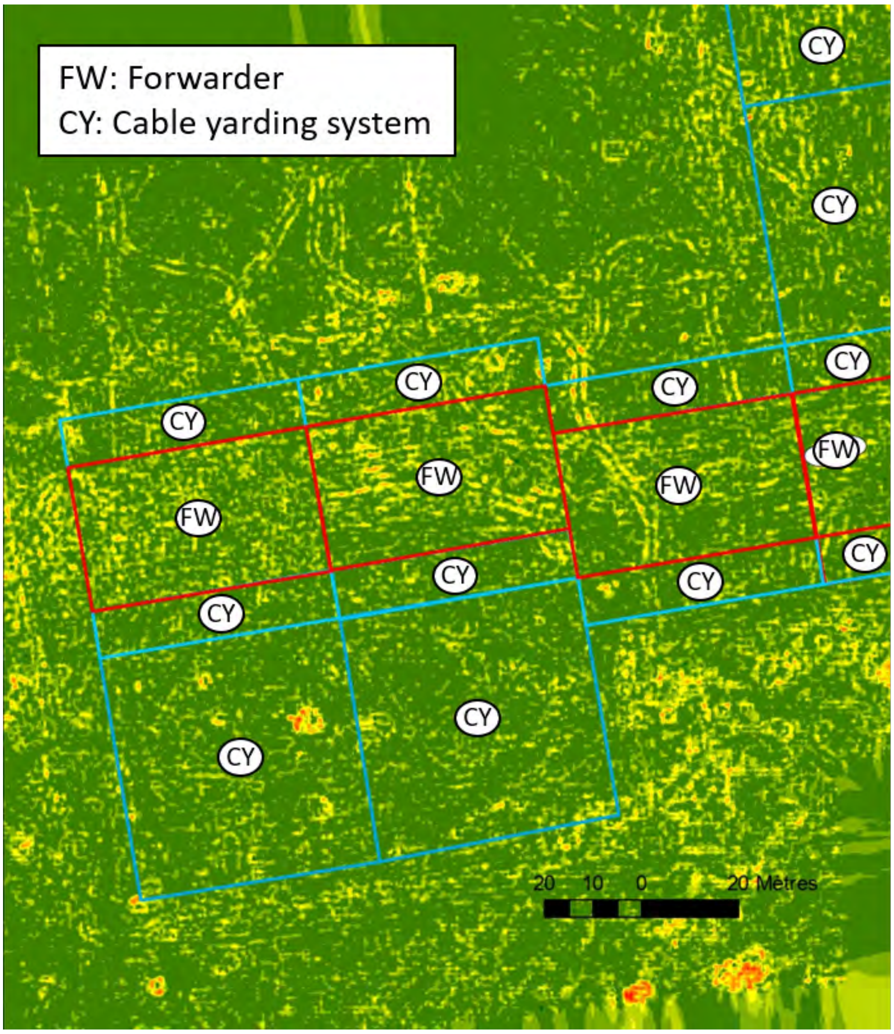

Figure 1. Digital elevation model (LIDAR based) showing the soil impact of logging operations by a 23 t forwarder compared to logging operations using a cable yarding system. Colors yellow and green indicate low and high elevations, respectively. Figure modified after Goutal-Pousse, Boc and Ranger (2014)

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2. Continuous cover forestry and extended rotations

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1. Description of the practice

Continuous cover forestry (CCF; see the full description in Helliwell and Wilson, 2012) includes many silvicultural systems which all involve continuous and uninterrupted maintenance of forest cover and which avoid clearcutting. It implies forest management that works with the characteristics of the site and with tree species that are well adapted to the location. It respects the processes inherent to the site, rather than imposing artificial uniformity, and will normally involve a mixture of tree species and ages. Management is based on the selection and favouring of individual trees (of all sizes) rather than the creation of areas of uniform tree size and spacing, and record keeping is based on periodic recording of stem diameters on sample areas, rather than by age and area of stands. Stand structure will be permanently irregular, although the process of transformation to an uneven-aged condition might involve temporary even-aged elements, possibly including small-scale clearfells, and group or irregular shelterwoods (Helliwell and Wilson, 2012). CCF could minimize soil disturbance because a larger portion of tree roots are preserved following wood harvesting, and because no soil preparation – such as ploughing– is done. On the other hand, it involves a greater number of soil trampling events as interventions are more frequent. CCF also requires more frequent and more technical interventions of the forest managers. CCF may limit changes to the soil microclimate due to smaller openings comparatively to clear-cutting with potential influence on soil organic matter decomposition. However, positive as well as negative impact of large canopy openings have been observed on organic matter decay rates (Mayer *et al.*, 2020).

2. Range of applicability

Applicability is worldwide, wherever even-aged forestry is practiced. Typically, over the past two centuries, conventional forest management approaches have favored the plantation of even-aged, single-species stands. Interest in alternative management approaches that involve continuous and uninterrupted maintenance of forest cover have greatly increased in many regions, particularly in developed economies (Puettman *et al.,* 2015).

3. Impact on soil organic carbon stocks

Although precise data on SOC changes are scarce, meta-analyses have revealed that clearcut harvesting results in reductions of < 10 percent of the soil C in the entire soil profile with greatest loss of the forest floor (Johnson, 1992; Johnson and Curtis, 2001; Achat *et al.*, 2015). In two meta-analyses of studies in temperate forests, forest harvesting reduced total soil C by an average of 6-8 percent: C storage declined by 22-30 percent in the forest floor, whereas the mineral horizons showed no significant overall change (Nave *et al.*, 2010; Achat *et al.*, 2015). Evidence that CCF reduces soil C losses in comparison to clear cutting is scant. Mayer *et al.* (2020) reported on the following studies: In Norway-spruce-dominated stands in Austria, single-tree-selection management resulted in 11 percent greater soil C stocks in the upper mineral soil compared to conventional even age-class management (Potzelsberger and Hasenauer, 2015). However, short-term losses were observed in shelterwood cuts in Chilean Patagonia (Klein *et al.*, 2008). In an oak-hardwood forest in New England, Warren and Ashton (2014) reported a decrease in the soil C stocks in the mineral soil, but neutral effects in the litter layer following shelterwood harvest. Others have found little or no difference between effects of partial, selection, shelterwood, and clearcut harvesting on soil C stocks (Hoover, 2011; Christophel *et al.*, 2015; Puhlick *et al.*, 2016). When differences in SOC content were observed, they were higher under CCF but of low magnitude (Pötzelsberger and Hasenauer, 2015; Jonard *et al.*, 2017). Similarly, two meta-analyses (Liao *et al.* 2010, 2012) showed a systematic loss of SOC in planted even-aged forests compared to naturally regenerated forests, but this difference in SOC storage could be linked to the fact that the naturally regenerating forests in these studies are partly primary forests, with SOC stocks probably at high level. In summary, information is too fragmentary to attribute any soil C changes with the adoption of CCF in replacement of a traditional even-aged silviculture system (Powers *et al.*, 2011). Local information on the effect of this practice on soil erosion, soil disturbance, as well as on impact on forest composition would be factors to consider due to their potential impact on soil C stocks. It is noteworthy that CCF systems involve light but more frequent interventions that could make changes in the soil C stocks at the whole rotation scale difficult to detect statistically.

4. Other benefits of the practice

4.1 Minimization of threats to soil functions

Table 3. Soil threats

Soil threats	
Soil erosion	On sites sensitive to erosion, maintaining a forest canopy as well as tree root systems could prevent erosion.
Nutrient imbalance and cycles	Especially if the adoption of this practice promotes mixed species stands.
Soil acidification	The absence of clearcut in CCF might slightly reduce the losses of cations induced by water leaching enhanced by clearcuts in sites prone to such losses (i.e. mainly soils with a high drainage regime but with a low buffering capacity of pH).
Soil biodiversity loss	In general, positive impact on biodiversity noting that some forest species need open canopy conditions or high disturbance levels. But the latter (i.e. ruderal species) are usually not a concern for biodiversity (Puetman *et al.* 2015).
Soil compaction	In general, the maintenance of the tree root systems might increase the resistance of soils to compaction. However, the current knowledge about this possible effect is scarce.
Soil water management	CCF enables better regulation of the water fluxes at the watershed scale.

4.2 Increases in production (e.g. food/fuel/feed/timber)

Some studies indicate equivalent or lower productivity (-20 percent) rates in CCF systems (reviewed by Lundmark *et al.*, 2016).

4.3 Mitigation of and adaptation to climate change

There is no strong evidence that soil C stock differs between CCF and clear-cut systems. Local situations should be examined carefully. In a broader perspective, mitigation benefits from the outflow of forest products that substitute the use of materials generating greater GHG emissions in addition to those related to the changes in C stocks, both in forest ecosystems and in wood products need to be considered. Lundmark *et al.* (2016) indicated that for Norway spruce in Sweden, biomass growth and yield is more important than the choice of silvicultural system per se for generating long term climate mitigation benefits associated with CO_2 emissions and C stock changes. In Canada, Paradis, Thiffault and Achim (2018) indicated that forest management systems that produce trees of greater size should increase the proportion of long-lived wood products, suggesting that the quality of the timber produced also has implication on GHG mitigation.

There is evidence that multi-species and multi-cohorts forest stands are more resilient to climate-change and to other threats, especially in the long-term (reviewed by Puettmann *et al.*, 2015). However, unexpected mortality of residual trees may occur with CCF, especially when foresters have little experience with such practice (Puettmann *et al.*, 2015).

Due to climate change and associated effects, it is envisaged to reduce these rotation durations to mitigate the risks associated with storms, fires or pathogen attacks (Roux *et al.*, 2017). This amounts to shifting from carbon sequestration in the ecosystem to carbon storage in products and increasing the share of substitution (Fortin *et al.*, 2012). From an ecosystem perspective, shortening rotations can have an impact on soil fertility and SOC with a general decreasing trend (Achat *et al.*, 2018). As the stand is renewed more often over the same period of time, biomass and nutrient exports are greater and the effects of soil preparation during forest soil regeneration on SOC are also amplified. Thus, the longer the duration of rotation, the more likely it is that SOC will increase, although the effect of very long rotations (*i.e.* several centuries) remains poorly known with variable results depending on the studies (Ji *et al.*, 2017; Leuschner *et al.*, 2014; Zhou *et al.*, 2006). Thus, simultaneously, the risk of climate change-related hazards increases (and may contribute to the reduction of the SOC stock in trees) with the length of the rotation , whereas the shorter the rotation, the greater the SOC losses related to forest management may be (Seely, Welham and Blanco, 2010). However, to date, there are no studies where forest stands are monitored longitudinally (not in chronosequence) over a longer timeframe and have experienced intensive silviculture (several short revolutions) on the one hand or extensive silviculture on the other (long rotation over the same time period). Only model-based studies can address the effect of the length of rotation on SOC (Johnson, Scatena and Pan, 2010; Wang *et al.*, 2013; Seely, Welham and Kimmins, 2002; Achat *et al.*, 2018). Numerical simulations are generally concordant and suggest a decrease in SOC stocks with a shortening of rotations (e.g. -15 to -20 percent after 360 years; Johnson *et al.*, 2010).

4.4 Socio-economic benefits

Continuous-cover forestry could generate more uniform cash flows (Puettmann *et al.* 2015); successful natural regeneration avoids the cost of plantation establishment.

Improvement of landscape visual quality and enhanced recreational opportunities (Puettmann *et al.* 2015).

5. Potential drawbacks to the practice

5.1 Decreases in production (e.g. food/fuel/feed/timber)

Some studies indicate equivalent or lower productivity (-20 percent) rates in CCF systems (reviewed in Lundmark *et al.,* 2016).

6. Recommendations before implementation of the practice

The greatest risk is the enhanced mortality of residual trees due to damages to roots and stems during operation or to greater exposure of residual trees to wind, drought or insects. It is advisable to start at small scale with a good knowledge of species autecology and with clear stand-density management goals.

7. Potential barriers to adoption

Table 4. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Implementation is not always easy depending on the current forest composition and structure, may need several steps.
Economic	Yes	Yields are less certain in regions with no tradition of CCF.

Photo of the practice

Photo 2. A mature stand of Douglas fir managed on CCF principles with a developing understorey of mixed conifers, including Douglas fir, western hemlock, western red cedar and grand fir | Coombs Wood, Cumbria, United Kingdom of Great Britain and Northern Ireland |

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3. Residue retention

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1. Description of the practice

Residue retention consists of leaving on-site all tree components with low merchntable value as well as litter following wood harvesting (i.e. removing only the stems of merchantable trees). Depending on the intensity of harvesting, forest harvest residues can be composed of leaves/needles, branches, twigs, low-quality or small-diameter stems, bark, dead wood, and roots. The amount of residue generated during harvest varies with forest type, productivity, climate, and rotation length. Retention of residues and litter provides carbon and nutrients to be recycled back into the soil. Residues are retained when stem-only harvesting is practiced, and the resulting residues are distributed back over the site. Residues are lost from the site when whole-tree harvesting is employed, when residues are collected for use in energy production, or when residues are piled and burned.

2. Range of applicability

Applicable globally.

3. Impact on soil organic carbon stocks

Removal of forest harvest residue removal has variable effects on SOC stocks (Table 5). Some meta-analyses (Clarke *et al.*, 2015; James and Harrison, 2016; Hume *et al.*, 2018) and reviews (Johnson and Curtis, 2001; Thiffault *et al.*, 2011; Clarke *et al.*, 2015) found no clear evidence of reduced SOC following the removal of forest residues, while others have reported significant reductions in SOC (Johnson and Curtis 2001; Achat *et al.* 2015a, 2015b). The benefits of residue retention for SOC are greatest in the surface organic layer (Clarke *et al.*, 2015; Wan *et al.* 2018), in soils that are coarse-textured (Oliveira *et al.*, 2018; Wan *et al.* 2018) and/or poor in organic matter (Thiffault *et al.,* 2011), in temperate rather than boreal forests (Achat *et al.*, 2015b), in conifer forests (Johnson and Curtis 2001), and at high harvest intensity (Achat *et al.*, 2015b). Very high SOC losses with whole-tree harvesting (up to 50 percent in the 0–5 cm soil layer) have been reported in short-rotation Eucalyptus forests in the Democratic Republic of the Congo and Brazil (Epron *et al.*, 2015; Rocha *et al.*, 2018). In contrast, on peaty soils in the UK, SOC stocks were higher under whole-tree harvesting compared to stem-only harvesting (Vanguelova *et al.*, 2010).

Few data are available on the long-term responses of harvest residue manipulation in tropical forests compared to temperate or boreal forests (Achat *et al.*, 2015b; Mayer *et al.* 2020).

Table 5. Changes in soil organic carbon stocks reported for harvest residue retention

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Years)	More information	Reference
Various	Temperate and subtropical	Various	NA	None	1 - 20+ after harvest	Review (Compilation of 53 studies)	Thiffault *et al.* (2011)
Finland	Continental	NA	NA	0.69	10	Forest floor + 0 - 10 cm mineral soil	Kaarakka *et al.* (2014)
Australia	Mediterranean	Podzol; Ferralsol	54. 7 - 72.7	0.62	5	0 - 20 cm	Mendham *et al.* (2003; 2002)
Canada	Continental	Inceptsols; Spodosols	46 - 65	-0.08 - 0.06	20	0 - 20 cm (14 sites)	Morris *et al.* (2019)
Finland	Continental			0.52	10 - 11	Forest floor + 0 - 10 cm mineral soil (6 sites)	Smolander *et al.* (2015)
Brazil	Tropical	Acrisols	18.2 - 23.1	0.04 - 1.82	3	0 - 10 cm	Oliveira *et al.* (2018)

4. Other benefits of the practice

4.1. Improvement of soil properties

By retaining nutrient-rich material such as leaves and bark on site, residue retention minimizes nutrient losses associated with harvesting (Achat *et al.*, 2015a; Paré and Thiffault, 2016). The increased organic matter with residue retention increases soil biological activity and foster earlier establishment of a stable microbial community structure (Achat *et al.*, 2015a; Baumann *et al.*, 2009; Smolander *et al.*, 2013). Retention of logging residues reduces topsoil compaction caused by heavy-machinery traffic during clear-cutting and logging operations (Achat *et al.*, 2015a), thereby mitigating possible breakdown of soil aggregates, and increases in soil bulk density and soil penetration resistance (Ampoorter *et al.*, 2007; Cambi *et al.*, 2015; Carter *et al.*, 2006).

4.2 Minimization of threats to soil functions

Table 6. Soil threats

Soil threats	
Soil erosion	Provide soil cover after harvesting, and protect soil from compaction, which could increase erosion susceptibility, particularly in steep terrains (Cambi *et al.*, 2015).
Nutrient imbalance and cycles	Minimize nutrient loss at harvesting and recycle nutrients back into the soil (Achat *et al.*, 2015a; Paré and Thiffault, 2016).
Soil acidification	Slightly increases soil pH and base cations, and reduces exchange acidity and exchangeable H and Al (Achat *et al.*, 2015a; Iwald *et al.*, 2013; Johnson *et al.*, 1991).
Soil biodiversity loss	Increases organic matter input to the soil, which increases soil biological activity; provides habitat for dead-wood-dependent organisms (Ranius *et al.*, 2018).
Soil compaction	Reduce topsoil compaction caused by heavy-machinery traffic during harvesting (Achat *et al.*, 2015a; Cambi *et al.*, 2015).
Soil water management	Residues act as surface mulch, which reduces evaporation and regulates soil temperature after harvesting (Thiffault *et al.*, 2011). Increases water infiltration by reducing soil compaction.

4.3 Increases in production (e.g. food/fuel/feed/timber)

The influence of retaining harvest residues on forest production is site-dependent. Positive impacts on tree growth of retaining harvest residues are observed particularly where the practice has promoted forest soil quality (Achat *et al.*, 2015a; Laclau *et al.*, 2010; Rocha *et al.*, 2018).

4.4 Mitigation of and adaptation to climate change

Improvement of soil quality and prevention of soil degradation through residue retention may reduce the need for fertilizer. Fossil fuels are used in the production, transport and application of fertilizer.

4.5 Socio-economic benefits

Residue retention reduces fertilizer requirements by reducing nutrient losses associated with forest harvest. By alleviating soil compaction, residue retention reduces the need for mitigative measures to improve soil health.

4.6 Additional benefits to the practice

Retaining logging residues may provide other ecological functions related to wildlife habitat, understory vegetation diversity, and water quality (Vance *et al.*, 2018).

5. Potential drawbacks to the practice

5.1 Increases in greenhouse gas emissions

Retention of harvest residues that could otherwise be used for bioenergy production and thereby reduce fossil fuel use could negatively affect GHG emissions. Soil organic C losses related to the removal of harvest residues has been argued to be negligible in comparison with the greenhouse mitigation benefit of avoided fossil-fuel emissions (Cowie *et al.*, 2006). In boreal forests, replacing fossil fuels by forest residues reduces the climate impacts associated with energy production by 7 percent to 62 percent (Repo *et al.*, 2012; Repo, Tuomi and Liski, 2011). Potential benefits of residue retention for GHG, such as increased C sequestration in tree biomass and SOC and potential reductions in fertilization must also be considered in calculations of net GHG emissions.

5.2 Conflict with other practice(s)

By creating obstacles, retention of harvest residues may make site preparation untenable or may restrict the machinery that can be used. Abundant residues can also make access difficult for mechanized or manual planters and reduce the availability of suitable planting spots. Retaining residues may increase fire risk in wildfire-prone forest ecosystems by increasing fuel load (Vance *et al.*, 2018).

5.3 Decreases in production (e.g. food/fuel/feed/timber)

In some cases, retention of harvest residues may impede forest regeneration, particularly in situations where residue promotes disease, insects, or wildfire susceptibility (Cleary *et al.*, 2013; Vance *et al.,* 2018). On cold sites, surface residues may reduce temperatures and growing season length, but may also reduce frost damage (Thiffault *et al.* 2011). On wet sites surface residues can hinder evaporation and make growing conditions less favorable. Residue retention can also slow down tree regeneration and early stand development by preventing soil preparation (Mayer *et al.*, 2020).

5.4 Other conflicts

On-site retention of woody residues may represent an opportunity cost in regions where there is an established market for using woody residues for bioenergy generation (He *et al.*, 2016; Repo *et al.*, 2015).

6. Recommendations before implementation of the practice

Decisions on whether to retain or remove residue are context-dependent. With regards to SOC, residue retention is most likely to have a beneficial effect in short-rotation intensive forestry plantations on soils that are coarse-textured and poor in SOC or nutrients. Residue retention is recommended in situations in which residue removal is uneconomical, operationally unfeasible, or environmentally unsustainable. Therefore, wood final use, market conditions and transport costs, machinery availability, requirement for other management practices such as site preparation, tree planting or fertilization), susceptibility of forest to wildfire, disease and pest risks, and interaction with should be considered when deciding about forest residue retention or removal. Policies, regulations, certification schemes, national guidelines and practical guides have been developed to ensure that forest biomass harvesting is sustainable (Stupak *et al.*, 2007). Regarding GHG emissions, the benefits of harvesting residues for bioenergy (i.e. replacing fossil fuels) must be weighed against the potential benefits of retaining residues (e.g. increased SOC, improved soil quality, reduced fertilizer requirement).

7. Potential barriers to adoption **Table 7.** Potential barriers to adoption

Barrier	
Biophysical	Land size (smallholder cannot invest in specialized machinery requirements); Topography can create restrictions for mechanized wood processing; Wildfire-prone forests because of accumulation of fuel load (Vance *et al.*, 2018).
Cultural	Total land clearing is adopted across multiple production sites (forestry and agricultural) and changes require long-term knowledge construction.
Social	Residues can be perceived as untidy and difficult access to forests; residues removal makes forest monuments more visible; On the other hand, stump harvesting can be perceived as negative due to great interference in the ground in a "uncaring" fashion (Ranius *et al.*, 2018).
Economic	Harvesting residues increase revenue in some locations; access to specialized machinery is required to process wood in the field and manage woody-debris during replanting operations.
Institutional	Lack of economic incentives and support from governments, including subsidies and payment for ecosystem services.
Legal (Right to soil)	Lack of policy regulation and standards in some regions.
Knowledge/capacity	Requires regional evaluations; practice complexity (multiple levels) and interactions with other management practices can discourage adoption.

Photo of the practice

Photo 3. Eucalypt plantation stem-only harvest. Wood has been pilled for site removal, and harvest residues (branches, treetops, and roots) are left on site. January 2014, Minas Gerais, southeast Brazil

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4. Inclusion of N fixing species

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1. Description of the practice

Some tree species have evolved an association with N-fixing bacteria, which infect the tree roots, stimulating the formation of nodules in which the bacteria proliferate, and consume energy compounds from the plant (Binkley and Fisher, 2019). The bacteria convert atmospheric N into ammonia and then amino acids, some of which is exported to the tree roots and incorporated into tree biomass. The fixed N is liberated to the soil via root or mycorrhizal exudates or as tree litter, which has higher concentrations of N than litter of non-N-fixing tree species. Tree genera with N-fixing root associates (hereafter referred to as "N-fixing tree species") include several genera of leguminous trees (such as *Acacia, Robinia, Falcataria, Albizia*), and "actinorhizal" species such as *Alnus, Eleagnus*, and *Casuarina* (Binkley and Fisher, 2019). Inputs of N from N-fixing tree species vary among species and sites but average about 75 kgN/ha/yr, which is more than an order of magnitude greater than typical annual rates of N input from atmospheric deposition (Binkley and Fisher, 2019).

Nitrogen-fixing tree species have multiple uses in forestry, both in pure stands and in mixtures with other species. In pure stands they are used to produce timber (Huong *et al.*, 2020), restore degraded soils (Mailly and Margolis 1992; Frouz *et al.*, 2009) and generate fuelwood (Mailly and Margolis, 1992). Nitrogen-fixing tree species are also planted to improve soil prior to planting commercial tree species (Voigtlaender *et al.*, 2012). Some N-fixing species are used for forest restoration (such as *Acacia koa*, Gugger *et al.*, 2018) or are used as nurse species to foster the regeneration of other native tree species (Chaer *et al.*, 2011). Nitrogen-fixing trees are also grown in mixtures with other commercial tree species such as Eucalyptus, pine and Douglas-fir (Forrester *et al.*, 2006). Potential ecological benefits of such mixtures include improved nutrient cycling, soil fertility, biomass production and carbon sequestration. Other benefits include reduced requirement for fertilizer, diversification of products, improved risk management and protection from pests and diseases (Forrester *et al.*, 2006).

2. Range of applicability

Nitrogen-fixing trees occur in tropical and temperate zones but are most abundant in subtropical and tropical latitudes (Mayer *et al.*, 2019). Species of *Acacia* are most often used for forestry in the tropics, and species from the *Leucaena*, *Casuarina*, *Albizia* or *Enterolobium* genera are also occasionally used. *Acacia mangium* is used in Eucalypt plantations in several tropical areas to increase the yield of Eucalypt without resorting to inorganic fertilizers. *Acacia* are also planted in monoculture, especially in South East Asia (Koutika and Richardson, 2019). Plantations of *Acacia mangium* and *Acacia auriculiformis* have been particularly important for small-holder forestry, providing important income for disadvantaged farmers (Huong *et al.*, 2020). *Acacia mangium* and *Erythrophleum fordii* and have been widely used for restoration of degraded soils in subtropical China (Luo *et al.*, 2016). In temperate latitudes, N-fixing tree species mostly belong to the *Robinia* or *Alnus* genera, and more rarely to the *Caragana* genus (Marron and Epron, 2019).

3. Impact on soil organic carbon stocks

Several studies in temperate and tropical regions have shown that soil C sequestration in forest soils is greater under N-fixing than under non-N-fixing species, given similar aboveground productivities (Forrester *et al.,* 2013; Binkley and Fisher, 2019). Comparisons of adjacent stands of N-fixing and non-N-fixing tree species have reported 20 percent–100 percent more soil C under the N-fixers; this would equate to 0.5–1.2 tC/ha/yr greater soil C accumulation in N-fixer forests than in comparable non-N-fixer forests (Resh, Binkley and Parotta, 2002). Analyzing 19 case studies with N-fixing species in temperate and tropical forests, Binkley (2005) reported a mean rate of C accretion in soils of 0.87 tC/ha/yr relative to non-N-fixing tree species. Overall, these soils accumulated about 12 to 15 g of C for every g of N accumulated. In a meta-analysis for north-temperate forests, Nave *et al.* (2009) reported a significant increase (+12 percent) in mineral SOC storage in response to N-fixing vegetation (Table 8).

Nitrogen-fixing trees increase SOC stocks much faster than other species on degraded soils such as post-mining sites (Frouz *et al.,* 2009), eroded soils (Zhang *et al.,* 2018), or after afforestation of savannas (Tang and Li, 2013). Inclusion of N-fixing *Acacia* trees into plantations of *Eucalyptus* can increase SOC stocks after a single rotation (Forrester *et al.,* 2013; Koutika *et al.,* 2014; Voigtlaender *et al.,* 2019). The greater rate of soil C increase under N-fixing trees results from a combination of greater stabilization of the organic matter added by the N-fixing trees, and reduced decomposition rates of older soil organic matter (Kaye *et al.,* 2002; Resh, Binkley and Parotta, 2002).

Table 8. Changes in soil organic carbon stocks reported for addition of N-fixing tree species

<table>
<tr><th>Location</th><th>Climate zone</th><th>Soil type</th><th>Baseline C stock (tC/ha)</th><th>Additional C storage (tC/ha/yr)</th><th>Duration (Year)</th><th>Depth</th><th>More information</th><th>Reference</th></tr>
<tr><td>Global</td><td>Tropical and temperate</td><td>Various</td><td rowspan="6">NA</td><td>0.87
0.25–1.80</td><td>-</td><td>-</td><td>19 case studies</td><td>Binkley (2005)</td></tr>
<tr><td rowspan="2">British Columbia (BC), Canada and Washington State, United States of America</td><td rowspan="5">Cold tempe-rate moist</td><td rowspan="2">Haplorthod, Xerochrept</td><td>1.13</td><td>23</td><td>O + top 50 cm</td><td>Alnus rubra mixed vs pure Pseudotsuga menziesii, infertile site</td><td rowspan="2">Binkley (1983)</td></tr>
<tr><td>0.53</td><td>23</td><td>O + top 50 cm</td><td>Alnus rubra mixed vs pure Pseudotsuga menziesii, fertile site</td></tr>
<tr><td>Southwest Washington State, United States of America</td><td>Haplumbrept</td><td>0.43–0.48</td><td>58</td><td>O + top 90 cm</td><td>Alnus rubra mix vs pure Pseudotsuga menziesii</td><td>Binkley et al. (1992)</td></tr>
<tr><td>Western Washington, United States of America</td><td>Boistfort series on Eocene basalt</td><td>0.75</td><td>40</td><td>O + top 20 cm</td><td>Alnus rubra vs adjacent Pseudotsuga menziesii</td><td>Bormann and DeBell (1981)</td></tr>
<tr><td>BC, Canada</td><td>Haplorthod = Humo Ferric Podzol</td><td>0.63</td><td>23</td><td>O + top 50 cm</td><td>Alnus sinuata mix vs pure Pseudotsuga menziesii</td><td>Binkley, Louisier and Cromack, (1984)</td></tr>
<tr><td>Pointe-Noire, Democratic Republic of the Congo</td><td rowspan="2">Tropical moist</td><td>Ferralic Arenosol</td><td>15.9</td><td>0.27</td><td>7</td><td>Top 25 cm</td><td rowspan="2">Acacia mixed vs pure Eucalyptus</td><td>Koutika et al. (2014)</td></tr>
<tr><td>Brazil</td><td>Ferrasols</td><td></td><td>-0.22</td><td>6</td><td>O + top 15 cm</td><td>Voigtlaender et al. (2012)</td></tr>
<tr><td></td><td></td><td>Andic Haplumbrepts</td><td>138</td><td>1.17</td><td>19</td><td>O + top 45 cm</td><td>Alnus rubra mixed with Pseudotsuga menziesii</td><td>Rothe et al. (2002)</td></tr>
</table>

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Year)	Depth	More information	Reference
Coastal Oregon, United States of America	Warm temperate moist	Dystrandept	35.6	1.19–1.81	12	Top 30 cm	*Ceanothus* vs *Pseudotsuga menziesii*	Binkley, Cromack and Fredriksen, (1982)
Victoria, Australia		Sodosol, Dermosal, Tenosol, Rudosol, Dermosol	31 – 43	1.00 to 1.44	9-17	Top 10 cm	*Acacia mearnsii*	Kasel *et al.* (2011)
Southeastern Australia		NA	NA	0.26	14	Top 20 cm	*Acacia dealbata* vs *Eucalyptus*	Hoogmoed *et al.* (2014)
				-0.07			*Acacia implexa* vs *Eucalyptus*	
Senegal	Tropical dry	Regosol		0.32	34	O + top 100 cm	*Casuarina*	Mailly and Margolis (1992)
Hawai'i	Tropical wet	Hydrudands		0.77	15	Top 40 cm	*Albizia vs Eucalyptus*	Resh, Binkley and Parotta, (2002)
				1.40		Top 50 cm		
		Hydrudands Andisol	128	1.1	16	Top 100 cm		Kaye *et al.* (2000)
		Hydrandepts	8	0.12–0.55	12	Top 20 cm		Garcia-Montiel and Binkley (1998)
Puerto Rico, United States of America	Tropical moist	Haplusterts	NA	1.25	16	Top 40 cm	*Casuarina vs Eucalyptus*	Resh *et al.* (2002)
		Troposamments		0.42	7			
				1.62	16		*Leucaena vs Eucalyptus*	
				0.71	7			
Victoria, Australia	Warm tempe-rate	Acrisols Dermosols	68	2.0	8	Top 30 cm	*Acacia mix with Eucalyptus*	Forrester *et al.* (2013)

4. Other benefits of the practice

4.1. Improvement of soil properties

When used for afforestation and reforestation of lands with degraded soils, N-fixing tree species have the potential to increase nutrient content and cycling (Koutika *et al.,* 2017; Voigtlaender *et al.,* 2019), reduce surface runoff and increase water infiltration (Paula *et al.*, 2019), and increase earthworm density (Zou, 1993) and aggregate formation (Garay *et al.,* 2004). In plantations in southeastern Australia, the macroaggregate SOC pool under *Acacia mearnsii* increased at least four-fold faster than that of *Eucalyptus* and other *Acacia* species (Kasel *et al.,* 2011). Colonization of *Eucalyptus* roots by AMF and acid and alkaline phosphatase activities in soil were significantly higher when grown in mixed systems with *Acacia mangium* (Bini *et al.,* 2018).

4.2 Minimization of threats to soil functions

Table 9. Soil threats

Soil threats	
Soil erosion	Afforestation and reforestation with N-fixing trees contributes to erosion control; (Vítková *et al.,* 2017); *Acacia* used to control gully erosion in Brazil (Chaer *et al.,* 2011). *Casuarina equisetifolia* used for coastal protection and to stabilize drifting sand dunes (Mailly and Margolis, 1992).
Nutrient imbalance and cycles	N-fixing tree species increase N content and availability especially in N-poor soils (Binkley and Fisher, 2019). *Alnus* also increases soil P cycling (Giardina *et al.,* 1995). Actinorrhizal N-fixing trees can accelerate rock weathering, and thereby enhance their access to multiple rock-derived nutrients (Perakis and Pett-Ridge, 2019).
Soil biodiversity loss	Higher bacteria, lower fungi, more earthworms under N-fixer (Zou, 1993). More litter-transforming macroarthropods, especially millipedes under *Acacia mangium* than Eucalyptus (Pellens and Garay, 1999; Zagato *et al.,* 2020). Higher soil macrofauna abundance and diversity in mixed plantations of Eucalypt and Acacia, than pure plantations of either species (Zagato *et al.,* 2020). Shifts in soil bacteria composition (Pereira *et al.,* 2017).
Soil water management	N-fixers used in afforestation and land restoration moderate water flows.

4.3 Increases in production (e.g. food/fuel/feed/timber)

Mixed forest plantations involving N-fixing species and other species such as eucalypt have been proposed to increase productivity in regions with N-deficient soils (Paula *et al.,* 2019). In a meta-analysis of 148 case studies from 34 plantations, mixed forest plantations involving at least equal proportions of N-fixing and non-N-fixing species are generally more productive than monospecific plantations (Marron and Epron, 2019). On average, mixed-tree plantations were 18 percent more productive than the non-N fixing monocultures. The size of the increase in the mixed stand was greater on lower productivity sites (Marron and Epron, 2019).

A rotation of N-fixing trees may improve site fertility prior to planting the more valuable commercial tree species. For example, twenty-year-old plots of pure Eucalyptus and pure *Falcataria* in Hawaii were harvested, and the plots replanted with the same or the opposite species. Plots with soils enriched in N by *Falcataria* in the first generation had twice as much available N and almost double the growth of *Eucalyptus* compared with soils that had Eucalyptus for both generations (Binkley and Fisher, 2019). Even intercropping *Acacia mangium* with *Eucalyptus* in Brazil led to much higher amounts of soil N in only one crop rotation (Voigtlaender *et al.,* 2019; Paula *et al.,* 2019).

4.4 Mitigation of and adaptation to climate change

Use of N-fixing tree species reduces the need for nitrogenous fertilizers and so eliminates GHG emissions associated with the production, distribution and application of N fertilizers (Paula *et al.,* 2019).

Mixed plantations with non-N-fixing trees and N-fixing species can sequester more C than pure plantations of non-N-fixing trees, such as Eucalyptus (Forrester *et al.*, 2006, 2013).

4.5 Socio-economic benefits

Mixed forest plantations provide a greater diversity of products than monospecific forest plantations (Paula *et al.,* 2019). Pairs of mixtures of fast-growing exotic species such as *Eucalyptus* sp. or *Pinus* sp. with N-fixing trees such as *Acacia mangium* or *A. mearnsii* have the potential to offer a wide range of products in the same area, including timber, firewood, charcoal, tannins, resins, and essential oils (Paula *et al.,* 2019). Intercropped plantations with fast-growing N-fixing trees and *Eucalyptus* assists the restoration of degraded areas and increases the economy of small- and medium-sized farmers in the *Cerrado* and Amazon regions of Brazil (Paula *et al.,* 2019). N-fixing trees such as *Erythrina* spp. are used to produce shade-grown cacao in Brazil (Gama-Rodrigues, 2020).

The tradition of using *Robinia* for afforestation In Central Europe has led to it being an important part of the economy in some countries. In addition to its valuable resistant wood, *Robinia* is used for honey-making and more recently for production of dead wood (Vítková *et al.,* 2017).

4.6 Additional benefits to the practice

Mixed-species plantations (including those containing N-fixing tree species) may be better protected from pests and diseases than are monocultures (Marron and Epron, 2019). Mixing tree species reduces the impact of specialist insect herbivores on individual susceptible tree species and the impact of specialist pathogens on host tree species (Bauhus *et al.,* 2017).

Afforestation and reforestation with N-fixing species can facilitate the recovery of natural forests on deforested and degraded soils, by creating conditions conducive to the establishment and growth of more nutrient-demanding native tree species (Yang *et al.,* 2009; Root-Bernstein *et al.,* 2017). For example, Dipterocarps have been grown in gaps of *Acacia* trees in the tropics (Norisada and Kojima, 2005). In the State of Amazônia, northern Brazil, biomass and richness of native plant species regenerating in the understory of planted N-fixing tree species was higher than in areas planted with non-N-fixing species (Chaer *et al.,* 2011).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Changes due to N-fixing species can be similar to those of atmospheric N deposition or N fertilization – soil acidification, elevated nitrate, and increased N losses through nitrate leaching and N_2O emission.

Addition of N through N_2 fixation can create or exacerbate P deficiencies, especially on soils with low organic matter and P reserves or with high P adsorption capacity (Sitters, Edwards and Olde Venterink, 2013; de Carvalho Balieiro *et al.,* 2020).

Table 10. Soil threats

Soil threats	
Nutrient imbalance and cycles	N fixation increases demand for the soil nutrients P and Mo, so fertilization may be necessary to realize growth potential (Augusto *et al.,* 2013; Dynarski, Pett-Ridge and Perakis, 2020; Mailly and Margolis, 1992) P supply depleted within 1 rotation of *Albizia* in Hawai'i (Binkley *et al.,* 2000). Elevated soil nitrate persists for a decade after removal of N-fixer (Nsikani *et al.,* 2017).
Soil acidification	Greater soil N associated with declines in soil pH and exchangeable Ca, Mg, and K (Yamashita, Ohta and Hardiono, 2008; Binkley and Fisher, 2019).
Soil water management	Greater soil N from *Alnus* associated with increased nitrate leaching and aluminum mobilization (Perakis *et al.,* 2013) and associated pollution of waterbodies.

5.2 Increases in greenhouse gas emissions

In temperate regions, N-fixation by *Alnus* species increased N_2O emissions compared to non-N-fixing species across 12 sites in an elevational gradient in Switzerland (Buhlmann *et al.,* 2017). Several studies have found greater concentrations of N, mainly NO_3, in soil solution under alder (Binkley *et al.,* 1992; Buhlmann, Körner and Hiltbrunner, 2016), and increased denitrification under alder versus non-N-fixing vegetation (Mogge, Kaiser, and Munch, 1998). Varying rates of N_2O emissions have been found in *Alnus glutinosa* forests in Switzerland (0.2-1.7 kg N_2O-N/ha/season; Buhlmann *et al.,* 2017), southern Germany (0.5-1.0 kg N_2O-N/ha/yr; Eickenscheidt *et al.* 2014; and 3.1 Kg N_2O-N ha/yr (Warlo *et al.,* 2019) and northern Germany (0.8-7.3 kg N_2O-N/ha/yr; Mogge, Kaiser, and Munch, 1998). Higher rates of N_2O emissions are associated with: Buhlmann *et al.* (2017) estimated stands of *Alnus viridis* in Switzerland would release 130 t N_2O-N per growing season (Mid-June–mid-Oct), which represents 1.5 percent total Swiss emissions of this GHG (FOEN, 2015).

In the tropics, leguminous plantations raise N availability in the soil, making these plantations a possible N_2O source (IPCC, 2003). *Acacia* are important leguminous trees for industrial plantations because of rapid growth and tolerance of acidic, nutrient-poor soils and more than 8 million ha of Acacia were planted in the tropics and sub-tropics by 2000, 96 percent in Asia (FAO, 2001). Stimulation of N_2O emissions has been documented in leguminous plantations in the Asian tropics (Arai *et al.* 2008), where these trees are not a dominant component of native forests (Primack and Corlett, 2005). Annual N_2O fluxes in *Acacia* plantations or tropical secondary forests containing native N-fixing species are eight times higher (2.56 Kg N_2O/ha/yr) than those from secondary forests without N-fixing species in Indonesia (Arai *et al.,* 2008; Konda *et al.,* 2008; Erickson *et al.,* 2001). In southern China, N_2O emissions from *Acacia auriculiformis* plantations (2.3 kg N_2O-N/ha/yr) were significantly greater than from *Eucalyptus urophylla* (1.9 kg N_2O-N/ha/yr) and was further stimulated by N fertilization to 3.1 Kg N_2O-N/ha/yr (Zhang *et al.,* 2014). This demonstrates that the elevated N_2O emissions result from the input of fixed N, and further suggests that elevated atmospheric N deposition will increase N_2O emissions from leguminous tree plantations in the tropics. Addition of P, as well as N, to *Acacia* stands mitigated the increased N_2O fluxes (reduced from 3.1 to 2.7 Kg N_2O-N) due to increased N uptake by vegetation in the presence of increased P availability (Zhang *et al.,* 2014). There are pronounced seasonal fluctuations in N_2O emissions in tropical forests, with higher emissions in the wet season than the dry season. Denitrification is accelerated in the wet season due to greater prevalence of anaerobic soil conditions (Werner *et al.,* 2007, Arai *et al.,* 2008, Konda *et al.,* 2010).

5.3 Conflict with other practice(s)

Relative to monocultures of commercial tree species, mixtures of commercial species with N-fixing tree species can be more challenging to manage.

5.4 Decreases in production (e.g. food/fuel/feed/timber)

When grown in mixtures, there is potential for N-fixing tree species to reduce productivity of the commercial tree species through competition for resources other than N (Lavery *et al.,* 2004). The success of mixed-tree plantations (i.e. when the mixture is more productive than the monoculture) is variable and difficult to predict, but most likely to succeed where N is limiting productivity (Marron and Epron, 2019).

5.5 Other conflicts

The introduction of N-fixing species outside of their natural ranges carries the risks associated with introduction of any non-native species, more so because the competitive advantage of N-fixing species can make them invasive, such as *Robinia pseudoacacia* in central Europe (Vítková *et al.,* 2017), *Acacia mangium* and *Falcataria moluccana* in tropical and subtropical regions (Hughes *et al.,* 2013; Koutika and Richardson, 2019). Although negative impacts of invasive N-fixing species have been reported in tropical and subtropical rangelands around the world, exotic N-fixing tree species (including *Robinia* and *Acacia* spp.) are used in many areas to meet the needs of people without major risk of invasion (Richardson *et al.,* 2011). A stratified approach, combining tolerance in some areas with strict eradication at some sites, has been recommended to allow for the sustainable use of N-fixing tree species (Vítková *et al.,* 2017). Invasion risk assessments are recommended prior to new plantings of N-fixing species (Koutika and Richardson, 2019).

6. Recommendations before implementation of the practice

- Assess likelihood of species becoming invasive;
- Investigate potential end uses and markets for candidate tree species;
- Survey local peoples' perception of species;
- Determine the N-fixing tree species and genotype best suited for the site and, if grown in mixture, for the companion tree species;
- Affirm that the productivity of the site is primarily limited by N;
- Assess likelihood of deficiencies of nutrients other than N;
- Mix or rotate species to avoid consequences of excess N relative to other nutrients

7. Potential barriers to adoption

Table 11. Potential barriers to adoption

Barrier	YES/NO	

Biophysical	Yes	Site productivity must be N-limited; site conditions must be suitable for N-fixing species.
Cultural	Yes	N-fixing tree species may not be traditionally used.
Social	Yes	Risk of species becoming invasive must be weighed; local peoples' perception of species desirability.
Economic	Yes	Potential declines in overall productivity, as well as additional management costs. Development of market for products of N-fixing tree species needed.
Knowledge	Yes	Impacts on productivity of commercial tree species needs to be better understood, as well as implications on drought, disease and pest damage.

Photo of the practice

Photo 4. Acacia mangium planted near Bintulu, in Sarawak, Borneo (Malaysia)

Table 12. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Soil fertility improvement of nutrient-poor and sandy soils in the Congolese coastal plains	Africa	7	6	1

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5. Forest fertilization

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1. Description of the practice

Forest fertilization refers to the periodic addition of nutrients considered to limit productivity, and is employed in order to increase the production of stemwood biomass of forests. Globally, the most common nutrients added to forest are nitrogen (N) and phosphorus (P); others such as potassium (K), sulphur (S), boron (Bo), calcium (Ca) or magnesium (Mg) may be added on certain sites or when secondary deficiencies of these nutrients may develop following addition of N and P (Binkley and Fisher, 2019). Single-nutrient deficiencies occur in one-third of cases, in the remainder two or more nutrients are co-limiting or arise as secondary deficiencies (Harpole *et al.*, 2011). Nitrogen is usually applied as urea or ammonium nitrate granules, and P as triple superphosphate or rock phosphate. Organic materials such as compost or municipal biosolids or industrial residuals such as wood waste or ash are occasionally applied as fertilizers. Under intensive forest management, a suite of elements demonstrated to yield the most profitable growth responses are applied every year or few years during a rotation. Under extensive forest management regimes, single-element fertilizers (usually N) are operationally applied once or twice during a rotation (Binkley and Fisher, 2019). Fertilization of nutrient-deficient trees leads to increases in photosynthesis rates, leaf biomass, stemwood biomass and coarse-root biomass, but not fine-root biomass, which may decline. The decline in fine-root biomass is associated with declines in root exudation and allocation to root symbionts (Maier *et al.*, 2004). This has implications for SOC, as in some forests root- and mycorrhiza-derived C constitutes the majority of long-lived soil C (Clemmensen *et al.*, 2013; Sokol *et al.*, 2019). Fertilization also accelerates stand growth and self-thinning and shortens economic rotation times (Binkley and Fisher, 2019).

2. Range of applicability

Globally, the productivity of most forests is limited by N; native forests in both temperate and tropical regions increase growth by about 20 percent when N fertilizer is added (Binkley and Fisher, 2019; LeBauer and

Treseder, 2008). Phosphorus deficiency is common in some temperate regions, especially on highly weathered soils, and is widespread in fast-growing plantations in the tropics (Binkley and Fisher, 2019). In weathered tropical soils, fertilization with N, P and K is necessary to increase primary productivity and wood production (Gonçalves *et al.,* 2008; Laclau *et al.*, 2009, Wright *et al.*, 2018). In areas typically N-limited that receive substantial atmospheric N deposition, primary deficiencies of P may become the norm (Hedwall, Bergh and Brunet, 2017).

In boreal conifer forests, N is applied to forests on mineral soils whereas P and K are used in peatland forests. Boron may be added to forests that had been under slash and burn cultivation in the past. A typical nitrogen (N) dose is 150 kg ha^{-1} and the growth response to this dose is 20–25 m^3/ha. Most of the response occurs within 5 years and the N-fertilization effect is over in 10–12 years (Pukkala, 2017).

Addition of N to Douglas-fir forests of western North America increases growth rates by 2–4 m^3/ha/yr for 8–15 years. In intensively managed pine plantations in the south-eastern U.S. average response to one-time fertilization with 225 kg N/ha plus 30 kg P/ha is 3.5 m^3/ha/yr over 8 years (Binkley and Fisher, 2019). Most conifer plantations in New Zealand and Australia are fertilized at least once in a rotation. Australian plantations of eucalypts and pine, are typically fertilized with N (50–70 kg/ha), P (25–40 kg/ha), and sometimes, K, S, copper and zinc, at planting, early-rotation, and mid-rotation (Binkley and Fisher, 2019). Average growth response is 3–10 m^3/ha/yr. Growth responses after fertilization with N, P, and K in South Africa are commonly 6–8 m^3/ha/yr (Binkley and Fisher, 2019).

Almost all tropical forest plantations are fertilized with P, and most are also fertilized with other elements including N, Ca, Mg, and K. Typical growth responses to fertilizer additions in Brazilian eucalypt plantations are 4–8 m^3/ha/yr for five years or longer (Gonçalves *et al.*, 1997).

3. Impact on soil organic carbon stocks

Increased stocks of SOC are often encountered in forests amended with nitrogenous fertilizers or simulated N deposition. In a meta-analysis of the impacts of elevated N inputs (including N fertilisation) on the storage of C in forest soils based on 72 experimental sites, Nave *et al.* (2009) found that N inputs increased total SOC stocks (combined forest floor and mineral soil) by 7.7 percent. Stocks of organic C increased predominantly in the mineral soil (by 12.2 percent). In the review by Johnson and Curtis (2001), N fertilization was the only forest management practice that had a clear positive effect on the SOC pool. Greater accumulations of humus are often noted following N fertilization of boreal forests (Nohrstedt, 1990; Mälkönen and Kukkola, 1991; Mäkipää, 1995; Olsson *et al.*, 2005) and simulated N deposition in temperate forests (Lovett *et al.*, 2013), as have greater SOC concentrations in mineral soil (Pregitzer *et al.*, 2008; Cusack *et al.*, 2011; Huang *et al.*, 2011). Added N may also increase accumulation of C in occluded particulate organic matter (Zak *et al.*, 2017). Increased mineral-associated C stocks in response to N fertilization have also been reported in temperate and tropical forests, even in the absence of a detectable increase in bulk SOC (Hagedorn, Spinnier and Siegwolf, 2003; Cusack *et al.*, 2011).

Table 13. Changes in soil organic carbon stocks reported for nutrient additions

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Years)	Depth (cm)	Fertilization rates	Reference
Sweden and Finland	Boreal	NA	NA	0.1 – 1.2	14–30	O horizon + 0 -10	Meta-analysis; Repeated additions of N or NPK; Cumulative amount 600-2340 kgN/ha	Hyvönen *et al.* (2008)
Sweden		NA	32.3	0.12 – 0.25	18–21	O horizon	-	Nohrstedt *et al.* (1989)
Massachusetts, United States of America	Cool temperate	Typic Dystrudepts	1.3	0.28	20 N dep	0–40	50 and 150 kgN/ha/yr	Frey *et al.* (2014)
New York State, United States of America		Lithic Dystrochrepts	NA	0.3–1.3	6	O horizon	50 kgN/ha/yr	Lovett *et al.* (2013)
Washington, United States of America		Vitrixerands, Hapludands, Dystroxerept	175	3.7	21–29	0–100	672–896 kgN/ha in 3 or 4 additions	Adams *et al.* (2005)
Pennsylvania, United States of America		Alfisols	0.5	0.42	22 N dep	14–21	100 kgN/ha/yr	Wang *et al.* (2019)
Michigan, United States of America		Typic haplorthod	1.8	0.7	10 N dep	0–70	30 kgN/ha/yr	Pregitzer *et al.* (2007)
China	Subtropical moist	Oxisols	NA	1.5 (50 kgN/yr) 1.3 (100kgN/yr) 0.9 (150kgN/yr)	14 N dep	0–50	50-150 kgN/ha/yr	Yu *et al.* (2020)
Puerto Rico	Tropical wet	Ultisols and Oxisols	0.9	0.005	7	0–25	300 kgN/ha/yr + PK	Li, Xu and Zou, (2006)
			79	5.7	3 N dep	0–40	50 kgN/ha/yr	Cusack *et al.* (2011)

4. Other benefits of the practice

4.1 Minimization of threats to soil functions

Table 14. Soil threats

Soil threats	
Nutrient imbalance and cycles	Addition of specific nutrients can balance overall nutrient supply.
Soil acidification	Urea can be applied to increase pH; ammonium nitrate to reduce pH.

4.2 Increases in production (e.g. food/fuel/feed/timber)

Intensive forest management, including fertilization, increases the rate of stemwood production and reduces the economic rotation time for the stand. In the southeast U.S., the average productivity of commercial *Pinus taeda* plantations is more than 4-fold higher than of natural stands, and 17 percent of this gain has been attributed to fertilizer amendments (Noormets *et al.*, 2015). The productivity of eucalypts in Brazil has nearly doubled in the last 2 decades, owing to intensive management techniques, including fertilization (Goncalves *et al.*, 2013). Average gains from fertilization in other types of plantations are given in Section 2. In addition to the value of the timber to the landholder, the potential to produce timber faster means that less land is needed to supply the same amount of timber. This could increase the amount of land available for other purposes such as agriculture, recreation or conservation.

Forest fertilization may alter the understory vegetation towards a flora indicative of more fertile sites – increases of grasses and herbs, declines in some bryophytes, lichens and dwarf shrubs. As such fertilization can improve forage production and quality for livestock in forest grazing or silvopastoral systems. Forest fertilization can also increase browse quality, which increases populations of herbivore and their predators, and can also lead to tree damage (Sullivan and Sullivan, 2018).

4.3 Mitigation of and adaptation to climate change

The production, transport and application of chemical fertilizers all emit greenhouse gases. However, the additional biomass produced following fertilization sequesters CO_2 equivalents greater than those emitted by at least an order of magnitude. Fertilization of mid-rotation pine stands in the southeastern U.S. with N and P sequestered 6.2 million tons of CO_2 in additional stem growth whereas the CO_2 equivalents corresponding to fertilizer production, transport, and application were 231,000 tons (Albaugh *et al.*, 2019). In a short-rotation Eucalyptus plantation in Brazil, for every 1 kg of CO_2-eq. emitted, 43.15 kg of CO_2-eq. were sequestered in the (utilized) biomass (Quartucci, Schweier and Jaeger,2015). The net positive effects of fertilization on overall C balance may be even larger if one factors in 1) the additional carbon sequestered in coarse root, branch, and foliage growth, 2) changes in soil CO_2 evolution and ecosystem carbon storage, 3) reductions in rotation length that increase carbon sequestration in the system, 4) increased tree size, which increases the proportion of dimension lumber that retain the sequestered carbon in the system for a longer time than smaller size trees, which may be used to make pulp and paper (Albaugh *et al.*, 2012), and 5) if the additional growth is used for bioenergy so substitutes for coal, oil and natural gas (Bergh and Hedwall 2013). For example, fertilizing 10 percent of Swedish forest land could reduce annual GHG emission by 11.9 million or 18.1 million t CO_2eq if the reference fossil fuel is fossil gas or coal, respectively (Sathre, Gustavsson and Bergh, 2010).

As forest fertilization can be applied within present management systems and will give immediate effects, it is probably the most efficient tool of forest management to affect the carbon cycle, at least in a short term. The increased amount of biomass can be used as a growing carbon stock or to replace fossil fuels (Hedwall *et al.*, 2014).

4.4 Socio-economic benefits

The economic benefit of forest fertilization to the landowner depends on the magnitude of response, its predictability, the time required before the benefit is realized (rotation length), and the risk of forest loss or damage during the rotation. Forest fertilization can offer socio-economic benefits on a national level – for example an assessment of the potential benefits of forest fertilization in Latvia, applying 436 kgN/ha to 2000 ha of forests annually would generate an additional 142 000 m^3 of wood. After 10 years this would generate 3.2 million USD per year and contribute to CO_2-eq sequestration of 12.7 million (Lazdiņš *et al.*, 2018).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Table 15. Soil threats

Soil threats	
Nutrient imbalance and cycles	Growth response from addition of one nutrient can create deficiency of another (Binkley and Fisher 2019). Chronic addition of N could reduce or alter the cycling of other nutrients (Wang *et al.*, 2019).
Soil acidification	nitrification induced by N fertilization may acidify soil and soil solution Alters pH depending on formulation and dosage (Binkley and Fisher, 2019)
Soil biodiversity loss	Fertilization usually reduces belowground C fluxes and microbial biomass and alters the saprotrophic community (Noormets *et al.*, 2015).

Soil threats	
	N and P fertilization reduce proportional allocation to roots and root symbionts (Maier *et al.*, 2004), and often reduces soil microbial biomass and activity (Högberg *et al.*, 2003) and changes soil microbial community (Morrison *et al.*, 2016).
Soil water management	Plant growth response to fertilization can increase water demand (Ward *et al.*, 2015). Increase in leaf biomass from fertilization can increase water usage and loss via transpiration (Ward *et al.*, 2015). Addition of more N or P than soils can retain risks eutrophicaton of waterbodies (Laclau *et al.*, 2010).

5.2 Increases in greenhouse gas emissions

Fertilization may have detrimental effects on GHG emissions (Liu and Greaver, 2009). Forest soils are sinks for 5 percent of global annual methane emissions, through methane oxidation (Schlesinger, 1997) and potential sources, or sinks for nitrous oxide through microbial nitrification and denitrification (Robertson and Groffman, 2007). Methane has 25 times, and N_2O has 298 times the global warming potential of CO_2 (Forster *et al.*, 2007). Although N addition increases the terrestrial C sink in forests, the CO_2 reduction may be offset by N stimulation of N_2O emissions and inhibition of CH_4 oxidation.

Fertilization effects on GHG fluxes are complex and vary with timing and chemistry of fertilizers, environmental variables and ecosystem properties (Figure 2). N fertilization can potentially increase N_2O emissions through microbial nitrification and denitrification (Levy-Booth, Prescott and Grayston, 2014). N fertilization can either stimulate CH_4 oxidation by supplying a limiting nutrient (N) to CH_4-oxidizing bacteria (Bodelier and Laanbroek, 2004) or by supplying NH_4 to ammonia-oxidizing bacteria that can also oxidize CH_4 (Jiang and Bakken, 1999). N fertilization can also inhibit CH_4 oxidation due to NH_4 competing with CH_4 for active sites on the methane monoxygenase enzyme (Dunfield and Knowles, 1995), or through suppression of methane monoxygenase enzyme due to alternative labile C sources being available to CH_4 oxidizers in N-amended soils (Fender *et al.*, 2012).

In a meta-analysis of the effects of N addition on the flux of three major GHGs: CO_2, CH_4 and N_2O (Liu and Greaver, 2009), N addition increased ecosystem carbon content of forests by 6 percent. Across all ecosystems, N addition increased CH_4 emission by 97 percent, reduced CH_4 uptake by 38 percent and increased N_2O emission by 216 percent. Nitrogen addition ranging from 25 to 200 kg N/ha/yr applied to forest ecosystems for 6–15 years increased ecosystem carbon by an average of 6 percent. Coniferous and deciduous forests with chronic anthropogenic N inputs show 30-50 percent reduced CH_4 uptake and 2-3-fold increases in N_2O emissions. Nitrogen enrichment of tropical forests, in particular, increased N_2O emissions by 739 percent. Contrastingly, in unpolluted temperate coniferous forests, N fertilization had no effects on GHG fluxes (Basiliko *et al.*, 2009).

In addition, volatilization of NH_3 from surface-applied urea may occur when soils are moist and relative humidity and air temperatures are high (Elliot and Fox, 2014). Enhanced efficiency fertilizers have been developed to minimize losses through NH_3 volatilization (Raymond *et al.*, 2016).

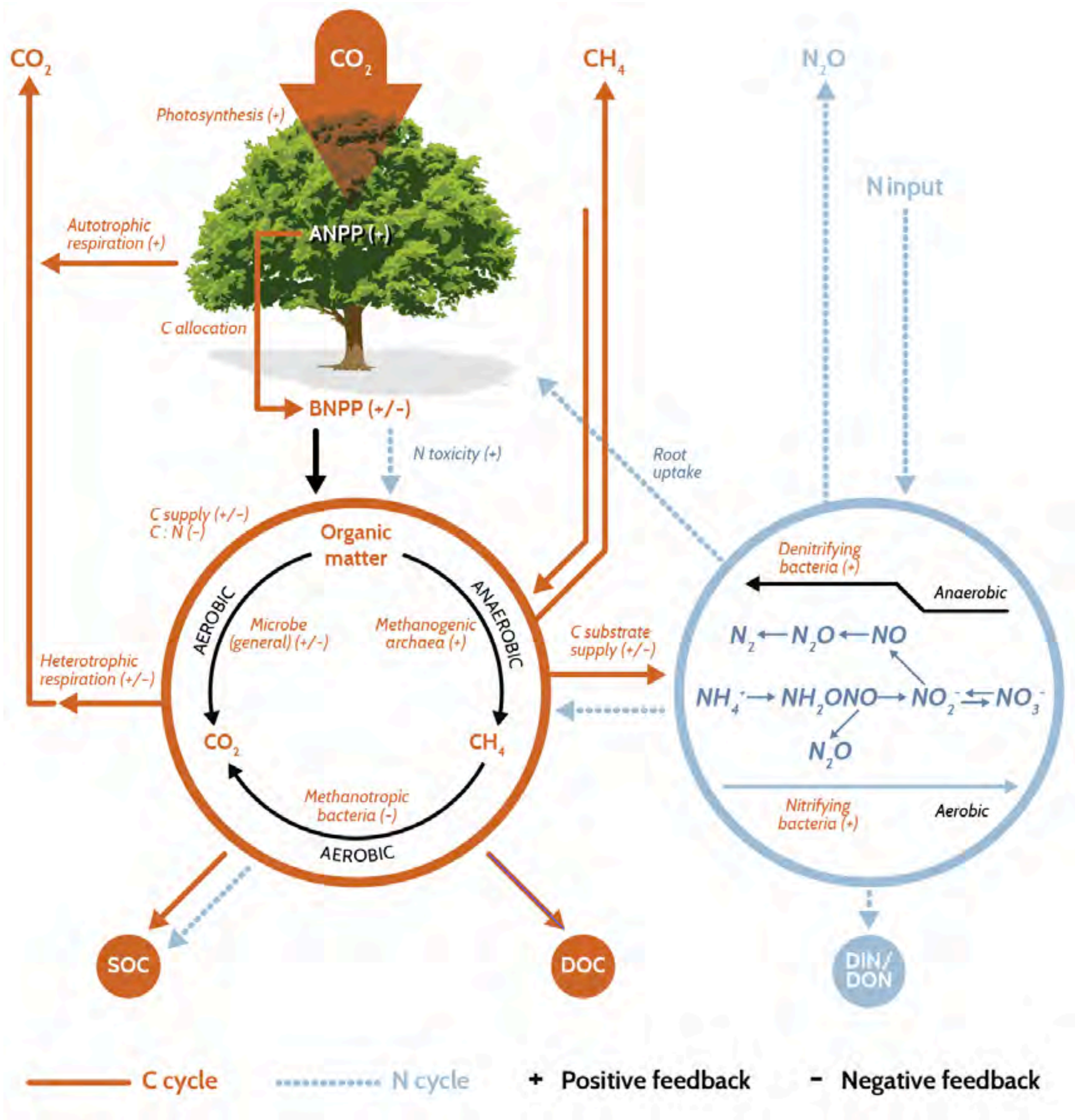

Figure 2. The potential mechanisms that regulate the responses of CO_2, CH_4 and N_2O production and consumption to elevated N (ANPP, aboveground net primary productivity; BNPP, belowground net primary productivity; SOC, soil organic carbon; DOC, dissolved organic carbon; DIN, dissolved inorganic nitrogen; DON, dissolved organic nitrogen). From Liu and Greaver (2009)

5.3 Conflict with other practice(s)

Fertilizers are expensive so the growth response and consequent economic benefits of fertilization versus other stand-tending practices must be predictable. Fertilization may stimulate growth of non-crop vegetation and necessitate vegetation control to realize the full benefit of fertilization (Binkley and Fisher, 2019).

5.4 Decreases in production (e.g. food/fuel/feed/timber)

Fertilization can intensify drought stress (Ward *et al.*, 2015) as a consequence of greater water demand and lower proportion of fine roots (Noormets *et al.*, 2015), especially in heavily fertilized plantations (Battie-Laclau *et al.*, 2014).

5.5 Other conflicts

The potential benefits of N fertilization for increasing tree growth and SOC stocks must be weighed against the associated environmental costs, as the production, transport and application of synthetic fertilizers all entail fossil fuel combustion and emission of CO_2.

There may be public concerns over potential eutrophication of drainage waters, although N fertilizer in forestry generally leads to small and transient increases in N concentrations in stream water (Binkley, Burnham and Allen, 1999; Smethurst, 2010). Leaching losses can be reduced by applying fertilizer in small doses rather than a single large application (Bergh *et al.*, 2008; Albaugh *et al.*, 2019).

6. Recommendations before implementation of the practice

Addition of N is not advisable in ecosystems where N is plentiful due to atmospheric N deposition or previous agricultural use, as it could stimulate losses as nitrate in water or as greenhouse gases through denitrification (Gao *et al.*, 2015).

Site water balance must be considered to prevent drought stress caused by increased leaf biomass and productivity without concomitant increases in root biomass.

In degraded soils with low organic matter and clay contents, N would be better added in organic forms such as composts or municipal biosolids that would increase retention of the N in the soil (Larney and Angers, 2012).

Investment in research is necessary to predict growth response to addition of nutrients. Across a landscape this involves determining relationships between growth response and site factors, soil or foliar nutrient concentrations, and remotely sensed factors such as leaf area index (LAI) and normalized difference vegetation index (NDVI) (Blinn *et al.*, 2019).

Possible effects of N addition on the fluxes of CH_4 and N_2O must be given careful consideration.

Given that equivalent SOC gains are achievable through incorporation of N-fixing tree species (Nave *et al.* 2010), this may be preferable to chemical fertilizers in many situations.

7. Potential barriers to adoption

Table 16. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	In degraded or coarse-textured soils there may be insufficient retention of added fertilizer, and organic fertilizers would be more effective.
Cultural	Yes	N is popularly viewed negatively as a pollutant due to association with atmospheric N deposition. Both N and P are associated with degradation of water quality resulting from excessive applications in agriculture.
Social	Yes	May be public concerns about effects on water quality and biodiversity, also association of chemical fertilizers with fossil-fuel use.
Economic	Yes	Cost of fertilizer mandates good ability to predict growth response, which requires research into relationships with site factors and leaf area (Albaugh *et al.*, 2019).
Knowledge	Yes	Belowground effects of nutrient addition need to be better understood before effects on SOC can be predicted (Noormets *et al.*, 2015)

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6. Forest afforestation, reforestation and natural regeneration

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1. Description of the practice

Afforestation is the conversion from other land uses into forest, or the increase of the canopy cover to above 10 percent (FAO, 2000). Afforestation is the reverse of deforestation and includes areas that are actively converted from other land uses. Afforestation includes conversion to forest through silvicultural measures or natural transitions into forest, for example on abandoned agricultural land or in burnt-over areas that have not been classified as forest during the barren period. The conversion should be long-term, *i.e.* the transition into forest is expected to last more than ten years. Local climatological conditions, land use contexts or the purpose of the analysis may however justify use of a longer time frame. If the area had been temporarily deforested, re-establishment of forest is termed "reforestation" (FAO, 2000).

Afforestation may occur passively (i.e. with no human intervention) following land abandonment, or actively through planting of tree seedlings or sowing seeds. It may also occur via assisted natural regeneration (ANR), in which specific actions are taken to reduce barriers to natural forest regeneration, such as control of grazing or weed competition, amelioration of soil or microclimate conditions or seed dispersal.

2. Range of applicability

Afforestation (and reforestation) is applicable in locations where the climate dictates that the natural vegetation would be forest (see Hotspot: Forests). Afforestation is not recommended in areas that would naturally be grassland (Silveira *et al.*, 2020) or other non-forest biomes.

3. Impact on soil organic carbon stocks

Afforestation increases SOC content especially of soils that have been degraded through mineral extraction or unsustainable cropping (Table 17). The rate of SOC accumulation is greatest in soils that have been depleted in C (Shi *et al.,* 2013; Wang and Huang 2020). Afforestation on lands made barren by mining increased SOC stocks by 15-fold (average 173 percent; meta-analysis by Nave *et al.* 2013). SOC stocks significantly increased within 15-25 years of afforestation and continued through subsequent decades. SOC accumulation on afforested mine sites averaged 2.46 tC/ha/yr in the first 10 years and 0.87 tC/ha/yr over the first 40 years (Frouz *et al.,* 2014). Afforestation on former cropland (that is, land used for cultivation of crops; FAO 2017) may result in a significant increase in SOC stocks (Mayer *et al.* 2020). In some studies, no new steady-state levels were reached within 100 years (Poeplau *et al.*, 2011; Bárcena *et al.*, 2014) while in others, modest decadal increases culminated in a ~15 percent net increase in SOC stock by the end of the first century (Nave *et al.*, 2013). Afforestation of degraded croplands under China's Grain for Green program averaged 0.26 tC/ha/yr (Deng *et al.,* 2014). In Ethiopia, afforestation of degraded croplands with *Eucalyptus* elevated SOC stocks to nearly 70 percent of levels found in a natural dry subtropical montane forest within 30 years (Assefa *et al.,* 2017).

In contrast to cropped lands, afforestation of grasslands and pastures has little effect on SOC stocks (Guo and Gifford 2002; Mayer *et al.* 2020), and often results in SOC losses (Poeplau *et al.* 2011). SOC also accumulates faster on highly degraded soils restored to grassland compared to those afforested (Wei *et al.* 2012; Frouz *et al.*, 2014). The high stocks of SOC in grasslands and pastures have been attributed to much greater fine root length and water content than in forest soils (Berhongaray and Alvarez, 2019; Deng *et al.* 2014).

Immediately following afforestation, SOC stocks may decrease due to site disturbance, soil erosion, and low NPP and C inputs from the young vegetation (Paul *et al.,* 2002). The decline usually lasts about 10 years when sites are actively managed (Laganière, Angers and Paré, 2010; Deng *et al.,* 2014; Li *et al.,* 2012; Nave *et al.,* 2013) but can last up to 35 years following agricultural abandonment (Paul *et al.,* 2002). Following this, SOC accumulates rapidly for a few decades (Laganière, Angers and Paré, 2010; Deng *et al.,* 2014; Li *et al.,* 2012; Nave *et al.,* 2013). SOC accumulates quickly in a surface organic layer and more slowly in mineral layers (Li *et al.,* 2012; (Bárcena *et al.,* 2014; Ledo *et al.*, 2020). The incorporation rate slows (Vinduskova and Frouz, 2013; Frouz *et al.,* 2014) as the SOC content approaches saturation or equilibrium with the site conditions (soil, climate, organic inputs, and fertility; Saeur *et al.,* 2012).

The SOC sequestration rate after afforestation is lowest in cold climates; the meta-analysis of Laganière, Angers and Paré (2010) reported average SOC losses of 1.5 percent following afforestation in the boreal zone results, compared with gains ranging from 7 percent to 17 percent in the other climate zones (the highest gains were in the temperate maritime zone (Figure 3). In a global meta-analysis, planted forests in the warm temperate zone had the highest rate of SOC accumulation (0.96 tC/ha/yr), whereas the cold temperate zone had the lowest rate (0.21 tC/ha/yr), and the tropical climatic zone (0.56 t/ha/yr) was close to the global average (0.50 tC/ha/yr) (Wang and Huang, 2020). The proportion of ecosystem C accumulated in the soil was highest in the warm temperate zone (27.0 percent) and lowest in the tropical zone (10.7 percent), with the cold temperate zone (17.9 percent) coming close to the global average (14.1 percent); the median proportion of ecosystem C accumulated in the soil was highest in the warm temperate zone. In tropical climates, mineral SOC accumulates faster and without the initial lag period often reported under temperate conditions (Don, Schumacher and Freibauer, 2011; Bárcena *et al.*, 2014a), but the greater productivity is countered by higher SOC losses. In a meta-analysis of afforestation in China, SOC accumulation rate was most strongly related to MAP, followed by MAT, stand age, and soil clay content (Wang and Huang, 2020).

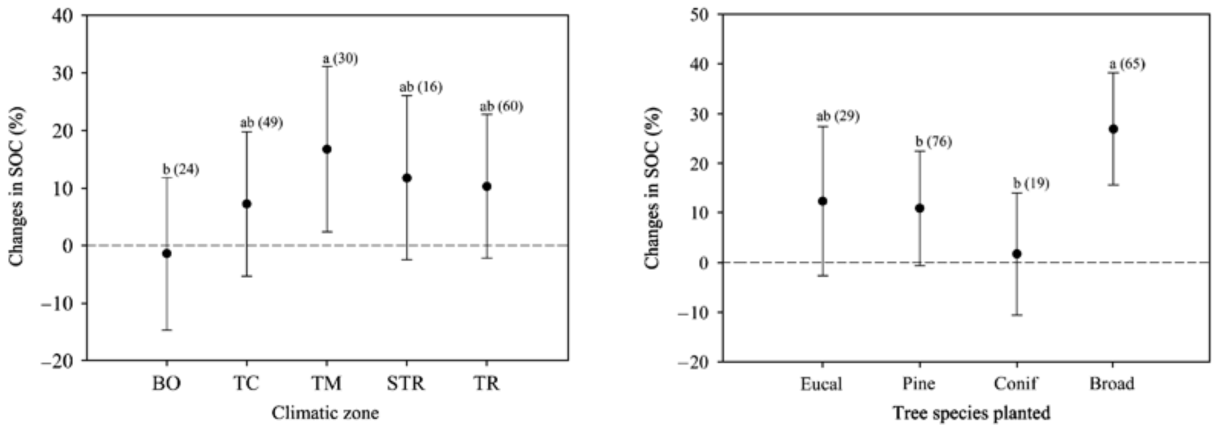

Figure 3. Change in SOC stocks after afforestation a) in different climatic zones and b) with different types of tree species. Error bars are the standard errors of the mean. Different letters indicate significant difference at $P < 0.10$. The number of observations is indicated in parentheses. The mean age of plantation is 22.9 years and the mean depth of sampling is 34.7 cm. BO, boreal; TC, temperate continental; TM, temperate maritime; STR, subtropical; TR, tropical. SOC, soil organic carbon; Eucal, Eucalyptus spp .; Conif, coniferous (excluding pine); Broad, broadleaf (excluding Eucalyptus spp.). Source: Laganière, Angers and Paré (2010)

Soil clay content is positively related to SOC change rate (Shi *et al.,* 2013; Wang and Huang 2020). Soils with high clay content (>33 percent) accumulated approximately 25 percent more C upon afforestation than coarse-textured soils (Laganière, Angers and Paré, 2010). The rate of increase in SOC following afforestation is greatest in organic layer and declines with soil depth (Shi *et al.,* 2013). Afforestation of shallow and deep peat soils may reduce SOC stocks as a consequence of drainage (Simola, Pitkänen and Turunen, 2012).

Cropland conversion to deciduous forests show faster increase in soil C stocks than to evergreen forest (Deng *et al.,* 2014; Guo and Gifford, 2002; Li *et al.,* 2012). In a global meta-analysis C, during the 2-3 decades following afforestation of agricultural soils, the average increase in SOC on sites afforested with broadleaf tree species was 25 percent, compared with 2 percent with conifers (and 12 percent with *Eucalyptus* or *Pinus* spp.; Laganière, Angers and Paré, 2010). In a global meta-analysis, soil C stock increased after afforestation with hardwoods such as *Eucalyptus*, but did not change after afforestation with softwoods such as pine (Li *et al.* 2012). On post-mining sites in the northern temperate zone, sites with deciduous forests accumulated SOC faster and deeper in the profile than sites with coniferous forests (Frouz *et al.,* 2014, Vinduskova and Frouz, 2013). The faster SOC accumulation under deciduous tree species is attributed to the high N and Ca concentrations in their leaf litter, which encourage bioturbation by earthworms (Morris *et al.,* 2007; Frouz *et al.,* 2013). Evergreen broadleaf species have higher average SOC sequestration rates (0.73 t/ha/yr) than the average for deciduous trees (0.42 t/ha/yr) or all evergreen trees (0.43 t/ha/yr), in a global quantitative review of afforestation studies (Hou *et al.,* 2020). On post-mining sites, SOC levels increase faster under trees with N-fixing root associates than under other tree species (Frouz *et al.,* 2009; Schiavo *et al.,* 2009; Kuznetsova *et al.,* 2011).

Table 17. Changes in soil organic carbon stocks reported following afforestation

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Years)	Depth (cm)	More information	Reference
Global meta-analyses								
Global	Various	Various	NA	0.50	Various	0–30	Afforestation meta-analysis	Wang and Huang (2020)
				0.49 total 0.34 organic layer 0.15 mineral layer		Standardized to top 100 cm		Li *et al.* (2012)
				0.42 0.46 0.15 0.09 0.05		Organic 0–20 20–40 40–60 60–100		Shi *et al.* (2013)
Regional meta-analyses and reviews								
Warm temperate zone	Warm temperate	Various	NA	0.96		0–30	Afforestation meta-analysis	Wang and Huang (2020)
Tropics	Tropical			0.56		NA		
			60	0.44	28		Afforestation of grassland meta-analysis	Don, Schumacher and Freibauer (2011)
			70	1.04	32		Afforestation of cropland meta-analysis	

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Years)	Depth (cm)	More information	Reference
			NA	1.3 0.41	20 100		Afforestation of agricultural land - review	Silver, Ostertag and Lugo (2000)
Cold temperate	Cold temperate			0.21 (0-30 cm)		0-30	Afforestation meta-analysis	Wang *et al.* (2020)
National and local studies								
South Carolina, United States of America	Warm temperate moist	Ultisols	32.5	1.0	40	0-60	Afforestation of abandoned agricultural land	Richter *et al.* (1999)
Southern and eastern Australia	Warm temperate moist & dry			0.57	6-45	0-30		England *et al.* (2016)
Southern Quebec, Canada	Cold temperate moist	Brunisols, Gleysols, Podzols, Regosols	80	0.18 (sandy) 0.86 (loamy)	22	0-100 (mineral soil)		Ouimet *et al.* (2007)
Michigan, United States of America		Typic Hapludalfs	51.8	0.35 deciduous 0.26 coniferous		NA	Afforestation of cropland	Morris *et al.* (2007)
Iowa, United States of America		mesic Typic Hapludolls	NA	0.56	15-50	0-30		Sauer *et al.* (2012)
Poland		Various	NA	0.34	10-50	0-100		Smal *et al.* (2019)
China	Various	Various	NA	0.13 0.26	40	0-20 0-100	Cropland conversion Grain for Green	Deng, Liu and Shagguan (2013)

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Years)	Depth (cm)	More information	Reference
Post-mining restoration								
Temperate zone of N. Hemisphere	Various	Various	NA	2.46 0.87	0–10 0–40	0–10 + organic	NA	Frouz *et al.* (2014)
				0.81 coniferous 2.32 deciduous	30			
United States of America				0.48-1.51	27–50	0–150		Amichev, Burger and Rodrigue (2008)
India	Tropical dry		66.7	2.6	8	0–30		Ahirwal, Maiti and Singh (2017)
Ohio, United States of America	Cold temperate moist		22.3	1.93 deciduous	NA			Akala and Lal (2001)
			4.86	1.19	31			Lorenz and Lal (2007)
Czechia			23.0	1.35–3.62	4–38	0–10 + organic		Sourkova *et al.* (2005)
			NA	0.16–1.32	22–31	0–20		Frouz *et al.* (2009)
				0.65 active restoration 0.93 passive restoration	40	0–10 + organic		Frouz *et al.* (2014)
				0.62	29	0–10 + organic		Frouz and Kalčík (2006)
Estonia				0.42–1.21	15–37	0–12 + organic		Karu *et al.* (2009)
				1.4–1.66	29–34	0–25		Reintam, Kar and Rooma (2002)

4. Other benefits of the practice

Afforestation of degraded soils has positive effects on a range of ecosystem services such as enhanced water quality, regulation of water flow, reduced erosion and avalanche risk, habitat for forest-related biodiversity, recreation, and production of timber and non-timber wood products.

4.1. Improvement of soil properties

By increasing rooting and organic matter content, afforestation of degraded soils improves physical properties such as bulk density, structure, porosity, permeability, and water-holding capacity (Feng *et al.,* 2011; Sauer *et al.,* 2012). Depending on the tree species planted, afforestation can raise or lower soil pH (Hong *et al.,* 2018).

4.2 Minimization of threats to soil functions

Table 18. Soil threats

Soil threats	
Soil erosion	Forestation reduced water erosion of soils within 20 years (Bonnesoeur *et al.,* 2019). Windbreaks reduce wind speeds and soil loss from agricultural land (Brandle, Hodges and Wight, 2000).
Nutrient imbalance and cycles	Afforestation increases content and cycling of nutrients (Prescott *et al.* 2019). N-fixing species increase N and other nutrients (Perakis and Pett-Ridge, 2019).
Soil salinization and alkalinization	Afforestation of saline soils with salt-tolerant trees can lower the water table, promote the downward movement of salts in the soil profile and rehabilitate these soils (Bell, 1999; Byers *et al.,* 2006; Wicke *et al.,* 2011). Mangroves desalinize water (Reef and Lovelock, 2015; also see Hotspot: Mangroves).
Soil contamination/pollution	Some tree species can be used for phytoremediation of contaminated sites (Mleczek *et al.,* 2017).
Soil acidification	Afforestation neutralizes soil pH: it lowers pH in relatively alkaline soil but raises pH in relatively acid soil (Hong *et al.,* 2018)
Soil biodiversity loss	Restores soil microbial and faunal communities (Prescott *et al.,* 2019).
Soil sealing	Some trees species adapted to these soil conditions (Prescott, Katzensteiner and Weston, 2020).

Soil threats	
Soil compaction	Bulk density declines with increasing SOC (Sauer *et al.*, 2012) Planting tree species that tolerate prolonged anoxic soil conditions can improve soil structure and aeration (Prescott, Katzensteiner and Weston, 2020).
Soil water management	Forestation reduced risk of moderate floods within 20 years (Bonnesoeur *et al.*, 2019). It can also decrease stream salinity (Ruprecht *et al.*, 2019).

4.3 Increases in production (e.g. food/fuel/feed/timber)

Afforestation of areas used for crop production reduces the area available for agricultural production and the food production potential (Sauer *et al.*, 2012). Effects on food production can be mitigated through forest landscape restoration mosaic approaches[1] in which sloping and marginal lands are afforested, leaving more productive land for agriculture. Agroforestry practices that promote food such as nuts and fruit from trees, incorporate trees into agricultural lands (e.g. intercropping, enhanced fallows, fertilizer trees, shade cropping) or integrate crops into the forest (e.g. forest gardens, taungya) provide benefits of trees while maintaining food production (e.g. Huang *et al.*, 2020; Giudice Badari *et al.*, 2020). Afforestation can include trees beneficial for fuel or fodder production. Afforestation with species of high value for timber production can be accomplished through monoculture plantations but these are less effective at storing C (Lewis *et al.*, 2019). Timber production can be balanced with other benefits of afforestation by mixing timber species with native species providing other ecosystem services (Amazonas *et al.*, 2013) or by including both in a multifunctional landscape approach (Stanturf *et al.*, 2015).

4.4 Mitigation of and adaptation to climate change

Afforestation and reforestation (i.e. reforesting unforested land) can contribute to climate change mitigation by increasing stocking density in forests, carbon sequestration in soils, and wood use in construction, or as a substitute of fossil carbon (e.g. coke) in the steel industry or energy facilities, thus avoiding fossil C-CO2 emissions (Fallot *et al.*, 2009). The IPCC's Fifth Assessment Report (IPCC, 2014) ranked afforestation as moderate for technical mitigation potential and high for both immediacy and ease. The global climate mitigation potential of reforestation (i.e. transition from non-forest to forest at a 30 percent tree-cover threshold) has been estimated at 3-10 Pg CO2eq/y (Griscom *et al.*, 2017) and 3-18 billion tonnes of CO2 per year (The Royal Society, 2018). If all areas that would naturally support forest and woodland were reforested, the extra 0.9 billion ha canopy cover would store 205 gigatons of C (Bastin *et al.*, 2019). A large-scale afforestation program that was economically, politically, and technically feasible would cover about 345 million ha and would sequester about 104 Gt of carbon (Nilsson and Schopfhauser, 1995).

[1] Also factsheet n°8 "Forest Landscape Restoration", this volume.

Afforestation and reforestation also generate changes in albedo (i.e. surface reflectivity of light). Afforestation leads to darker surfaces (lower albedo), especially at high latitudes (e.g. boreal areas); the associated alteration of radiative forcing can weaken the benefits from increased C storage (Kirschbaum *et al.,* 2011). Afforestation can also influence local and regional climate by evaporative cooling (Bonan *et al.*, 2008; Locatelli *et al.*, 2015; Alkama and Cescatti, 2016) and increasing rainfall by recycling it via transpiration (Ellison *et al.,* 2017). Overall, in afforested areas, the warming effect of decreased albedo dominates the cooling effect of increased surface roughness and evaporation at high latitudes, leading to warmer local climate, while the reverse occurs in tropical areas where surface temperatures of afforested lands are generally lower than surface temperatures of grasslands or croplands (Jackson *et al.*, 2008; Bonan, 2016; Peng *et al.*, 2014).

4.5 Socio-economic benefits

In addition to eventual products from forests (timber, fuel, fodder, food), requirements for seedlings can generate business opportunities. For example, the widespread adoption by farmers in Java of small rural woodlots has led to the development of local seedling vendors and the creation of processing industries to use the timbers (FAO, 2005).

Payment for ecosystem services such as C sequestration can also generate socio-economic benefits from afforestation. For example, the International Small Group & Tree Planting Program (TIST; https://program.tist.org/) encourages small groups of subsistence farmers to improve their local environment and farms by planting and maintaining trees on degraded land. Over 93 000 farmers in 4 countries have successfully planted more than 19 000 000 trees and captured over 5 500 000 metric tons of carbon dioxide to date. As the trees grow, carbon captured is quantified and verified and certified greenhouse gas credits are sold in the global carbon market. TIST farmers receive annual carbon pre-payments for each tree established and 70 percent of the net profit when credits are sold. Smallholder farmers also derive significant non-carbon related benefits verified to exceed USD 8 per tree. Trees provide fruit, fodder, fuel, windbreaks, shade and stabilize riverbanks plus participants have access to health information and training and safe cooking stoves.

Stakeholders interested in the potential benefits of tree planting beyond just the commercial timber companies include farmers and rural communities, water managers and hydroelectric power generating agencies and protected area managers (FAO, 2005).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Table 19. Soil threats

Soil threats	
Soil erosion	Erosion is usually low on afforested lands, except under some tropical plantations that do not have an understorey and/or a thick litter floor (e.g. some rubber and teak plantations; Liu *et al.*, 2016; Lacombe *et al.*, 2018; Neyret *et al.*, 2020).
Nutrient imbalance and cycles	Changes in SOM inputs and pH can lead to differences in soil N mineralization and nitrification (Li *et al.*, 2014).
Soil salinization and alkalinization	In sub-humid areas (i.e. aridity index between 0.5 and 0.65) replacing open arable lands/grasslands by planted forests can lower the water table and increase salt concentration in soils and groundwater (Toth *et al.*, 2014)
Soil acidification	Trees can acidify soil (Berhongaray *et al.*, 2013; Ritter, Vesterdal and Gundersen, 2003), especially N_2-fixing species (Russell, Hall and Raich, 2017; Dubiez *et al.*, 2019).
Soil water management	Higher water use of forest may reduce total water supply (Bonnesoeur *et al.*, 2019). In arid regions, afforestation can exacerbate water shortages (Cao *et al.*, 2011).

5.2 Increases in greenhouse gas emissions

Afforestation of wet sites can increase CO_2 flux from soil if drainage is required. Replacement of grassland and pasture can cause loss of C from soil.

5.3 Conflict with other practice(s)

Conversion of biomes such as grasslands and savanna can reduce soil C stocks (Guo and Gifford, 2002; Shi *et al.*, 2013) and compromise other ecosystem services, such as soil nutrient cycles (Berthrong, Jobbágy and Jackson, 2009), hydrological regulation, erosion mitigation and water yield (Bonnesoeur *et al.*, 2019) and reduce biodiversity (Bond, 2016; Veldman *et al.*, 2015).

5.4 Decreases in production (e.g. food/fuel/feed/timber)

Afforestation with native species and/or mixed species may be less cost-effective than monocultures of exotic species. This can be mitigated by growing them in mixtures (e.g. Eucalypts and native species; Amazonas *et al.,* 2013) or by underplanting native species in plantations of exotic species (Parrotta, Turnbull and Jones, 1997).

5.5 Other conflicts

Unless carefully planned to suit local situations, afforestation can have unintended negative ecological and social consequences – including reduced water supply, destruction of native grasslands, spread of invasive tree species, increased social inequity, displacement of farmland, and increased deforestation elsewhere (Holl and Brancalion, 2020).

Afforestation can create conflict where ownership is unclear or disputed, or where the land is already being used by local communities for other purposes. Attempts to reforest land subject to conflicting land-ownership claims are unlikely to be successful because of deliberate vandalism by disadvantaged parties. Collaborative and participative approaches involving local communities are necessary under such situations (Stanturf *et al.,* 2015; FAO, 2005).

Many sites available for afforestation are poor or degraded, with the better land usually being used for agriculture. Rehabilitation of the site and soil may be necessary prior to planting trees (Prescott, Katzensteiner and Weston, 2020).

Afforestation can be expensive and subject to considerable risk (e.g. fires, droughts, disease and changing markets), with long periods before any financial return is possible. Direct subsidies, joint ventures between landowners and an industrial partner, low-interest loans, micro-credit, tax concessions or payment for ecosystem services schemes may be necessary (FAO, 2005). Contracting, open bidding, and other market-based mechanisms for carrying out various operations can support development of local business (Yin, Sedjo and Liu, 2010).

Focus on acreage expansion and neglecting forest management can result in low stocking levels of both natural and plantation forests (Yin, Sedjo and Liu, 2010). Some subsidy programs lead to over-reliance on single species, poor stand quality and reversion to cropland if subsidies end (Yin, Sedjo and Liu, 2010). Emphasis should be on 'tree-growing' rather than 'tree-planting' (Lyons, 2019).

Limited silvicultural knowledge is a common impediment to successful reforestation and most foresters currently rely on a handful of well-known species for plantation development. Often, little is now known of the identity, ecology, silviculture or site requirements of many indigenous species (FAO, 2005).

6. Recommendations before implementation of the practice

- Engage local communities to develop strong local support. Resolve land ownership and tenure conflicts.
- Ensure institutional support and long-term funding to incentivize and support afforestation programs.
- Clarify distribution of costs and benefits, and responsibilities and rewards, among all parties involved in program.
- Develop appropriate methods for tree species to be used in afforestation programs.
- Assess site conditions and ameliorate to improve likelihood of trees survival and growth.
- Identify tree species that are able to tolerate conditions at degraded sites available for afforestation and will not become invasive or negatively influence the environment.
- Have a monitoring plan in place and adaptive management protocols to prevent plantation failure.
- Train local silviculturalists in site preparation, weed control, species and provenance choice, nutrition, stand tending and fire prevention.
- Consider agroforestry or forest landscape restoration approaches to accommodate food production and restoration demands.

7. Potential barriers to adoption

Table 20. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	If site has inadequate stability, fertility, soil depth, water supply or water-holding capacity, rehabilitation measures are needed prior to planting.
Cultural	Yes	Tree species should be familiar to locals and useful/desired. Trees may not be desired where landscape has been deforested for generations.
Social	Yes	Planting must be supported by local community.
Economic	Yes	Funding needed to develop capacity to produce seedlings of right species and genotype locally, for site preparation and establishment, and long-term monitoring and maintenance. Economic return only at the end of the rotation, which can last several years or decades.

Barrier	YES/NO	
Institutional	Yes	Successful reforestation programs have occurred when national governments have made a serious and prolonged effort over a number of years by providing such supportive incentives and policies.
Legal (Right to soil)	Yes	Local community should have right to use products or be subsidized.
Knowledge	Yes	Knowledge of appropriate species for environment and for sustainable use by locals, and knowledge of the best management practices.

Photo of the practice

Photo 5. Successful afforestation involving a mix of species in a lowland rain forest near Rio de Janeiro, Brazil. The site was previously farmland. This is part of a major recovery effort for the Atlantic Forest of Brazil

Table 21. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Agroforestry, silvopastoral systems and water funds initiatives contribute to improve soil capacity to remove and store carbon in Colombia	Latin America and the Caribbean	9, 20 and 40	4	36
Soil fertility improvement of nutrient-poor and sandy soils in the Congolese coastal plains	Africa	7	6	1
Afforestation of a former farmland in Japan	Asia	5	6	2
Reforestation of highlands in Javor Mountain, Republic of Srpska, Bosnia and Herzegovina	Europe	15	6	5
Natural afforestation of abandoned mountain grasslands along the Italian peninsula	Europe	23 to 72	6	6
Afforestation of vineyards in Italy	Europe	Up to 30	6	7
Afforestation by planting in bench terraces: Kalimanska watershed, Grdelica gorge, Southeastern Serbia	Europe	60	6	8

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7. Rehabilitation of forest soils affected by wildfires

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1. Description of the practice

Fires are one of the most frequent and important disturbances of forest ecosystems. When, after a severe fire, heavy rainfall occurs in a burnt area of pronounced relief, there is a potential risk of strong alterations in the hydrological behavior of the affected basins. This can lead to large increases in surface runoff and intense erosive episodes (Vega, Fernández and Fontúrbel, 2018). These phenomena favor soil degradation and can cause floods and landslides, threatening human life, infrastructure and various valuable resources within and outside the burnt area.

Traditionally, hydrological-forestry restoration after fires has been mainly focused on the recovery of the destroyed vegetation cover and on the reduction of soil and sediment losses after fire that, in most of the environments, occurred during the first year (Fernández and Vega, 2016b). Therefore, it is necessary to apply emergency soil stabilization measures to try to reduce: a) runoff and soil erosion risk and consequently, the maintenance/conservation of the quality of water and aquatic habitats and b) the degradation of soil, which is an essential element for the recovery of the affected ecosystem. These measures favor indirectly the maintenance/recovery of most physical, chemical, biochemical and microbiological soil properties, which are related to the soil organic carbon (SOC) stock and hence to the maintenance/conservation of the quality of the burnt soil. Given the risk of large-scale hydrological events, these actions allow, above all, to protect human life and a set of valuable resources that can be critically threatened in a very short time after the fire events. To this end, their objectives are to protect the burnt soil, to limit its disintegration and subsequent loss of C and nutrient stocks, uprooting and transport, as well as to reduce runoff, while stabilizing the watercourses, where appropriate. The most efficient treatment to achieve that objective is the application of a mulching of different plant materials (straw, wood strands, wood chips) over the burnt soil surface (see the reports by Robichaud, Beyers and Neary, 2000, Vega *et al.*, 2013a and Fernández *et al.*, 2019a). This reproduces the natural conditions in a pine forest affected by a low or medium severity wildfire, when the fall of pine needles forms a mulch that protects the soil from erosion; in this case, the implementation of these emergency soil stabilization measures is not necessary.

2. Range of applicability

The use of soil stabilization techniques is applicable worldwide, especially in fire-prone areas. However, apart from United States of America and NW Spain, post-fire soil stabilization measures are not being widely implemented in other fire-prone regions. During the last decade, a protocol of these urgent measures, applied by land forest managers in the integrated fight against wildfires, has been elaborated and annually implemented by forest managers in the temperature humid zones (Galicia, NW Spain) (Vega *et al.*, 2013a). The most effective and widely used technique is wheat straw mulching applied (Robichaud, Beyers and Neary, 2000; Vega *et al.*, 2013a; Fernández *et al.*, 2019a), which has shown an immediate effectiveness in increasing ground cover and reducing soil erosion losses during the first months after the fire in areas burned at high severity. Straw mulching can be spread over the soil surface via ground (hand) or aerial (helicopter) applications (Figure 4). All soil erosion mitigation measures are costly treatments (more than 3000 €/ha, 2020), but straw mulching has the highest benefit-cost ratio and its use is limited to a reduced extension of burnt surface areas, that are highly susceptible to suffer soil erosion (for example, high severity fires, great extension of burnt areas, sloping terrain, abundant high-intensity rainfall events following wildfires, proximity to surface and subsurface waters). This protocol implemented in Galicia is specific for this temperate humid zone (NW Spain). Following these emergency measures, medium- and long-term rehabilitation and restoration strategies for the recovery of the burnt forest ecosystems (soil – microorganism - tree vegetation) should apply. A detailed description of this long-term process of the recovery of the burnt forest ecosystems as well as the temporal context of measures implemented is shown in Figure 4 (Vega *et al.*, 2013a).

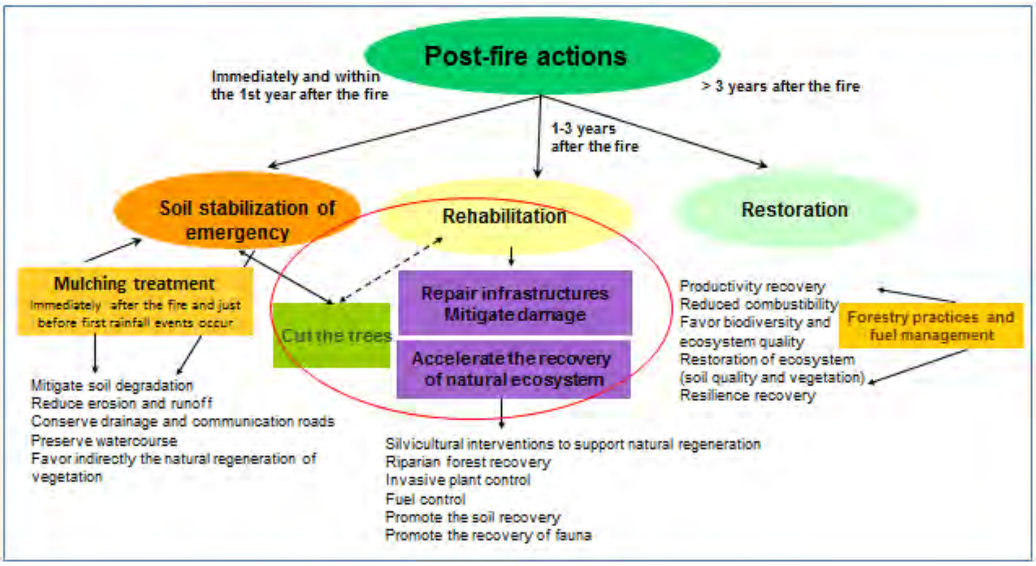

Figure 4. Temporal context of the rehabilitation and restoration strategies for the recovery of the burnt forest ecosystems Modified from Vega *et al.,* (2013a)

3. Impact on soil organic carbon stocks

Degraded burnt forest soils located in the temperate humid zone (with low levels of OC) can contribute notably to global C sequestration since they have a high potential for storage of organic C when they are subjected to soil recovery techniques to restore their pre-fire quality. Emergency strategies such as mulching application on the burnt soil surface are the first step to stabilize the soil, to avoid C loss and later on to increase C storage following medium- and long-term rehabilitation and restoration strategies. It should be noted, however, that soil potential for C sequestration is finite and limited by soil depth (marked effects only in the first 0–10 cm). Moreover, specific environmental conditions (soil type, farming or forestry system and climate) are also determinants in C sequestration because the soil organic matter (OM) content at equilibrium depends on the interaction of factors as OM inputs, rates of endogenous SOM and exogenous OM mineralization, soil texture and climate (Johnston, Poulton and Coleman,2009).

Available data on SOC changes in the short term after the fire and mulching application cover are still scarce and provided by local studies (e.g. Díaz-Raviña *et al.,* 2012; Fontúrbel *et al.*, 2012; Berryman *et al.*, 2014; Fernández-Fernández *et al.*, 2016; Lucas-Borja *et al.,* 2019a). The summary of the results of the experiments longer than 1 year are shown in Table 22. In most cases, the absence of short-term effects of mulch application on soil carbon content is the most common result. However, most of the changes in carbon stocks as a consequence of wildfire depend on fire severity and initial carbon content (Vega *et al.*, 2013b) and there are not consistent data about the effects of mulch application on high-severity burnt soils on soil carbon storage. The most determinant short-term effect of mulching treatments on the carbon cycle is related to the significant reduction in soil erosion losses (Robichaud *et al.*, 2013b; Fernández, Vega and Fontúrbel, 2016a) of up to 95 percent and in consequence the reduction in the loss of soil carbon. However, following the fire the C losses by erosion have rarely been quantified. Gómez-Rey *et al.* (2013b) measured 10-fold lower carbon losses in mulched soils compared to untreated one in a moderately burnt soil in NW Spain. Pierson *et al.* (2019) reported that mulching treatments reduced C and N losses by up to 75 percent compared with untreated areas in different areas burnt at high-severity in the western United States of America. The information on carbon sequestration on vegetation is still very scarce and inconclusive (Fernández, 2021).

Table 22. Changes in soil organic carbon stocks reported for soils affected by forest fires after mulching treatments in different environments

Only those experiences longer than 1 year are reported.

Location	Climate zone	Soil type	Control C content* OC percent (sd)	Additional C storage OC percent (sd)	Duration (Years)	Depth (cm)	Treatment after fire	Reference
SE Washington (United States of America)	Cool temperate moist	Ashy silt loam (Limberjim Series) Alfic Udivitrand	2.39 (0.12)	3.07 (0.91) p<0.01	4	0–10	Wheat straw mulching	Berryman *et al.* (2014)
				3.40 (0.27) p<0.01		0–10	Wood strands mulching	
Galicia (NW Spain)	Temperate	Udorthent	12.1 (1.3)	15.4 (0.2) nd		0–2	Seeding	Díaz Raviña *et al.* (2018)
				17.1 (0.1) nd		0–2	Straw mulching	
			11.6 (0.9)	12.9 (0.2) nd		2–5	Seeding	
				13.6 (0.2) nd		2–5	Straw mulching	

*burnt soil, no treatment after fire; nd: no statistical differences compared with the control; sd: standard deviation into brackets

The SOC sequestration during the whole process of the rehabilitation and restoration of burned forest ecosystems is long and depends on the persistence of negative soil effects induced by wildfires (more than 10 years) (Prieto-Fernández, Acea and Carballas, 1998) as well as the age of the pre-fire vegetation (around 5 years in a shrubland, 20-40 years in a pine forest or more than 100 years in an oak forest). In fact, according to our knowledge, there are no studies about SOC sequestered in restored burned forest ecosystems after such a long time period. In addition, the high spatial variation in OC content values observed for forest ecosystems (Hope *et al.*, 2015; Díaz-Raviña *et al.*, 2018) makes the quantification of SOC sequestrated difficult. We hypothesize that the soil potential of additional C storage in the temperature humid zone during the very long restoration and rehabilitation process of these burnt soils can contribute notably to global soil C storage. This is coincident with the findings of Barreiro, Bååth and Díaz-Raviña (2016) in a laboratory study performed with a burnt acidic soil following the incorporation of different mulching plant materials with a high C/N ratio (72-680, milled materials) into the soil by mixing. The data showed clearly an increased microbial activity, especially the growth of fungi, which were negatively affected immediately after the fire and have a higher potential for C sequestration than bacteria. Under field conditions, we are unaware of the existence of experimental plots where these long-term experiments can be carried out. Nevertheless, the study of Díaz-Raviña *et al.* 2018 (Table 22) allows us to quantity the maximum potential of organic C storage for the burnt soil using as reference the organic C of unburnt control soil (climax vegetation, 26.6 percent and 19.5 percent organic C in 0-2 cm and 2-5 cm soil depth, respectively). Thus, in principle, if the adequate rehabilitation and restoration strategies are implemented, around 14.4 percent and 7.9 percent organic C could be theoretically sequestered in 0-2 cm and 2-5 cm soil depth, respectively.

4. Oher benefits of the practice

4.1. Improvement of soil properties

The available information on the effects of mulching treatments on soil properties in field studies is still scarce. In addition, monitoring of the recovery of the burnt ecosystem is limited to a short-time period (up to one year when straw mulching still remains over the soil surface and it is not incorporated into the soil and decomposed). The availability of information for the medium- and long-term is even more scarce (Díaz-Raviña *et al.*, 2018). In different ecosystems, straw mulching has been demonstrated to have favorable short-term effects on soil physical, chemical and microbial properties (Bautista *et al.*, 1996, 2009; Kribeche *et al.*, 2013; Lucas-Borja *et al.,* 2019a), although some negative effects or absence of changes have been also indicated (Lucas-Borja *et al.*, 2019a). These beneficial effects are related to the ability of mulch residues to form a soil cover surface that conserves soil water and attenuates temperature changes, provides stable organic matter and nutrients, improves soil structure, and thus stimulates microbial communities. After fires, the effects of mulching on soil properties depend on soil type and previous soil properties, vegetation type and cover, fire severity and climate and environmental conditions. Table 23 shows the changes in soil properties that have been analyzed in different biomes and conditions.

Studies conducted in central and southeastern Spain, in arid and semi-arid Mediterranean ecosystems affected by wildfire reported increases of soil moisture and infiltration capacity and reductions of soil penetration resistance between one and two years after mulching treatments (Bautista, Bellot and Vallejo, 1996; Bautista, Robichaud and Blade, 2009; Kribeche *et al.,* 2013; Santana, Alday and Baeza, 2014). Prats *et al.* (2013) also report increases in soil moisture and soil shear strength and reductions in soil water repellency in a burnt area in central Portugal treated with hydromulching. In contrast, in severely burnt soils in NW Spain, no significant effect of straw mulch or wood strands mulch was observed in soil shear strength and soil penetration resistance (Fernández *et al.*, 2011; Fernández and Vega, 2021). In this region, and after a high severity fire, Díaz-Raviña *et al.* (2012), also found no change in aggregate stability, soil moisture, water-holding capacity or water repellency in soils treated with mulching and seeding.

The study of Pereira *et al.* (2018) summarizes the impacts of post-fire soil rehabilitation treatments on soil chemical and microbial properties during the first year after experimental and wildland fires in NW Spain, when the soil is most vulnerable to disturbance. The authors indicate that, in general, the effects of mulch application under field conditions are barely evident in most cases (Díaz-*Raviña et al.*, 2012, 2018; Fontúrbel *et al.* 2012; Gómez-Rey *et al.*, 2013a; Gómez-Rey and González-Prieto, 2014, 2015; Barreiro *et al.*, 2015; Lombao *et al.*, 2015). The fires considered were of low-moderate severity, except in Díaz-Raviña *et al.* (2012, 2018) and Gómez-Rey and González-Prieto (2014, 2015) studies, which were of high severity. However, other studies conducted after moderate to high-severity fires in different environmental conditions report positive effects of post-fire rehabilitation treatments. Thus, Kribeche *et al.* (2013) observed increases in soil respiration consistent with the improvement of physical properties under pine forests in SE Spain, one year after seeding and mulching treatments. Berryman *et al.* (2014) reported increases in total N and microbial respiration of wood stakes in a mixed-conifer forest in the United States of America treated with wheat straw and wood strands. Lucas-Borja *et al.* (2019a) also observed increases in soil C, pH, basal respiration, microbial biomass and enzyme activities after mulch application in a Mediterranean pine forest.

The above results highlight the importance of site characteristics and environmental conditions in the response of soil to post-fire rehabilitation treatments. In arid and semi-arid environments, the application of mulch or other forest residue materials can improve post-fire microenvironmental conditions more effectively than in temperate ecosystems, which typically have higher vegetation cover. However, in all cases there is still an urgent need of long-term assessments.

In the temperate humid zone (NW Iberian Peninsula) field studies performed at a longer time scale (4-8 years) showed that significant changes after straw mulching cover in both soil quality as well as vegetation recovery (vegetation cover and species composition) were not detectable (Díaz-Raviña *et al.*, 2018; Fernández, 2021). However, Morgan *et al.* 2014 in a field study performed with a burnt forest ecosystem located in Washington (United States of America) observed that mulch treatment apparently influenced plant cover and diversity up to six years following its application. At long-term, as was mentioned previously, a higher positive effect of the mulching cover on soil quality and hence on soil C storage is expected to be observed in humid conditions than in arid and semiarid ones. Maintenance/conservation of soil quality due to a reduction in soil losses after pre-emergence soil stabilization measures will improve soil quality if these measures are accompanied for those soil restoration and rehabilitation strategies at medium- and long term (Cerdá and Robichaud, 2009). This is supported by the findings of Barreiro *et al.* (2016) in a previous laboratory experiment.

Table 23. Results of studies about the short-term effects of post-fire straw mulching on soil physical, physicochemical, chemical, biochemical and microbial properties, analyzed in different biomes and climate conditions

Ecosystem/vegetation	Soil parameter	Change (respect to burnt untreated)	Reference
Physical properties			
SE Spain, semiarid ecosystem/pine forest	Moisture Penetration resistance	Increase Decrease	Bautista, Bellot and Vallejo (1996) Bautista, Robichaud and Blade (2009)
NW Spain, oceanic climate/shrubland	Shear strength	None	Fernández *et al.* (2011)
NW Spain, Mediterranean climate/pine forest	Aggregate stability Moisture Water repellency Water-holding capacity	None None None None	Díaz-Raviña *et al.* (2012, 2018)
Western United States of America, continental climates/coniferous forest	Water repellency	None and Decrease	Robichaud *et al.* (2013a)
SE Spain, semiarid ecosystem/pine forest	Infiltration capacity Penetration resistance	Increase Decrease	Kribeche *et al.* (2013)
SE Spain, Mediterranean climate/ Abandoned old-field terraces	Moisture Soil temperature	Increase Decrease	Santana, Alday and Baeza (2014)
NW Spain, oceanic climate/pine stands, shrubland	Aggregate stability (dry mean weight diameter)	None	Fernández *et al.* (2016a)
Central Spain, semiarid ecosystem	Moisture Hydraulic conductivity	Increase Decrease	Lucas-Borja *et al.* (2019b)
NW Spain	Penetration resistance Shear strength	None None	Fernández and Vega (2021)
Physicochemical properties			
NW Spain, Mediterranean and temperate climate / shrublands and pine stands	pH	None	Barreiro *et al.* (2015); Díaz-Raviña *et al.* (2012, 2018); Fontúrbel *et al.* (2012);

Ecosystem/vegetation	Soil parameter	Change (respect to burnt untreated)	Reference
			Gómez-Rey *et al.* (2013a); Gómez-Rey and González-Prieto (2014); Lombao *et al.* (2015)
Central Spain, semiarid ecosystem/pine forest	pH	None	Lucas-Borja *et al.* (2019a)
NW Spain, Mediterranean and temperate climate / Pine stands and shrublands	Electric conductivity	None	Barreiro *et al.* (2015); Díaz-Raviña *et al.* (2012, 2018); Lombao *et al.* (2015)
Central Spain, semiarid ecosystem/pine forest	Electric conductivity	Increase	Lucas-Borja *et al.* (2019a)
Chemical properties			
NW Spain, temperate climate /pine plantation and/or shrublands	Organic carbon Total nitrogen	None None	Barreiro *et al.* (2015); Díaz-Raviña *et al.* (2018); Gómez-Rey *et al.*, (2013a); Lombao *et al.* (2015)
Central Spain, semiarid ecosystem/pine forest	Organic carbon Total nitrogen	Increase Increase	Lucas-Borja *et al.* (2019a)
SE Washington, United States of America, cool temperate moist / mixed-conifer and grand fir	Total carbon Total nitrogen	None Increase	Berryman *et al.* (2014)
NW Spain, temperate climate/pine plantation and/or shrubland	Gross N mineralization, Ammonium immobilization, Nitrification, Nitrate immobilization	None None None Decrease	Gómez-Rey and González-Prieto (2015); Fernández-Fernández *et al.* (2016)
NW Spain, temperate climate/pine plantation and shrublands	Labile C pools (water soluble C and carbohydrates)	None	Lombao *et al.* (2015); Díaz-Raviña *et al.* (2018)
NW Spain, temperate climate/pine stands and shrublands	Macronutrients Trace elements	None None	Gómez-Rey and González-Prieto (2014); Fernández-Fernández *et al.* (2016)
Biochemical and microbial properties			

Ecosystem/vegetation	Soil parameter	Change (respect to burnt untreated)	Reference
NW Spain, temperate climate / pine stands and/or shrublands	Microbial biomass, Soil respiration, *Enzyme activities*: ·-glucosidase, acid phosphatase, urease	None None None None None	Fontúrbel *et al.* (2012); Lombao *et al.* (2015); Díaz-Raviña *et al.* (2018)
SE Spain, semiarid ecosystem/pine forest	Soil respiration	Slight Increase	Kribeche *et al.* (2013)
Central Spain, semiarid ecosystem/pine forest	Microbial biomass, Soil respiration, *Enzyme activities*: ·-glucosidase, acid phosphatase, urease	Increase Increase Increase Increase Increase	Lucas-Borja *et al.* (2019a)
SE Washington, United States of America, temperate and humid ecosystem/ mixed-conifer and grand fire	Soil respiration of a standard wood substrate	Increase	Berryman *et al.* (2014)
NW Spain, temperate climate / pine stands and shrublands	Bacterial activity	None	Díaz-Raviña *et al.* (2018)
NW Spain, temperate climate / pine stands and/or shrublands	Microbial structure (PLFAs pattern) Specific microbial groups biomass (bacteria, fungi, actinobacteria, G^+ and G^- bacteria)	None None	Barreiro *et al.* (2015); Díaz-Raviña *et al.* (2018)
NW Spain/temperate climate/pine forest and/or shrubland	Microbial biomass Microbial functional diversity	Slight increase Slight increase	Fontúrbel *et al.* (2016)

4.2 Minimization of threats to soil functions

Table 24. Soil threats

Soil threats	
Soil erosion and related soil C and nutrient losses in the top layer (0–2 cm, 2–5 cm)	A significant reduction in soil erosion, organic C and nutrient stocks losses is the most important effect of mulch application in burnt soils.
Modification of nutrient cycling and subsequent unbalance of these elements **Loss of C and nutrients (macro- and micro-nutrients) in the eroded sediments**	No short and long-term effect on N availability was found in burnt soils after mulch application (Gómez-Rey *et al.*, 2013b; Gómez-Rey *et al.*, 2014; Jonas *et al.*, 2019).
Soil biodiversity loss	Small changes in soil microbial structure or microbial community composition. Information about soil biodiversity is scarce and there is not information respect loss biodiversity. In contrast, an increase of soil biodiversity is expected at long-term with increasing organic C content.
Soil water management	In climates with a strong summer hydric stress the mulch cover maintains/increases soil moisture (Santana, Alday and Baeza, 2014; Lucas-Borja *et al.*, 2019b).

4.3 Increases in production (e.g. food/fuel/feed/timber)

By reducing both soil erosion and loss of organic C and nutrients, maintaining thus soil quality, the mulching techniques help to preserve site productivity. Increased plant regrowth after mulching has been reported in dry environments and related to an increase in soil moisture under the mulch cover (Fernández and Vega, 2014; Fernández *et al.,* 2016b). However, information concerning forest productivity is not available and that related to biomass accumulation after fire and mulching is still inconclusive (Fernández, 2021). In the long term, the positive changes in soil quality can be accompanied by changes in forest production (fuel, timber, food).

4.4 Mitigation of and adaptation to climate change

These measures have a positive impact on C sequestration when vegetation cover is enhanced and soil carbon sinks are maintained. At long-term, increased organic C and an improvement of all soil properties related to it (water retention, C and nutrient availability, soil structure…) is expected (Barreiro, Bååth and Díaz-Raviña, 2016). The magnitude of this increase in C storage should be strongly related with the initial pre-fire soil quality and effectiveness of restoration and rehabilitation strategies implemented (Cerdá and Robichaud, 2009). These positive changes in soil quality would be accompanied by changes in forest production (for example, fiber, fuel, food, honey, mushrooms, aromatic plants, nuts, truffles, forest berries) and other activities related to soil ecological functions such as C sequestration in vegetation.

4.5 Socio-economic benefits

The application of a mulch cover as an urgent measure to protect the burnt soil avoid significantly soil erosion losses, protecting water bodies and aquatic habitats. Mulching treatment tends to mitigate the floods provoked by post-fire water erosion and avoid eroded sediments reaching the aquatic ecosystems. If the latter process takes place, the burnt sediments can contaminate the surface and underground water and hence diminish the quality of water reservoirs and affect negatively the sea water quality, which may have important consequences on the shellfish industry (death of bivalves by anaerobiosis). Besides that, damage to infrastructures is also avoided. The restoration and rehabilitation techniques of forest ecosystems allow us to restore the value of the forest as a place of leisure and recreation (biodiversity, landscape, mountain villages, culture, gastronomy, *appellation d'origine* products). This is essential for people's health and in nowadays has a great economic potential to combat work stress through different activities related to ecotourism (natural parks, forest with different plants and animals, mountain sports, skiing, climbing, trekking, walks…).

4.6 Additional benefits to the practice

The soil, in addition to the production function, also performs ecological functions such as the C sequestration and mitigation of climate change, mentioned in the previous section, water purification and soil contaminant reduction, climate and flood regulation, nutrient cycling, habitat for organisms, source of biodiversity. Therefore, since these urgent measures to stabilize burnt soil try to maintain/improve soil quality of burnt soils, these ecological soil functions are recovered with the subsequent important benefits to maintain the life in the planet.

5. Potential drawbacks to the practice

5.1 Increases in greenhouse gas emissions

The possible modification of the equilibrium between mineralization (source of C) and humification (sink or C) processes of the organic matter can affect the GHE emissions (CO_2, CH_4, NO_2).

5.2 Conflict with other practice(s)

Mulching application after a fire can also reduce the possible impacts of salvage logging (Fernández and Vega, 2016b). Besides that, logging residues can also be used locally as a mulch cover (Fernández *et al.*, 2007), especially when non-commercial stands are affected (Fernández *et al.*, 2019b).

5.3 Decreases in production (e.g. food/fuel/feed/timber)

When the application rates of the straw mulch is not adequate, some delay in the recovery of the vegetative cover can occur (Dodson and Peterson, 2010). Inappropriate forestry practices that reduce soil quality will have a negative impact on production.

5.4 Other conflicts

These urgent measures need to be accompanied with further good forestry practices for the sustainable burnt soil management. It should be in accordance with current legislation.

6. Recommendations before implementation of the practice

Initially, it is necessary to identify the burnt areas susceptible to suffer post-fire erosion and apply the selected criteria to delineate the burnt area for implementation of these measures. To achieve the treatment goals, at least 75 percent of the burnt soil needs to be covered with the mulch material. It is important to check the availability of different materials taking into account that the amount of material to reach the desired cover is variable depending on the residue (from 2.5 t/ha with wheat straw to 10 t/ha with wood strands). The wheat straw is the best option for using as a mulch treatment because it generally had no significant effects on soil quality, but reduced significantly erosion, organic C and nutrient stock losses compared to that of burnt soil control (70-90 percent reduction values) (Vega *et al.* 2013a). In addition, straw mulching has a better cost/benefit ratio that observed in soils with other plant materials mulching (wood strands, wood chips).

7. Potential barriers to adoption

Table 25. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	The accessibility to the soil surface affected by the wildfire (mountainous area, high slope, absence of roads…). In this case, mulching can be applied by helicopter (heli-mulching).
Cultural	Yes	All sectors of society need to be made aware of the serious economic and ecological damage caused by wildfires and hence of the urgent need for soil protection against post-fire erosion. They should also know that the wheat straw mulching, applied manually or by helicopter, is the best option to mitigate these damages. This information has to be disseminated among the population (press, radio, television, web pages …).
Social	Yes	The perception of the necessity of soil protection is increasing worldwide. A shift into emergency measures after fire is still needed.
Economic	Yes	Availability of funds for urgent measures implementation
Institutional		The support of the administration is necessary: the elaboration of an operating protocol specific for the climate region, the formation of specialized forest managers and the inclusion of these measurements in the integrated fight against forest wildfires (prevention, extinction, impacts, recovery).
Legal (Right to soil)	Yes	The implementation of these practices must be in accordance with current legislation.
Knowledge	Yes	Due the urgent character of these measures immediately after fire, a fire severity analysis is necessary to apply these measures only in high-severity affected soils.
Other	Yes	The adverse effects of wildfires can extend far beyond those areas directly affected by the fire and cause significant ecological and economic damage affecting many sectors of society. Therefore, scientists, administration, forest managers, forest land owners and representatives of both the productive sectors should collaborate in the evaluation of environmental impact, risk assessment and in the preparation of the emergency measures protocol accompanied by medium- and long-term restoration and rehabilitation strategies. Long-term field studies monitoring the C sequestration following the rehabilitation and restoration strategies for the recovery of the burnt forest ecosystems are needed.

Photos of the practice

Photo 6. Helicopter straw mulching application after wildfire in Galicia (NW Spain)

Table 26. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Straw mulch and biochar application in recently burned areas of Algarve (Portugal) and Andalusia (Spain)	Europe	1	6	10

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8. Forest landscape restoration

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1. Description of the practice

Forest Landscape Restoration (FLR) is the process of regaining ecological integrity and functionality to enhance human well-being and livelihoods across deforested or degraded landscapes, bringing back through restoration the biological productivity of an area (IUCN and WRI, 2014; Lamb, Stanturf and Madsen, 2012; Schultz, Jedd and Beam, 2012). FLR does not necessarily seek to convert deforested and degraded lands to forests, but to balance landscape functions by integrating forests and woodlands strategically in a mosaic of land uses (Mansourian, Vallauri and Dudley, 2005), often assessing present, past, and reference land uses to assess the feasibility of restoring landscape functionality (Schulz and Schröder, 2017). A successful FLR practice would address the drivers of deforestation and landscape degradation, which often includes implementation of sustainable management of non-forest landscape units such as improved management of pastures or conservative agriculture (Mansourian, Vallauri and Dudley, 2005; Stanturf, Mansourian and Kleine, 2017). This landscape-wide approach to restoration thereby covers multifunctional and interdependent land uses and socioeconomic activities that are reintegrated as a mosaic landscape (Photo 8) to recover degraded lands with the purpose of balancing the provision of goods and ecosystem services (Hobbs, 2002; Mansourian, Vallauri and Dudley, 2005; Schulz and Schröder, 2017), thereby differing from traditional site-specific restoration efforts and *ad hoc* treatments (Lamb, Stanturf and Madsen, 2012; Stanturf *et al.*, 2015).

FLR has guiding principles that define its dynamic nature (IUCN and WRI, 2014), namely to manage adaptively for long-term resilience, tailor to local conditions using a variety of approaches, focus on the landscape level, maintain and enhance natural ecosystems, restore ecosystem multi-functionality, allow for multiple benefits, and involve stakeholders to support participatory landscape governance (Zhou *et al.*, 2008).

FLR practices are diverse and can include one or more of the following activities (IUCN and WRI, 2014; Maginnis, Rietbergen-McCracken and Sarre, 2007):

Table 27. Example of forest landscape restoration (FLR) activities across the landscape

In forest land	In agricultural land	In protective land and buffers
Afforestation/reforestation: Planting of forests and woodlots. **Natural forest regeneration:** Passive/assisted planting on degraded forests and marginal agricultural sites. **Silviculture and improved forest management:** Rehabilitation of degraded primary forests, management of secondary forests.	**Agroforestry and on-farm trees:** Multi-strata crops, live fences and windbreakers, intercropping, agrosilvopastoral systems, tree gardens. **Improved fallow:** establishment/management of trees or shrubs on fallow and shifting cultivation land. **Improved rangeland and cropland:** sustainable management to increase soil carbon and ecosystem functionality.	**Mangrove restoration:** natural regeneration through hydrological restoration, and/or mangrove planting. **Revegetation for protection and erosion control:** creation of riparian buffers, floodplain reconnection, slope revegetation.

FLR is therefore more than planting trees and restoring forests or implementing agroforestry, yet these activities are often a key component of FLR approaches. A focused review of Forest Restoration (i.e. Afforestation, Reforestation, Assisted Natural Regeneration, and Restoration of Mangroves) is available in Factsheet n°6 of this volume, whereas in Volume 3, factsheets n°38, 39 and 40 reviews Agroforestry practices. Improved agriculture practices such as crop rotation and diversification, soil organic cover, or other conservative agriculture practices are described in detail in Volume 3 of this manual.

2. Range of applicability

There are 2.2 billion hectares of degraded land around the world that have FLR potential (Figure 5), particularly mosaic landscapes (Minnemeyer *et al.*, 2011; Stanturf *et al.*, 2015) and strategic areas where restoration maximizes ecological gains (Lamb, Stanturf and Madsen, 2012) and/or where diverse landscape functions can be enhanced (Schultz, Jedd and Beam, 2012).

Numerous studies show multiple benefits associated to FLR practices (Chavez-Tafur and Zagt, 2014; Corbeels *et al.*, 2019; Gourevitch *et al.*, 2016; Lal, 2004; Nave *et al.*, 2013; Schultz, Jedd and Beam, 2012; Stanturf *et al.*, 2015; Stanturf, Lamb and Madsen, 2012) due to the increase in vegetative cover and the restoration of landscape functionality across a variety of landscapes and climates. The benefits of this practice are also identified in the restored soils, namely the enhancement of carbon sequestration and storage, improvement of water quality on the watershed, increase of ecosystem health and productivity, and increase of biodiversity. These are described in more detail in section 4 below. Based on these soil benefits, priority FLR locations to improve soil conditions and increase landscape functionality are summarized in Table 28.

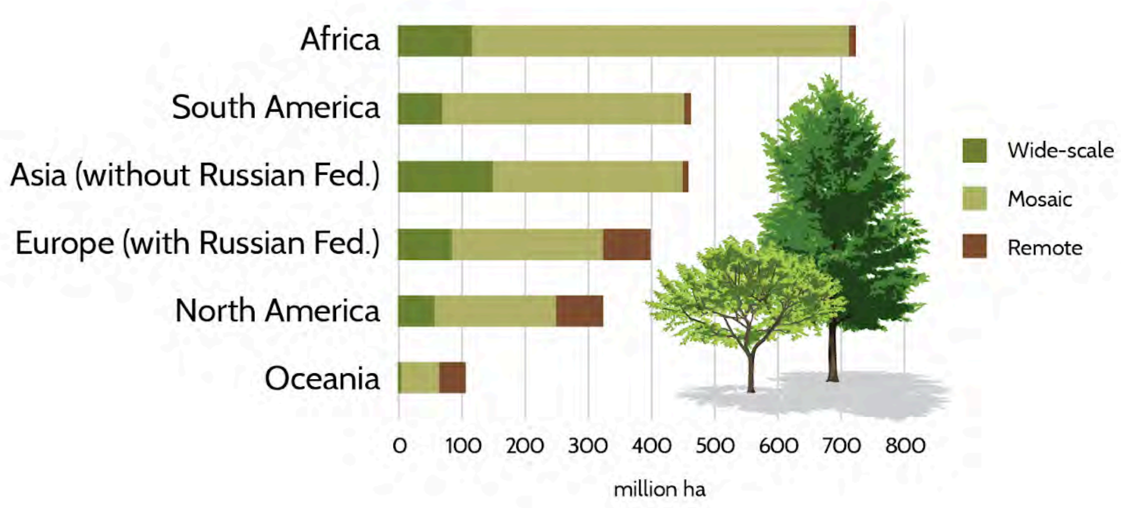

Figure 5. Global Forest Landscape Restoration opportunities. Stanturf *et al.* (2015), adapted from Bastin *et al.* (2019). "Remote restoration" refers to areas with less than one person per km2 in a 500 km radius; "wide-scale restoration" refers to areas with over

Table 28. Priority FLR areas to improve soil aspects of landscape functionality across climatic regions and geographies

Source: Adapted from Lamb, Stanturf and Madsen (2012)

Priority FLR site	Potential soil-based FLR benefits
Steep slopes	Protect erosion-prone soils, improve soil structure and aggregation
Riparian strips	Protect erosion-prone soils, filter and trap runoff and sediments, improve soil structure and aggregation, improve soil water retention
Areas prone to sheet erosion and with compacted soils	Protect erosion-prone soils and increase infiltration capacity, increase soil aeration, improve soil structure, improve soil biodiversity
Groundwater recharge areas	Increase evapotranspiration and water table depth
Coastal protection zones	Decrease soil loss and erosion, increase sedimentation and soil aggregation
Agrosilvopastoral systems	Increase land productivity, reduce erosion, increase water holding capacity, improve soil structure

3. Impact on soil organic carbon stocks

Table 29. Changes in soil organic carbon stocks reported for forest landscape restoration projects

Location	Climate zone	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration (years)	More information	Reference
Jiangxi, China	Subtropical moist	26.6, top 40 cm	0.77	19	Broadleaf afforestation in degraded land	Zhou *et al.* (2008)
			0.35		Conifer afforestation in degraded land	
Copenhagen, Denmark	Cool temperate moist	15.0, top 5 cm	0.36	29	Spruce afforestation in abandoned cropland	Vesterdal, Ritter and Gundersen (2002)
Guangdong, China	Tropical moist	4.0, top 20 cm	0.17	56	Eucalyptus sp. plantation in degraded land	Zhang *et al.* (2019)
			0.81		Broadleaf mixed secondary forest in degraded land	
Perth, Australia	Warm temperate dry	43.3, top 30 cm	0.15	26	Eucalyptus sp. reforestation in former cropland	Harper *et al.* (2012)
Nebraska, United States of America		36.2, top 15 cm	0.11	35	Windbreak agroforestry	Sauer, Cambardella and Brandle (2007)
Ontario, Canada	Cool temperate moist	68.5, top 20 cm	1.34	13	Alley cropping agroforestry	Peichl *et al.* (2006)
Limón, Costa Rica	Tropical wet	94, top 100 cm	4.63	16	Silvopastoral system	Amézquita *et al.* (2004)
Puntarenas, Costa Rica	Tropical dry	129, top 100 cm	0.10	10		
Florida, United States of America	Subtropical moist	0.6, top 30 cm	2.44	19	Planted mangroves in barren land	Osland *et al.* (2012)
Louisiana, United States of America		19.3, top 10 cm	0.12	17	Mangrove natural regeneration in marsh	Henry and Twilley (2013)

4. Other benefits of the practice

4.1. Improvement of soil properties

Implementing FLR in degraded landscapes ultimately increases organic matter (SOM) inputs to the soil in the long term, and a decrease in losses of soil and organic matter (Sanderman and Baldock, 2010). This net SOM increase drives FLR's improvement of soil properties, specifically by increasing nutrients and fertility, water holding capacity, and soil health and biodiversity (Lal, 2004; van Noordwijk *et al.*, 1997). Further, SOM improves soil aggregation, which in turn reduces erosion and soil loss (Harper *et al.*, 2012; van Noordwijk *et al.*, 1997; Zhou *et al.*, 2008), an important FLR benefit at the landscape level. Although difficult to quantify, an study estimated that 50 percent of the aggregate carbon formed in just 2.5 decades in Midwestern United States of America (Wick, Ingram and Stahl, 2009), while a 1 percent increase of SOM resulted in 18 percent higher average aggregate size in a West Indies site (Blanchart *et al.*, 2004). Moreover, Canada's government soil data (Stone and Hilborn, 2012) shows that increases in SOM content produce proportional changes in the erodibility factor *K* (average soil loss in tons per hectare) of the Universal Soil Loss Equation (USLE).

4.2. Minimization of threats to soil functions

Table 30. Soil threats

Soil threats	
Soil erosion	SOM favors aggregation, reducing risks of erosion. Vegetation cover reduces the impact and damage to the soil of erosion agents such as runoff or wind (Harper *et al.*, 2012; Lal, 2004; Zhou *et al.*, 2008).
Nutrient imbalance and cycles	Organic matter is rich in nutrients, fertilizing the soil. In the soil, it increases both the fixation of ions or nutrients and soil microbial diversity, fostering nutrient cycling (Lal, 2004; Osland *et al.*, 2012).
Soil salinization and alkalinization	FLR improves water balance in the watershed, reducing risks of salinization (Harper *et al.*, 2012). Vegetation reduces soil temperature and thus evaporation. On occasion, however, trees can accelerate surface water loss (Zhang and Shangguan, 2016) and could potentially increase soil salinity.
Soil contamination / pollution	Similar to the nutrient cycling, vegetation and SOM increases cycling and transformation of elements, filtering pollutants and reducing their accumulation in the soil (Lal, 2004; Zhang *et al.*, 2019).
Soil biodiversity loss	FLR improves soil health and biodiversity thanks to increased organic inputs and active rhizospheres (Gourevitch *et al.*, 2016; Harper *et al.*, 2012).
Soil sealing	Vegetation aerates the soil and reduces ponding (Zhang and Shangguan, 2016) and intercepts runoff and sediments (Lal, 2004), decreasing the risk of soil sealing with FLR.

Soil threats	
Soil compaction	Belowground biomass and SOM increases soil aeration and porosity, reducing soil compaction (Harper *et al.*, 2012; Nair *et al.*, 2009).
Soil water management	FLR increases water infiltration capacity (Lamb, Stanturf and Madsen, 2012) and improves water cycling as trees increase evapotranspiration and SOM increases water holding capacity (Lal, 2004; Zhang and Shangguan, 2016).

4.3. Increases in production (e.g. food/fuel/feed/timber)

FLR, albeit diverse and context-dependent, can increase the production of wood products, non-timber forest products, bioenergy materials, and game abundance and diversity (Stanturf, Mansourian and Kleine, 2017). The selection of species to bring back to the landscape is driven by the function(s) that are wanted to recover, the local customs on what species to grow, and the economic limitations on materials and/or species available. The functions to recover do not necessarily compete but they often do; for example, FLR for timber production (e.g. woodlots) is not best suited for food production or biodiversity support (Daoxiong *et al.*, 2015), while agroforestry, on the other hand, has great potential to combine fuelwood production with food and feed production (Nair *et al.*, 2009; Stanturf, Mansourian and Kleine, 2017). Because FLR looks at the entire landscape or watershed unit, a mosaic of interventions that cover a variety of functions allows to combine benefits and goods generated beyond a single field site, e.g. integrating woodlots, crops, pastures, and riparian buffers or natural forests with improved management practices, among a myriad of options.

4.4. Mitigation of and adaptation to climate change

Unsustainable removal of trees and other vegetation from the landscape or halting vegetation growth will inevitably yield a net emission of GHG if they result in deforestation and/or degradation (Stanturf, Mansourian and Kleine, 2017). Because FLR entails the recovery of efficient carbon-sequestering vegetation and the concomitant increase in soil carbon sink capacity, it represents an important climate change mitigation strategy, as enhancing vegetation cover in the landscape increases carbon capture from the atmosphere and sediment retention in the watershed.

Studies show that SOM increases its recalcitrance under FLR when woody species like trees and shrubs are integrated in the landscape (van Noordwijk *et al.*, 1997; Zhang *et al.*, 2019), a process that allows FLR to enhance SOM's long term carbon sink capacity and ability to be a carbon pool more resilient than biomass (Nair *et al.*, 2009; Vesterdal, Ritter and Gundersen, 2002). FLR is a key climate change adaptation strategy, particularly in coastal areas and floodplains where tree coverage reduces flooding as well as the impact of severe storms and accelerated sea-level rise (IUCN and WRI, 2014; Stanturf *et al.*, 2015; Stanturf, Mansourian and Kleine, 2017). By balancing land uses at the landscape level, FLR increases the resilience of the land and the communities that live and depend on it.

5. Potential drawbacks to the practice

FLR is usually in conflict with conventional agriculture and urban expansion (Minnemeyer *et al.*, 2011), land uses usually more economically productive whose focus is rarely environmental enhancement. While implementing FLR would have negligible negative environmental impacts compared to these more profitable land uses, replacing FLR with conventional agriculture (e.g. tillage, fertilizer application, and/or irrigation) would result in the soil threats listed above (Section 4) no longer being prevented or minimized. Developed lands can provide more economic benefits to a region in the short-term than FLR, yet smallholders might not necessarily experience a significant income increase with development expansion. Furthermore, a land use that can be defined as “profitable” because it provides high immediate economic benefits despite its environmental impact is likely to have significant economic costs on the long term, unless it is adapted to become resilient (Wei *et al.*, 2020). The landscape and its communities are therefore usually less resilient without FLR (IUCN and WRI, 2014; Stanturf, Mansourian and Kleine, 2017). However, where conventional agriculture is replaced by climate smart or conservative agriculture, which can include agroforestry implementation, some of the soil and ecosystem benefits FLR brings to the landscape would be provided. Natural components in the landscape replaced with urban settings, on the other hand, will virtually eliminate most FLR benefits even under urban forestry implementation. To balance both environmental and socioeconomic needs across the landscape FLR proposes a landscape view that goes beyond a single site, where trade-offs are assessed and land uses are placed strategically in the landscape to maximize multifunctionality (Mansourian, Vallauri and Dudley, 2005; Schulz and Schröder, 2017).

Alternative and more profitable land uses or practices that remove or degrade carbon stocks in the landscape will result in a net GHG emission and a loss of that sink capacity, unless the carbon in the landscape is maintained and enhanced using alternative approaches or balancing land uses.

6. Recommendations before implementation of the practice

Because of the multiple factors driving landscape degradation and the diverse actors to involve in a successful FLR implementation, wide-scale recovery of degraded landscapes is challenging. Some of these barriers for adoption and recommendations to overcome them are described below (see Section 7). In addition to these specific ones, there are a number of cross-cutting recommendations to implement FLR:

- Clearly define the scale of the intervention, functions to recover, target land uses or landscape units, and overall goal of the restoration (Hobbs, 2002).
- Monitor and evaluate outcomes to implement an adaptive management approach (Lamb, Stanturf and Madsen, 2012).
- Map stakeholders, intervention trade-offs, and multifunctional landscape hotspots to identify priority approaches that maximize beneficiaries (Schulz and Schröder, 2017).

- Engage the multiple stakeholders and actors of the landscape to address early their interests facilitates collaborative management and balancing land use trade-offs (Maginnis, Rietbergen-McCracken and Sarre, 2007; Zhou *et al.*, 2008).
- Ensure capacity to manage conflict and negotiate is available, to successfully integrate and generate consensus around diverse perspectives of what a functional landscape is under the given circumstances and idiosyncrasies (Mansourian, Vallauri and Dudley, 2005).
- Smallholders, key for FLR in fragmented landscapes, face often implementation challenges; ensuring adequate supply of resources (e.g. seedlings, equipment, guidance) and creating partnerships to support production and distribution of resources facilitates smallholder engagement and efficient implementation of best management practices (Stanturf *et al.*, 2015).

7. Potential barriers to adoption

Table 31 describes the main types of barriers to adopt FLR, namely biophysical, social, economic, institutional, legal, and technical, and proposed approaches to overcome them.

Table 31. Potential barriers to adoption

Barrier	Explanation	Options to overcome barriers
Biophysical	Herbivory, wildfires, extreme weather, historical drivers of site degradation, and severity of soil degradation prior FLR implementation are barriers to address in early implementation stages (Gong *et al.*, 2013; Stanturf *et al.*, 2015; Stanturf, Lamb and Madsen, 2012).	Assessment of current and historical landscape conditions give insights on the drivers of degradation and on the feasibility and potential success of restoration approaches (Hobbs, 2002; Schulz and Schröder, 2017). Integrating potential future scenarios under altered climate can improve the resilience of the FLR strategy.
Social	Involvement of local entities and communities is key for FLR implementation (Zhou *et al.*, 2008). Smallholders face challenges engaging in restoration when institutional support is scarce and where conflict of interest with unclear management and rights arise (Corbeels *et al.*, 2019).	Community engagement in planning and implementation, as well as benefit-sharing, can reduce unsustainable resource exploitation and its resulting degradation (Hawkins *et al.*, 2010; Stanturf, Lamb and Madsen, 2012). Stakeholders need to be mapped to include all perspectives in a trade-offs assessment (Lamb, Stanturf and Madsen, 2012; Mansourian, Vallauri and Dudley, 2005).

Barrier	Explanation	Options to overcome barriers
Economic	Funds beyond planting are often necessary until the ecosystem can self-regenerate (Chavez-Tafur and Zagt, 2014). Furthermore, there can be competing demands for the land that make alternatives to FLR more economically profitable or apparently better suited to support livelihoods (Schedlbauer and Kavanagh, 2008; Stanturf *et al.*, 2015).	Implementation of FLR plans that alleviate food insecurity and poverty would have a direct positive economic impact at the local level (Smith, 2008). A trade-offs assessment can help identify economic constraints of stakeholders and incentives to overcome them (Mansourian, Vallauri and Dudley, 2005). Potential options to cover implementation and adoption costs are payment for ecosystem services schemes, private sector investment, and/or government support, among others.
Institutional	Successful FLR implementation typically requires engagement of multiple stakeholders, participatory planning, and coordination of multiple institutional levels and agencies (IUCN and WRI, 2014; Stanturf *et al.*, 2015).	Government engagement in adopting FLR targets can help in the creation of decision-making structures and taskforces to coordinate efforts across institutions or agencies and develop integral strategies (Stanturf, Mansourian and Kleine, 2017).
Legal	Insecure land tenure is frequent in mosaic landscapes and can threaten the success of FLR and lead to conflict.	Clear governance and oversight needs government engagement and support to reduce legal insecurities and favor fair conflict resolution.
Knowledge	Due to the diverse nature of FLR, limited standardized guidelines and implementation protocols are available (Corbeels *et al.*, 2019; IUCN and WRI, 2014; Stanturf *et al.*, 2019), challenging the ability to increase capacity and thus implementation.	International, national, and subnational institutions can facilitate the dissemination of FLR guidance materials, training resources, and standard operating procedures. ´Trainer Of Trainers´ approaches can improve in-country capacity building and knowledge management, further engaging local communities.

Photos of the practice

Photo 7. Forest natural regeneration. California, United States of America

This conifer forest in Southern California was burned by wildfires. The forest, under protection, and was allowed to regenerate naturally without assisted planting. Burned trees, standing or downed, were left on the landscape to maintain the ecology of the site while young trees mature.

Photo 8. Mosaic landscape. Guilin, China

This heterogeneous landscape in rural China presents a mosaic of land uses combining biophysical and socioeconomic components, integrating landscape productivity to sustain local livelihoods through rice paddies, crops, and farms, with tree coverage along crops or as forested areas.

Photo 9. Mangrove restoration. Quang Ninh, Vietnam

Shrimp and aquaculture farming along coastlines is a major driver of mangrove deforestation around the world. The picture shows mangroves thriving after restoring the biophysical setting of the abandoned aquaculture pond area.

Photo 10. Multistrata agroforestry, shaded coffee. Quindío, Colombia

Crops like coffee or cocoa, among others, maintain productivity and quality when grown under the shade of trees. Coffee shrubs in this plantation increased their productivity thanks to the ability of trees to maintain optimal coffee growth temperatures by shading the crop.

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Wetlands

9. Avoiding conversion and conservation of wetlands

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1. Description of the practice

Avoiding conversion, which includes avoiding drainage of wetlands for use in agriculture, aquaculture and other land-uses (

Photo **11**), and conservation of wetlands in their natural state (Photo 12 and Photo 13) maintains ecosystem function of wetlands. Ecosystem functions, including carbon sequestration, nutrient cycling, water purification, flood mitigation, habitat for fisheries and biodiversity, and coastal protection provide essential ecosystem services that support productive landscapes and human well-being (Barbier *et al.,* 2011; Zedler, 2003). Intact wetlands also trap sediments and build soils, thereby maintaining coastal elevation (Shepard *et al.,* 2011). Wetland types included in this practice include all marine and coastal wetlands, and inland wetlands, classified as Lacustrine, Riverine, Palustrine, Marine and Estuarine (Ramsar Convention on Wetlands, 2018), excluding peatlands (see Chapter 4.2.1 Peatland) and human-made wetlands. Hydrological modifications of wetlands (e.g. drainage or excavations for aquaculture or salt ponds), land clearing, fertilizing, grazing, and any other factors that cause degradation of wetland vegetation and soils result in decreases in ecosystem functions. Particularly important is managing hydrology which if altered can result in changes in plant and fauna communities. Additionally, activities like drainage and construction of pond walls to prevent water flows can lead to increases in aeration of soils which results in CO_2 emissions as organic matter in soils decomposes (IPCC, 2014). Soil structure can also be damaged resulting in erosion and subsidence, and thereby increased vulnerability to sea level rise, severe storms, and other impacts of climate change. Loss of soil organic carbon (SOC) as a consequence of wetland damage varies regionally with climate, and locally with soil salinity and texture, and the organic matter available (Lovelock *et al.,* 2017). Identifying the location, extent and ecological character of wetlands, and the ecosystem services they provide to people, such as their SOC sequestration and storage, are important for valuing the socio-economic benefits from avoiding conversion and conservation of wetlands and to support decision-making in conservation and sustainable management actions.

2. Range of applicability

The most recent estimate of global natural wetland area is 15 x 10^6 km^2 – 16 x 10^6 km^2, of which about 91 percent is inland and only 9 percent is coastal and marine (Davidson and Finlayson, 2019). Decline in global wetlands is occurring across almost all classes of inland and marine or coastal natural wetlands (Davidson and Finlayson, 2018). The Satellite-based Wetland Observation Service (www.swos-service.eu) has been developed to assist countries with mapping and monitoring changes in their wetlands (Weise *et al.,* 2020).

Despite their small global area compared to terrestrial forests and grasslands (Griscom *et al.,* 2017), wetlands have high importance in the global carbon cycle and greenhouse gas (GHG) emissions (Duarte *et al.,* 2013; Moomaw *et al.,* 2018) and in nutrient cycling (Jickells *et al.,* 2016; Reddy *et al.,* 1999, also see Hotspot "Wetlands"). Conservation of wetlands is an important management practice because of their rapid global degradation (Davidson, 2014; Gedan *et al.,* 2009; Millennium Ecosystem Assessment, 2005; Valiela *et al.,* 2001; Waycott *et al.,* 2009), which has led to significant CO_2 emissions (Pendleton *et al.,* 2012) as well as losses in other ecosystem services. An increasing number of countries are motivated to develop sustainable management actions for wetlands in order to conserve and restore wetlands (Finlayson, 2012). For example, by listing wetlands of international importance under the Ramsar Convention on Wetlands and managing for sustainable use. Although the ecological-character of Ramsar listed wetlands have been found to be significantly better than those of wetlands generally, overall there has been a widespread deterioration of wetlands (Davidson *et al.,* 2020), and there are calls for further action to reverse ongoing wetland loss and degradation (Finlayson *et al.,* 2019). Monitoring of change in wetland extent and condition and the drivers of change (e.g. Goldberg *et al.,* (2020) for mangroves) can assist in the design of management strategies to avoid losses of wetlands and thereby support their conservation. Priorities for conservation are areas where losses have been substantial, such as saltmarsh conversion in Europe (Gedan and Bertness 2009), mangrove conversion in south-east Asia (Thomas *et al.,* 2017), or where biodiversity is particularly vulnerable, for example wetland sites of international significance for migratory birds (Waliczky *et al.,* 2019). Wetland conservation is also important where the provision of ecosystem services to communities is highly valued, for example flood protection from tidal marshes and mangroves in cyclone risk areas (Hochard *et al.,* 2019; Narayan *et al.,* 2017), water purification within watersheds (Zedler, 2003), or where carbon stocks are particularly high and therefore where degradation leads to high levels of CO_2 emissions (e.g. Sasmito *et al.,* (2020) and Serrano *et al.,* (2019) for mangroves, tidal marshes, and seagrasses).

The management of suitable hydrological regimes and nutrient levels are necessary to maintain wetland plant communities and avoid SOC losses (Alhassan *et al.,* 2018; Millennium Ecosystem Assessment, 2005) and high levels of methane and nitrous oxide emissions (Beaulieu *et al.,* 2019; Dalal and Allen, 2008; Ma *et al.,* 2018). Conservation of wetlands can encompass sustainable use of wetlands to support livelihoods and generate income such as bio-charcoal production, timber and non-timber product creation, sustainable fisheries and aquaculture, and tourism (Gosling *et al.,* 2017; Thompson and Friess, 2019).

3. Impact on soil organic carbon stocks

Conservation of wetlands maintains SOC, avoiding CO_2 emissions, and secures carbon sequestration due to long-term SOC accumulation (Bridgham *et al.,* 2006; McLeod *et al.,* 2011; Nahlik and Fennessy 2016) and in the case of saline tidal wetlands, is associated with low levels of methane and nitrous oxide emissions (Kroeger *et al.,* 2017). SOC stocks and sequestration rates varies with wetland type (Table 32), being generally higher in tidal coastal wetlands than freshwater wetlands per unit area, and is influenced by latitude and precipitation (Atwood *et al.,* 2017; Hinson *et al.,* 2019), hydro-geomorphology (Sasmito *et al.,* 2020) and within-wetland variation in vegetation (Pearse *et al.,* 2018). A review of key data sources (Table 32) shows that overall more data on SOC stocks and accumulation are needed for tidal marshes and freshwater wetlands in tropical environments. Globally the area of near pristine wetlands – up to 1.15 million km^2 of coastal wetlands (seagrass, mangrove, tidal marsh), and 3.6 million km^2 of freshwater wetlands, sequesters 0.86 and 0.72 Gt (gigaton) CO_2 per year in the soil respectively (Bridgham *et al.,* 2014; McLeod *et al.,* 2011). There is high uncertainty in estimates of wetland extent in many nations because of the limited number of national inventories; although governments and non-government organizations are improving inventories (e.g. see Canada Wetland Inventory, https://maps.ducks.ca/cwi/). The global degradation of coastal wetlands is likely to have emitted an estimated 0.15-1.02 Gt CO_2 per year (Pendleton *et al.,* 2012), equivalent to 3-19 percent of global anthropogenic CO_2 emissions due to deforestation (van der Werf *et al.,* 2009). Reducing impacts on coastal wetlands could result in 141-466 Tg (metric ton) of avoided CO_2 emissions per year associated with decomposition of above and below ground biomass and SOC (Griscom *et al.,* 2017). Sediments in coastal wetlands can accumulate 1.9-3.9 mm/year (Breithaupt *et al.,* 2012), which contributes to their carbon sink capacity. Accumulation of SOC in coastal wetlands is likely to increase with sea level rise (Rogers *et al.,* 2019), although thresholds of SLR are likely to be 6.1-7.6 mm/year (Saintilan *et al.,* 2020). Wetland conservation is therefore particularly important given the high levels of carbon stocks in soils that may have accumulated over 100 to 1000s of years, which if disturbed liberate CO_2 to the atmosphere (Lovelock *et al.,* 2017) (see also Mangrove Hotspots and Wetland Hotspots presented in volume 2 of this manual).

Table 32. Changes in soil organic carbon stocks reported for coastal and freshwater conserved wetlands across the world

Wetland & Location*	Climate zone	Baseline SOC stock (tC/ha)	Additional SOC storage (tC/ha/yr)	Duration	Depth (cm)	More information	Reference
Coastal and inland wetlands							
Tidal marsh, mangrove and freshwater, Global	Temperate, subtropical, tropical	NA	Means: All wetlands 1.85; Tidal marsh 2.48; Mangrove 2.30; Freshwater marsh 1.97; Peatland 0.77	Literature to 2017	NA	473 soil/sediment cores from various wetlands	Cheng *et al.* (2020)
Coastal and inland wetlands, United States	Temperate	Means (± SE): Eastern Mountains and Upper Midwest (478 ± 58); Interior Plains (195 ± 25); Coastal Plains (198 ± 21); West (216 ± 30); Tidal saline wetlands (340); Freshwater inland wetlands (295)	NA	2011	120	967 soil pits from wetland sites across the US	Nahlik and Fennessy (2016)
Coastal wetlands							
Tidal marsh, mangrove, seagrass, Global	Temperate, subtropical, tropical	NA	Mean (± SE): Tidal marsh 2.18 ± 0.24 (range 0.18–17.13);	Literature and unpublished data 2003-2010	NA	Data from 96 tidal marsh sites, 34 mangrove sites and 123 seagrass sites	McLeod *et al.* (2011)

Wetland & Location*	Climate zone	Baseline SOC stock (tC/ha)	Additional SOC storage (tC/ha/yr)	Duration	Depth (cm)	More information	Reference
			Mangrove 2.26 ± 0.39 (range 0.2–9.49); Seagrass 1.38 ± 0.38 (range 0.45–1.90)				
Mangrove, Global		Mean (± SD): 283 ± 193	NA	Literature from 1994-2016	100	1 230 sampling locations from 48 countries. Gives variation across countries	Atwood *et al.* (2017)
		Mean (± SD): 361 ±136 (range 86 –729)	NA	Literature and unpublished data from 1990-2016	100	Spatial modelling based on global mangrove SOC database (1 812 soil profiles from 47 countries). Gives variation across countries	Sanderman *et al.* (2018)
		Means (± SE): Global 333.7 (± 11.2); West Africa 278.4 (± 16.5); Asia 294.8 (± 20.4); Middle East 110.8 (± 11.6); Southeast Asia 375.6 (± 17.5); Oceania 447.9 (± 19.0); Americas 350.9 (± 19.9); Central America 401.9 (± 20.0); South America 154.9 (± 11.7)	NA	2007–2017	100	Soil cores from 190 sites across five continents	Kauffman *et al.* (2020)
		NA	Mean (± SD): 2.31 ± 2.09	Literature and unpublished data from 1989-2012	NA	65 sediment cores from Brazil, Colombia, Malaysia, Indonesia, China, Japan, Vietnam, and Thailand,	Breithaupt *et al.* (2012)

Wetland & Location*	Climate zone	Baseline SOC stock (tC/ha)	Additional SOC storage (tC/ha/yr)	Duration	Depth (cm)	More information	Reference
						and additional data from Mexico and the US	
Tidal marsh and mangrove, Brazil	Tropical	Mean: Tidal marsh 257; Mangrove 341	NA	unknown	135–›300	Field surveys 9 mangrove and 3 tidal marsh sites mouth of Amazon River	Kauffman *et al.* (2018)
Tidal marsh, mangrove, seagrass, Australia	Arid, semi-arid, temperate, subtropical, and tropical	Mean (± SD): Tidal marsh 168 ± 127; Mangrove 251 ± 155; Seagrass 112 ± 88	Mean (± SD): Tidal marsh 0.39 ± 0.3; Mangrove 1.26 ± 0.9; Seagrass 0.36 ± 0.3	Literature and unpublished data	100	Database of 1 553 wetland sites (593 tidal marsh, 323 mangrove, 637 seagrass) on SOC stocks (1103 cores), and SOC sequestration rates (352 cores). Gives variation across regions	Serrano *et al.* (2019)
Tidal marsh, mangrove, seagrass, South-eastern Australia	Temperate	Means (± SD): Tidal marsh 57.96 ±2.90 (range 23.33 - 291.18); Mangroves 50.64 ±1.35 (range 23.34 -77.81); Seagrass 23.48 ±0.57 (range 23.33 - 73.42)	NA	2014	30	Spatial modelling based on 287 sediment cores from 96 coastal wetlands across Victoria (125 tidal marsh, 60 mangrove, and 102 seagrass)	Lewis *et al.* (2020)
Tidal marsh, mangrove, tidal freshwater wetlands, United States		Mean 270	NA	Literature and unpublished data 1998–2017	100	Spatial modelling based on 1 959 soil cores from 49 studies across US	Holmquist *et al.* (2018)

Wetland & Location*	Climate zone	Baseline SOC stock (tC/ha)	Additional SOC storage (tC/ha/yr)	Duration	Depth (cm)	More information	Reference
Freshwater wetlands							
Freshwater wetlands, Global	Temperate and tropics	NA	Tropical: mean 1.29 (range 0.42-3.06); Temperate: 1.43.	2004–2009	50–300	9 soil cores each in 5 wetland sites in northern Ohio, eastern and western Costa Rica, and Botswana	Mitsch *et al.* (2013)
Freshwater wetlands, South-eastern Australia	Temperate	Alpine wetlands 290 ± 180; Shallow freshwater marsh 200 ± 200; Saline wetlands 64 ± 48; Freshwater meadow 130 ± 100; Deep freshwater marsh 230 ± 190; Permanent open freshwater 110 ± 120; All wetlands 186 ± 176	Permanent open freshwater sites 2.3 ± 0.7; Shallow freshwater marshes 0.91 ± 0.27; deep freshwater marsh 1.6 ± 0.5; All wetlands 1.9 +/1 0.4	2015–2016	50–100	>1 600 samples across 103 temperate, alpine, and semi-arid wetland sites	Carnell *et al.* (2018)
Saline and freshwater marshes, North-east China	Temperate	NA	Freshwater marsh: mean 2.03; Saline marsh: mean 0.62; Overall mean 1.65 ± 0.67.	2011–2012	40	12 wetland sites in Heilongjiang and Jilin Provinces	Zhang *et al.* (2016)
Freshwater wetlands, North America	Temperate	NA	Mean 0.17 (range 0–6.16)	From 1996	NA	Literature and published sources	Bridgham *et al.* (2006) Bridgham *et al.* (2014)

*Soil types are rarely published in studies and therefore have not been included in the table.

4. Other benefits of the practice

4.1. Improvement of soil properties

Avoiding conversion and conservation of wetlands contributes to sediment accretion, nutrient retention, biodiversity of soil fauna (including bioturbators), maintenance of soil structure suitable for plant recruitment and high rates of water infiltration, and avoids soil compaction (Spivak *et al.,* 2019). Conserved riparian forests in Brazil (i.e. those with low degradation) have been shown to have higher soil carbon contributing to nutrient supply and soil structure, and maintaining water quality and stream habitat in riverine wetlands (Celentano *et al.,* 2017).

4.2. Minimization of threats to soil functions

Table 33. Soil threats

Soil threats	
Soil erosion	Wetland vegetation cover is crucial for avoiding soil erosion and compaction. For example, conversion of saltmarshes to agriculture has resulted in soil loss and up to 9m subsidence in the Sacramento-San Joaquin River Delta in California (Ingebritsen *et al.,* 2000).
Nutrient imbalance and cycles	Intact wetlands retain nutrients and help purify surface waters (Alongi and McKinnon 2005; Saderne *et al.,* 2020). For example, conversion of wetlands in the River Thames has contributed to nutrient pollution in the estuary (Jickells *et al.,* 2016). Similarly, water quality declined in the Upper Midwestern region of the US, when about 60 percent of wetlands were drained, mostly for agriculture (Zedler 2003).
Soil salinization and alkalinization	Coastal wetlands can protect adjacent lands from tidal inundation through attenuation of tidal flows and storm surge and waves (Gedan *et al.,* 2011; Silliman *et al.,* 2019; Temmerman *et al.,* 2013).
Soil contamination/pollution	Intact wetlands retain metal and organic pollutants (Barbier *et al.,* 2011; Rabaoui *et al.,* 2020). In many cases wetlands are constructed to assist with treatment of pollutants (Vymazal and Bfezinova 2015).
Soil acidification	Wetlands tend to be acid due to the biogeochemical processes associated with organic matter decomposition (Spivak *et al.,* 2019). However, coastal wetlands can export alkalinity which contributes to the net CO_2 sink capacity of wetlands (Maher *et al.,* 2018). Drainage of wetlands can also cause wetlands to become highly acidic, because of acid sulphate soils being exposed to air, releasing sulphuric acid (Cook *et al.,* 2000).

Soil threats	
Soil biodiversity loss	Conservation of wetlands maintains biodiversity of soil fauna and microbes which contribute to the unique biogeochemistry of wetlands (Spivak *et al.*, 2019) and strongly influence soil organic matter decomposition (Jackson *et al.*, 2017). Benthic fauna in coastal wetlands are important in food webs that support coastal fisheries (Abrantes *et al.*, 2015).
Soil compaction	Conservation of wetlands avoids compaction, which reduces ecosystem functions of water infiltration as well as recruitment of seedlings and root growth (Ola *et al.*, 2020).
Soil water management	Intact wetlands have unique hydrological regimes to maintain soil conditions and native plant communities (Spivak *et al.*, 2019), which prevents dieback (Duke *et al.*, 2017) and loss of SOC (Alhassan *et al.*, 2018).

4.3. Increases in production (e.g. food/fuel/feed/timber)

Many wetlands are important for subsistence activities of local communities which can be compatible with conservation if managed appropriately. These activities include extraction of timber (e.g. fuel wood, charcoal, building materials) and non-timber forest products (e.g. honey, waxes, plant fibres), fishing and collection of crabs, and ecotourism (Gosling *et al.,* 2017; Uddin *et al.,* 2013). Coastal wetlands are also important nursery grounds and food sources for fisheries (Whitfield, 2017), and support commercial fisheries (Barbier *et al.,* 2011). Wetland conservation can support food security and incomes of communities contributing to multiple sustainable development goals (Friess *et al.,* 2019).

4.4. Mitigation of and adaptation to climate change

Wetland conservation contributes directly to climate change mitigation and adaptation because wetlands are significant carbon sinks, and provide coastal protection from flooding and erosion, buffering the impacts of sea level rise, increased storm surges, and wave activity associated with climate change (Duarte *et al.,* 2013; Menéndez *et al.,* 2020). Avoiding conversion and drainage of wetlands prevents CO_2 emissions, and also prevents increases in methane and nitrous oxide emissions from conversion to rice, aquaculture, other agricultural land uses or freshwater wetlands (IPCC, 2014, 2019; Moomaw *et al.,* 2018), which can contribute to global warming.

4.5. Socio-economic benefits

Avoiding conversion of coastal and inland wetlands secures provision of their ecosystem services which have initially been valued globally at US $13,165 billion per year, around 60 percent of which is from coastal wetlands (Costanza *et al.,* 1997). Intact wetlands provide a number of key ecosystem services to communities that include coastal protection from storms (valued at US \$8,240 per ha annually in the US (Costanza *et al.,* 2008)), fisheries (valued at AU \$25,741 and AU \$5,297 per ha annually for saltmarsh and mangrove respectively in Australian estuaries (Taylor *et al.,* 2018)), flood mitigation (valued at \$65 billion per year for mangroves (Menéndez *et al.,* 2020)), erosion control, water purification (pollutant control), forest products, and carbon sequestration (Barbier *et al.,* 2011; Zedler 2003). In Thailand, mangroves have an aggregate value of \$19,000 per ha which was estimated for provision of coastal protection, wood products and habitat-fishery linkages (Barbier *et al.,* 2008). They also support tourism valued at US \$42,000 per year for the Sundarbans mangrove reserve in Bangladesh (Uddin *et al.,* 2013), but much higher (US \$104 million per year) for Can Gio Mangrove Biosphere Reserve in Vietnam (Kuenzer and Tuan 2013), or when considering the number of visitors attracted to mangroves worldwide (Spalding and Parrett 2019). The conversion of coastal wetlands globally, and associated CO_2 emissions, has been estimated to result in economic damages attributable to climate change of US \$6–42 billion annually (Pendleton *et al.,* 2012).

5. Potential drawbacks to the practice

5.1. Increases in greenhouse gas emissions

Wetlands are carbon sinks, but can emit methane (IPPC, 2014), particularly when polluted and at higher temperatures (Al-Haj and Fulweiler, 2020). Methane emissions decrease with increased salinity, and have been found to decrease as salinity in mangrove and tidal marsh soils increase (Al-Haj and Fulweiler 2020; Poffenbarger *et al.,* 2011; Purvaja and Ramesh, 2001). Overall methane emissions from tidally influenced mangrove and tidal marsh are low, and even lower in seagrass (Al-Haj and Fulweiler, 2020; Negandhi *et al.,* 2019). When balancing carbon sequestration and methane emissions, seagrasses have been found to remain as net GHG sinks (Al-Haj and Fulweiler, 2020). At low elevations, mangroves and tidal marsh also remain net GHG sinks (Negandhi *et al.,* 2019; Rosentreter *et al.,* 2018), however may become net GHG sources in some systems (Al-Haj and Fulweiler, 2020). Over a long time period, freshwater wetlands have also been found to become net GHG sinks (Mitsch *et al.,* 2013). Whilst drainage of wetlands or vegetation degradation generally reduces methane emissions, nitrous oxide emissions are higher, especially near human settlements and agriculture regions (Dalal and Allen, 2008; Ma *et al.,* 2018). Methane and nitrous oxide fluxes have large spatial and temporal variability in wetlands (Dalal and Allen, 2008; Kayranli *et al.,* 2010) and further measurements are required to fully account for GHG fluxes in wetlands.

5.2. Conflict with other practice(s)

Avoiding conversion and conservation of wetlands conflicts with aquaculture and agricultural land uses. In coastal regions there are conflicts with shrimp farming, rice production, palm oil and salt production (Goldberg *et al.,* 2020; Richards and Friess, 2016; Thomas *et al.,* 2017; Valiela *et al.,* 2001).

5.3. Decreases in production (e.g. food/fuel/feed/timber)

Whilst wetland conservation might be perceived to have a negative impact on seafood production and export incomes, many aquaculture ponds are disused in south-east Asia due to low water quality and disease (Primavera *et al.,* 2011). Therefore, productivity and incomes from low productivity aquaculture can be similar to those from maintaining natural ecosystems for fishing (Thompson *et al.,* 2017). In many nations, community-based forest agreements allow local communities to sustainably manage their local forests, including wetlands, to support their livelihoods from collection of fuelwood, charcoal, construction materials, and fishing (Datta *et al.,* 2010; Feurer *et al.,* 2018; Roy *et al.,* 2012). Furthermore, rice and other agriculture on coastal floodplains are at risk of salinization with subsidence and rising sea levels, and increased pollution, which is negatively impacting food security (Boretti, 2020). Wetland conservation and restoration enables the provision of ecosystem services necessary to maintain food and forest production (see Sections 4.3 and 4.5). For example, managed realignment to restore saltmarshes in Europe reduces the costs of coastal defence against rising sea levels whilst providing environmental benefits (Watts *et al.,* 2003). In addition, rehabilitation of abandoned aquaculture ponds enhances carbon storage and coastal protection in the coastal zone (Duncan *et al.,* 2016).

6. Recommendations before implementation of the practice

Identifying the location, extent and ecological character of wetlands, and the ecosystem services they provide to people, such as their carbon sequestration and storage, are important to quantify the socio-economic benefits from the conservation and sustainable management of wetlands and support decision-making. Quantifying the potential payments for ecosystem services from wetlands are also necessary to enable wetland conservation and restoration to be considered as economically productive systems, alongside other land uses (Millennium Ecosystem Assessment, 2005).

The maintenance of suitable hydrological regimes and nutrient and sediment levels should be prioritized to maintain native plant communities and avoid SOC losses and high levels of methane and nitrous oxide emissions (Al-Haj and Fulweiler, 2020; Alhassan *et al.,* 2018; Ma *et al.,* 2018; Millennium Ecosystem Assessment, 2005). An ecosystem-based approach to wetland conservation should be applied that encompasses connectivity among multiple wetlands (i.e. basin-scale management, or integrated coastal zone management) and considers the trade-offs between wetland ecosystem services are more likely to maximize the provision of environmental and socio-economic benefits (Thorslund *et al.,* 2017). In addition, climate change impacts on hydrology, wetland extent and carbon cycling should be considered to adapt conservation measures to allow long-term wetland SOC storage (Moomaw *et al.,* 2018).

7. Potential barriers to adoption

Table 34. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Changes in rainfall and temperatures and rising sea levels due to climate change, reductions in water flows and sediment transport due to damming upstream, freshwater withdrawal for irrigation, excessive nutrient loads, and introduction of invasive species (Millennium Ecosystem Assessment 2005).
Cultural	Yes	Conflicts with traditional and/or cultural land-use practices that result in overharvesting and overexploitation.
Social	Yes	Social perceptions of the threats to and value of wetland ecosystem services (Boulton *et al.*, 2016).
Economic	Yes	Conflicts with productive aquaculture and agricultural practices, and national economic development plans for commodities.
Institutional	Yes	Lack of funds, resources and governance to regulate unsustainable practices and promote conservation or sustainable management.
Legal (Right to soil)	Yes	Contested land tenure, and lack of recognition of Indigenous land tenure rights (Fa *et al.*, 2020; Lovelock and Brown 2019).
Knowledge	Yes	Lack of knowledge sharing between scientists and decision-making by land owners and managers, and integrating the knowledge of Indigenous communities in wetland conservation and management. Lack of information on the trade-offs between different wetland ecosystem services (Boulton *et al.*, 2016).
Other	Yes	Clearing and drainage of inland wetlands for agricultural expansion and irrigation, and conversion of coastal wetlands for urban expansion, infrastructure development, agriculture and aquaculture (Goldberg *et al.*, 2020; Millennium Ecosystem Assessment 2005). Coastal squeeze also prevents coastal wetland expansion and conservation under climate change (Luo, 2018).

Photos of the practice

Photo 11. Converted wetlands – drained wetlands for grazing pasture in Queensland, Australia (left) and degraded mangrove during conversion to aquaculture in Myanmar (right)

Photo 12. Conservation of tidal wetlands – mangroves (Ceriops sp., left) and tidal marsh (Sarcocornia sp., right) in Queensland, Australia. SOC stocks and sequestration rates are generally higher in tidal wetlands than freshwater wetlands (Photo 13) per unit area

Photo 13. Conservation of freshwater wetland (Juncus sp. and Melaleuca quinquenervia) in Queensland, Australia

Table 35. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Management of Common Reed (Phragmites australis) in Mediterranean wetlands, Spain	Europe	Unknown	6	18
Preserving Soil Organic Carbon in Prairie Wetlands of Central North America	North America	Various	6	19
Maintenance of Marshlands in Urban Tidal Wetlands in New York City, United States	North America	100	6	31

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10. Wetland restoration (water supplementation and promoting plant growth)

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1. Description of the practice

For wetlands other than peatland, the most recommended practices in wetland restoration for C sequestration are water supplementation (such as rewetting of drained wetland) and the promotion of plant growth (Carnell *et al.*, 2018). Water supplementation can build an anaerobic environment and reduce soil C loss (Villa and Bernal, 2018). Promoting plant growth can provide more additional organic C to soil.

Water supplementation means rewetting the drained wetland or raising the water table of the degraded wetland. The most recommended water level is about ~10~30 cm above ground (Nadeau and Conway, 2015; Wang *et al.*, 2017). A water level that is too low cannot provide the anaerobic environment, while a very deep water table may restrain plant growth. Therefore, the best way is maintaining a lower water level in the growing season to promote planting growth and maintaining a deep water level in the non-growing season to sequester C (Wang *et al.*, 2017).

Promoting plant growth aims to add more organic C to soil. Wetland vegetation can be restored through natural succession via its seed bank in the soil if it is still in place (Wang *et al.*, 2015). Sowing or planting native wetland plants is also recommended (Renzi *et al.*, 2019). But there should have a diversity of plants, only having one or two species may reduce the overall biomass and biodiversity. In some wetlands (e.g. salt marsh), the soil in its natural state can limit plants growth, plants that can grow well in the barren soil should be selected for C sequestration (Doherty *et al.*, 2011). In addition, in degraded wetland with barren soil, at the beginning of wetland restoration soil improvement practices such as fertilization and tillage could be a choice to promote vegetation recovery. However, these practices should be used carefully and get adequate assessment. For example, although fertilization can promote plant growth and increase soil C in most conditions (Xu *et al.*, 2020), this practice should be used with the proper dosages for the wetland environment in order to prevent potential pollution. Furthermore, these practices, especially tillage and other soil mechanical disturbance, should be strictly prohibited after 1-2 years of restoration (Xu *et al.*, 2020).

Apart from the mentioned practices above, conventional agricultural cultivation and other human activities that lead to wetland degradation should be completely banned (Xu *et al.*, 2019). If the degradation continues, the ecosystem services provide by wetland such as water cleaning, water storage and biological habitat would lost.

2. Range of applicability

In general, wetlands degraded due to use for rainfed farming can restore better than that for other land-uses. Wetland restoration benefits soil C sequestration in boreal, subarctic, and temperate degraded wetlands, but it shows less results in term of increasing C sequestration in tropical degraded wetlands (Xu *et al.*, 2019). The restoration of seasonal wetlands and freshwater wetlands increases soil C sequestration more than that of tidal wetlands and salt wetlands, in which the restoration may have no positive effect to C sequestration (Xu *et al.*, 2019). Restoration practices should be conducted as soon as possible, as if the wetlands had been degraded more than 15 years, the restoration will be much less effective (Xu *et al.*, 2019).

3. Impact on soil organic carbon stocks

Wetland restoration has been implemented worldwide and is considered to be an effective way to recover the soil C lost as a result of wetland cultivation (Crooks *et al.*, 2011). Wetland restoration can accelerate carbon sequestration and greatly contribute to mitigating climate change if proper measures are utilized (Table 36). Most restored wetlands have an soil C content that is higher than that of cultivated wetlands but still lower than that of natural wetlands (Xu *et al.*, 2019).

Table 36. Changes in soil organic carbon stocks reported for wetland restoration

Location	Climate zone	Soil type	Baseline C stock or content	Additional C storage or content	Duration (Years)	Depth (cm)	More information	Reference
Global	NA	NA	NA	9% increase (95%CI: -3%~23%)	NA	NA	Meta-analysis; Measured data; Wetland restoration	Xu *et al.* (2019)
Eastern England	NA	NA	59 (±10 SE) tC/ha	0.65~1.04 tC/ha/yr	0-100	0-30	Model combined with field chronosequence;	Burden *et al.* (2019)

Location	Climate zone	Soil type	Baseline C stock or content	Additional C storage or content	Duration (Years)	Depth (cm)	More information	Reference
							Saltmarsh restoration after degradation from agriculture	
Nebraska, United States of America	NA	Mollisols; Fillmore, Scott, and Massie series	96.4 (±1.56 SE) tC/ha	5.1 (±2.72 SE) tC/ha	20–30	0–50	Measured; total carbon; Freshwater marsh restoration (sediment removal)	Daniel *et al.* (2017)
Yellow River Delta, China	Warm temperate continental monsoon climate	NA	3.54~3.86 g/kg	1.4~6.89 g/kg	7	0–20	Coastal marsh restored with freshwater supplement	Wang *et al.* (2011)
Yangtze River estuary, China	Subtropical monsoon climate	NA	26.44 (±2.69 SE) tC/ha	2.99 (±0.57 SE) tC/ha	6	0–10	Coastal marsh restored with agricultural abandon	Li *et al.* (2012)
Sangjiang Plain, China	Monsoon climate of medium latitudes	NA	33.41(±1.45 SE) g/kg	25.65-42.12 g/kg	3–12	0–20	Freshwater marsh restoration with agricultural abandon	Song *et al.* (2012)
Prairie Pothole region, Canada	Temperate continental climate	Mineral soil	116 tC/ha	2.5-6.1 tC/ha/yr	1–12	0–30	Freshwater mineral soil wetlands	Badiou *et al.* (2011)

4. Other benefits of the practice

4.1. Improvement of soil properties

Wetland restoration has positive effects on soil properties. One of the main benefits is the increase of biological diversity and activity, enzymatic activity and biogeochemical cycles, which are important for the restoration of wetland ecological functions (Meli *et al.*, 2014; Zedler and Kercher, 2005). Soil bulk density of restored wetlands is usually lower than that of degraded wetlands, and the restored soil can hold more water with higher proportion of porosity and organic matter (Suir *et al.*, 2019). The effects of wetland restoration on soil pH or CEC mainly depend on the original soil condition and type of wetland.

4.2 Minimization of threats to soil functions

Table 37. Soil threats

Soil threats	
Soil erosion	Plant growth can reduce the soil wind erosion, as well as water erosion. Raised water level can reduce soil wind erosion (Luo *et al.*, 2015).
Nutrient imbalance and cycles	Restored wetlands are effective at removing N, whereas P can be released for several years after restoration (Audet *et al.*, 2020; Wang *et al.*, 2015).
Soil salinization and alkalinization	Wetland restoration can decrease the pH of salt marsh with increasing soil C. The salinization could be decreased with increasing water as the salt can drift with the water (Wang *et al.*, 2017).
Soil contamination/pollution	Wetland restoration is an important path to control pollution. Plant growth can reduce the pollution, and the wetland environment can retain contamination (Su *et al.*, 2019).
Soil acidification	Wetland restoration can remediate drained acid sulfate soil with pH increase and sulfur reduction (Johnston *et al.*, 2014).
Soil biodiversity loss	The soil biodiversity loss will be halted with wetland restoration. Soil microbial diversity and fauna diversity will increase after wetland restoration (Li *et al.*, 2020; Xu *et al.*, 2017).
Soil compaction	Wetland restoration can reduce the soil bulk density and relieve the soil compaction caused by wetland degradation (Suir *et al.*, 2019).
Soil water management	Wetland restoration can increase the ability to store water and regulate runoff.

4.3 Increases in production (e.g. food/fuel/feed/timber)

The plants in wetland could be food, fibre, fuel, fodder, herb or timber depending on the vegetation type and community in the restored wetland. In addition, some wetlands can provide aquatic products such as fish, crab and shrimp, as well as increase bird population (Ducks Unlimited Canada, 2020).

4.4 Mitigation of and adaptation to climate change

Wetland restoration can transform degraded wetland to a C sink and can reduce GHG emissions. The restoration of coastal wetland in northeast China has the potential to reduce CO_2 emission from 609 mg CO_2/m/h to 278 mg CO_2/m/h (Chen *et al.*, 2018; Olsson *et al.*, 2015). For degraded wetland without drainage, the restoration may decrease the emission of CH_4, as well as N_2O, with a decrease of more than 20 percent (Gleason *et al.*, 2009).

4.5 Socio-economic benefits

Wetlands give us natural places to play, learn and explore. They are destinations for hiking, hunting, canoeing, photography and more (Ducks Unlimited Canada, 2020). Restored wetlands can be a tourism resource particularly with more wetland parks being built around the world. Typical wetland parks based on restored wetlands include Everglades National Park in United States of America, Jiuli Lake Wetland in Xuzhou in China, Tianjin Lingang Ecological Wetland Park in China and so on.

4.6 Additional benefits to the practice

Wetlands can regulate microclimate, clean water and store flood water, especially in a city or suburb, which can benefit to intensive landscapes and human settlements (McLaughlin and Cohen, 2013). Wetlands protect wildlife by providing hundreds of species with safe places to eat, sleep and raise young (Ducks Unlimited Canada, 2020).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Table 38. Soil threats

Soil threats	
Soil erosion	In certain conditions, e.g. riparian, water level increase may increase water flow and lead to water erosion (Thi and Minh, 2019).
Nutrient imbalance and cycles	Wetland restoration may lead to P limitation with increasing restored duration (Smith *et al.*, 2011).
Soil salinization and alkalinization	For some cultivated coastal salt marsh for farming, the restoration of tidal hydrology may result in soil salinization (Li *et al.*, 2012).
Soil water management	Wetland restoration may influence the regional water management due to its water demand (Hodge and McNally, 2000).

5.2 Increases in greenhouse gas emissions

The restoration of drained wetland with water level increase always results in higher CH_4 emission as the anaerobic environment can promote CH_4 emission, while upland soil often was a slight CH_4 sink (Tangen *et al.*, 2015). For cultivated wetland without drainage in the Prairie Pothole Region, the CO_2 emission may increase after restoration (Gleason *et al.*, 2009). The effects of wetland restoration on GHG are context dependent and may not be significant in certain environments (Gleason *et al.*, 2009).

5.3 Conflict with other practice(s)

Wetland restoration is in conflict with agricultural activities such as tillage, crop planting and applications of pesticide, especially the rainfed farming. In addition, other human activities, including intensive livestock grazing, infrastructure development etc., are in conflict with restoration as they may cause degradation.

5.4 Decreases in production (e.g. food/fuel/feed/timber)

Wetland restoration may decrease the production of grains or other agricultural products as the agricultural activity lead to wetland degradation should be stopped.

6. Recommendations before implementation of the practice

First, wetlands restoration may bring economic loss to landowners as the production activities lead to wetlands degradation must stopped, and they must reestablish livelihood-generating activities that are not in conflict with restoration. Thus, conflicts between landowner and restoration performer must be fully solved before restoration.

A proper plan based on the field condition is required. The restoration should take in a full consideration of the recovery effects and costs, which depend on the original land use of the wetland, recovery area, water demand, the hydrological regulation method, revegetation method and so on.

Before wetland restoration, the applicability of the restore practices should be evaluated. A pilot project can provide much useful information for the conduction of wetland restoration before the large-scale restoration.

7. Potential barriers to adoption

Table 39. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	It is hard to restore degraded wetlands to their natural status, because restoration is a long term process and the environment or climate in a large-scale may change (Moreno-Mateos *et al.*, 2012).
Cultural	Yes	Depending on the emphasis degree of public on natural environments (Zhu *et al.*, 2016), and raising awareness can be a useful strategy to foster public support for wetland restoration (Scholte *et al.*, 2016).
Social	Yes	Urban expansion, agriculture, aquaculture and industrial developments are main reasons of wetland losses and they may prevent wetland restoration (Zedler and Kercher, 2005), especially in developing countries in which the conflicts between development and ecological conservation are more prominent (Marambanyika and Beckedahl, 2016).
Economic	Yes	Wetland restoration may lead to landowners suffer economic losses due to the original producing activities on the degraded wetlands had to be stopped. The loss of land or income decrease landowners' willingness to participate in wetland restoration (Zhu *et al.*, 2016).
Institutional	Yes	Land ownership, e.g. private land ownership, in some region may be barriers for wetland restoration as wetland restoration needs the support of landowners (Marambanyika and Beckedahl, 2016; Zhu *et al.*, 2016).
Legal (Right to soil)	Yes	Incomplete environmental protection act and land act, as well as poor implementation of legislation, may affect the restoration, because the restoration project may be hard to conduct when legal support lacks (Marambanyika and Beckedahl, 2016).
Knowledge	Yes	Wetland restoration requires professional knowledge to assess the ecosystem conditions, the prefered practices, the potential effects and risk (Zhu *et al.*, 2016).
Other	Yes	Some wetlands may be difficult to restore, e.g., the seriously polluted wetland that significantly affects plant growth, and the lost coastal wetland due to sea level rise (Zedler, 2004).

Photos of the practice

Photo 14. Wetland restoration project in Qixinghe National Natural Reserve, Sanjiang Plain, China. The left photo shows the cultivated wetland before restoration (in 2014), which had been reclaimed for corn for about 30 years. The right photo shows the wetland after restoration (in 2018). Restorations consisted of reflooding by removal of ditches and tile lines. The restored wetland site was dominated by Phragmites australis and Carex species

Photo 15. Wetland restoration project in Niuxintaobao National Wetland Park, Songnen Plain, China. The left photo shows the degraded wetland before restoration (in 2010). The right photo shows the wetland after restoration (in 2013). Restoration practices included water supplement in combined with the promotion of plant growth and reproduction, e.g., raking and plant (or rhizome) transplantation. The restored wetland site was dominated by Phragmites australis

Table 40. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Preserving Soil Organic Carbon in Prairie Wetlands of Central North America	North America	Various	6	19
Maintenance of Marshlands in Urban Tidal Wetlands in New York City, United States	North America	100	6	31

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11. Conservation of pristine peatlands and avoiding drainage of peatlands

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1. Description of the practice

Conservation of peatlands, especially the avoidance of draining areas, means to refrain from all actions that disrupt natural ecosystem functions. Activities like drainage, fertilizing, tree cutting, slash and burn, infrastructure development (e.g. construction of roads, trails), and other factors may cause the lowering of natural water table levels, leading to SOC loss by aerobic degradation and dissolution of SOC in water (DOC). Degradation of the peat layer results in negative effects on land, causing subsidence (i.e. lowering of the peat surface) GHG emissions, and decreases community resilience to climate change (See Hotspots: Peatlands).

For conservation it is essential to map peatlands, develop management and monitoring plans, and ensure that local populations have access to livelihood sources that do not threaten peatlands. Knowing the location, extent, and carbon storage of peat deposits, as well as understanding the ecosystem services they provide (see sections 3 and 4 below), are necessary steps towards establishing suitable conservation strategies. Conservation is often a very cost-effective land-use management option (Joosten, Tapio-Biström and Tol, 2012) that still allows a range of non-invasive, livelihood-development activities (e.g. collection of berries, honey production, fibre, feed, small-scale livestock rearing, or development of non-degrading ecotourism).

2. Range of applicability

Pristine peatlands occur in more than 160 countries (see Hotspots: Peatlands) and currently many initiatives are working to better map and assess peatlands (FAO, 2020). An increasing number of countries are interested in knowing the actual location and extent of their peatlands, and to develop sustainable, drainage-free livelihood options. The assessment of peatlands is the starting point for avoiding their drainage as their conservation can be better realized. Conservation as a management practice should be applied to all pristine peatlands globally, in particular and most urgently in zones with high pressure for peatland drainage and vegetation removal.

Maintaining the hydrological conditions and the nutrient status of a peatland ought to be the objective to avoid SOC losses. In order to achieve this, it is critical to maintain typical biotic communities, natural succession, and linkages between peatlands within a landscape. On the other hand, drainage and nutrient discharge should be avoided. However, peatland-specific analysis of the situation and threats can better inform the prioritization of conservation activities. In some cases, ecotourism and non-timber product extraction can be compatible with conservation and restoration purposes, while contributing to the creation of incentives to promote conservation.

3. Impact on soil organic carbon stocks

Peatland conservation maintains SOC in the peat, avoiding anthropogenic GHG emissions, and securing carbon sequestration due to long-term peat accumulation. Globally the area of near-pristine peatlands – more than three million km^2 – are estimated to sequester 0.37 Gt CO_2 per year in the soil. Similarly, degrading peatlands worldwide emit estimated 1.3 Gt CO_2 per year, equivalent to at least 5 percent of global anthropogenic CO_2 emissions (IPCC, 2014). An estimated 336–644 Gt SOC can be potentially conserved in the peat soil if peatlands are conserved, and drainage avoided. Peat usually accumulates slowly, at the rate of about a millimetre per year, which makes peatlands slow carbon sinks. Also, their potential for SOC maintenance lies in the avoided carbon emissions of the carbon stored, and not on the yearly sequestration rate, which is only considerable over long temporal scales (Soares *et al.*, 2015; Donato *et al.*, 2011).

4. Other benefits of the practice

4.1. Improvement of soil properties

Avoiding drainage of pristine peatlands contributes to land stabilization, nutrient balance, soil biodiversity and moisture regimes, allowing the peat to continuously accumulate SOC.

4.2 Minimization of threats to soil functions

Table 41. Soil threats

Soil threats	
Soil erosion	Avoiding drainage, conservation of the peat layer and pristine vegetation cover is crucial for avoiding peat erosion. Pristine peatlands regulate water run-off, whereas dry and bare peat is exposed to oxidation and is easily eroded, even in flat areas, causing quick surface lowering (land loss or subsidence), water and carbon losses (Joosten, 2015).
Nutrient imbalance and cycles	Intact peatlands retain nutrients and help purify surface waters (Succow and Joosten, 2001).
Soil salinization and alkalinization	Conserved coastal peat swamps act as a buffer between salt- and freshwater systems, preventing saline intrusion into coastal lands (Silvius, Joosten and Opdam, 2008) and the eventual loss of adjacent land (Joosten, 2015).
Soil contamination/pollution	Intact peatlands retain pollutants and help filtering and purifying surface water (Wichtmann, Schröder, and Joosten, 2016).
Soil acidification	Depending on peatland type: ombrotrophic peatlands are already acidic (Succow and Joosten, 2001).
Soil biodiversity loss	Maintaining the pH, SOC (Mandic-Mulec *et al.*, 2014), plant cover and specific physicochemical characteristics of peat (Opelt *et al.*, 2007) might be determinant of the composition of a microbial community.
Soil compaction	Compaction caused by soil decomposition and mineralization is avoided when peatlands are wet.
Soil water management	Intact peatlands have unique and vulnerable soil moisture regimes (Dommain, Couwenberg and Joosten, 2010). Maintenance of the natural regime prevents irreversible drying – when peat soils are exposed to prolonged, intensive drainage, the peat becomes hydrophobic, rendering it very difficult to rewet and restore given that the peat soil diminishes its capacity to retain water. Peats reaching this stage lose water holding capacity (up to 40–75 percent) (Andriesse, 1988) compared to wet peat soil.

4.3 Increases in production (e.g. food/fuel/feed/timber)

Although intensive productive activities should be prevented, some subsistence and income-generating activities related, for example, to extraction of non-timber products, like honey, and other activities, like fishing, and hunting, can be compatible with conservation activities. In fact, conserved peatlands ensure, in many cases, local food security and can regulate water supply for irrigation of downstream productive sites outside the peatlands.

4.4 Mitigation of and adaptation to climate change

Peatland conservation and avoiding drainage contributes directly to climate change mitigation as well as increasing adaptive capacity. Preventing drying caused by drainage or other activities avoids new GHG emissions from degradation and from fires. It also allows conserved peatlands to protect the coastal areas from extreme weather events, while regulating water supply and reducing fire risks in the catchment (see Hotspot "Peatlands" in volume 2 of this manual).

4.5 Socio-economic benefits

Peatland conservation and avoiding drainage are the most cost-effective ways to manage peatlands, and lower the social and climate cost of peatland degradation (Joosten *et al.,* 2012). These activities secure the provision of ecosystem services that support economies and human well-being (see Hotspot "Peatlands" in volume 2 of this manual).

5. Potential drawbacks to the practice

5.1 Increases in greenhouse gas emissions

None are known. Pristine peatlands are climate neutral, and in most cases are slow carbon sinks (see above section 3 on impact on SOC sequestration). Conservation would also make peatlands less vulnerable to drying and other climatic changes, helping to reduce non-anthropogenic GHGs (Swindles *et al.*, 2019).

5.2 Conflict with other practice(s)

Avoiding drainage of peatlands and promoting their conservation reduces options to establish other practices on peatlands, except in some cases ecotourism and extraction of non-timber products on a small scale (Crump, 2017). However, this is justified by the long-term benefits of conservation and avoided drainage in the landscape, ecosystem services provision, and reduction of environmental risk to communities.

5.3 Decreases in production (e.g. food/fuel/feed/timber)

In the short term, preserving pristine peatlands implies reduced availability of peatlands for degradation and conversion into drainage-based production systems (e.g. oil palm, timber plantations). Intensive monoculture agricultural practices result in long-term land loss and degradation, disruption of ecosystem services, increased emissions and loss of adaptive capacity of the landscape, including surrounding communities. Considering that peatlands represent only three percent of the earth's surface, refraining from using these lands as productive areas will secure long-term ecosystem services and a positive long-term impact on the regions where they are located.

6. Recommendations before implementation of the practice

Peatland location and boundaries should be clearly demarcated in the land-use planning maps:

- Each peatland should be considered as one hydrological unit, meaning that draining the water in a part of a peatland, will affect the groundwater table in the whole peatland. Therefore, management measures should address the entire peatland entity to reap maximum benefits in terms of biodiversity or carbon conservation. The maintenance of high water tables should be prioritized to conserve the SOC in peat soils, noting that changes in hydrology of the landscape outside the peatland area may affect peatland hydrology.
- Conservation of the peatland vegetation cover is important to secure peat formation and therefore long-term carbon storage.
- Due to their relevance for ecosystem services, hydrology, and the adaptive capacity of surrounding communities and landscapes, peatlands should be excluded from drainage-based activities, i.e. set aside as conservation areas.

7. Potential barriers to adoption

Table 42. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Potential changes in the rainfall due to climatic changes and land cover changes, such as deforestation.
Cultural	Yes	Conflicts with traditional and cultural land-use practices that include drainage for production/extraction.
Social	Yes	Social acceptance of prohibition/limitation of activities on the peatlands; social perceptions of different conservation interventions (Harrison *et al.*, 2019).
Economic	Yes	Conflict with activities that generate short-term revenue but cause important losses in the longer term (Harrison *et al.*, 2019).
Institutional	Yes	Lack of funds and incentives to promote conservation over drainage-based alternatives (Harrison *et al.*, 2019).
Legal (Right to soil)	Yes	Frequent lack of clarity regarding wetlands' legal status and responsibility for different areas and activities, conflicting/unclear laws and ineffective law enforcement (Harrison *et al.*, 2019).
Knowledge	Yes	Lack of data and knowledge to assess conservation potential and fire impacts, the effect of conservation efforts, specific dynamics and characteristics of each peatland type and location (Harrison *et al.*, 2019).
Other: Drainage-based livelihoods	Yes	Drainage-based peatland management such as grazing, cropping, plantations and forestry activities often prevent conservation and the maintenance of the natural ground water level (FAO, 2012).

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12. Restoration of peatlands

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1. Description of the practice

The restoration of degraded peatlands consists of two equally important steps: i) rewetting of the peatland and ii) revegetation with peat-generating species of ideally economic viability, if needed (FAO, 2014; FAO, 2020).

Rewetting consists of raising the mean annual groundwater table of the whole hydrological unit of a drained peatland, ideally back to surface level (Jauhiainen, Page and Vasander, 2016; Tannenberger *et al.*, 2020). This way, the oxygen-deficient soil conditions are re-established and further oxidation of the peat is minimized, which also reduces subsidence[2] (Jauhiainen, Page and Vasander, 2016) and the loss of SOC as greenhouse gas emissions (Günther *et al.*, 2020; Wakhid *et al.*, 2017) and as organic carbon dissolved in water (DOC). Moreover, with restoration, the risk of peat fires is minimized. For effective results, and following the greenhouse gas emission reduction method of the Wetland Supplement (IPCC, 2014), the whole peatland hydrological unit needs to be rewetted. Rewetting is done by:

- blocking drainage canals and ditches (e.g. with peat collected at site);
- raising overflow heights of weirs, "dams" or "canal blockings" and sluices;
- establishing and allowing obstructions in water courses (trees, rocks, vegetation growth, beaver dams);
- removing subsurface drainage pipes;
- reducing evapotranspiration from tree growth in the peatland (only in originally treeless peatlands);

[2] Subsidence is the lowering of the surface caused by peat compaction and oxidation (see e.g. FAO, 2020).

- establishing hydrological buffer zones with higher water levels; and/ or
- in low-lying coastal peatland areas, such as 'polders'3 in the Netherlands, which are protected by dikes or other artificial barriers and drained through active pumping, the pumping must be reduced to establish close-to-surface water tables (Couwenberg, 2018).

Revegetation is the re-establishment of vegetation that is adapted to wet soil conditions. Revegetation can happen through natural regeneration or active sowing or planting of the site´s native wetland plants. (FAO, 2014; FAO, 2020). Rewetting addresses the negative effects of drainage, while revegetation helps regulate water balance and is necessary for the peatland to recover, and potentially become a carbon sink again.

2. Range of applicability

The reasons for peatland restoration are growing and include the achievement of peatland biodiversity and/or recuperation of water regulation and other ecosystem functions and services, such as avoidance of greenhouse gas emissions and other carbon losses (FAO, 2014). The potential and need of restoring peat soils for SOC preservation has been developed in degraded boreal, subarctic, temperate, as well as tropical peatlands (Joosten, Tanneberger and Moen, 2017; Giesen and Nirmala, 2018).

3. Impact on soil organic carbon stocks

Peat soils consist of partly preserved dead plant material that built up often over thousands of years at an average rate of 0.5–1 mm per year in temperate and boreal regions, and up to 2.5 mm per year in tropical zones (Page *et al.*, 2010). Vegetation[4] cover of native peatland species is crucial to re-establish SOC and restoring peat accumulation over time (Jauhiainen, Page and Vasander, 2016). That said, carbon sequestration in peat soils is a slow process that happens over centuries. When the IPCC Wetlands Supplement was published (2014) there was a lack of data on soil carbon sequestration rates in years or decades following peatland rewetting (IPCC, 2014). However, a recent study from temperate climates illustrated that the carbon sink function of a restored peatlands can be approximately 1 tonne C per ha per year, depending on the vegetation and water regime (Swenson *et al.*, 2019).

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[3] A polder is a low-lying tract of land that forms an artificial hydrological entity, enclosed by embankments known as dikes. In polders, ground-water tables need to be kept artificially low through permanent pumping.

[4] To note: revegetation (including replanting) is necessary only when natural plant regeneration is not possible due to e.g. burning of the soil seed bank or over 5 km distance from peatland areas with endemic wetland vegetation (see e.g. FAO, 2020).

4. Other benefits of the practice

4.1. Minimization of threats to soil functions

Peatland restoration positively affects soil characteristics. Water run-off of restored peatlands can be reduced by 25 percent compared to a peatland with no restoration efforts (Shantz and Price, 2006) due to increased water storage capacities and despite increased hydraulic conductivity of newly regrown soil vegetation (Wallage and Holden, 2011). Reduced bulk density increases water and nutrient movement, reduces soil moisture fluctuations, and prevents soil erosion (Wichtmann *et al.*, 2016). However, the establishment of pre-drainage hydrological and soil conditions necessarily takes place over a similar temporal scale as peat formation itself and, in some cases, restoration efforts will never achieve the pristine state found prior to drainage due to changed site conditions.

4.2 Minimization of threats to soil functions

Table 43. Soil threats

Soil threats	
Soil erosion	Erosion by surface water run-off is minimized through canal blocking and managed grazing. Wind erosion from bare peat is minimized by permanent vegetation cover (Wichtmann, Schröder and Joosten, 2016).
Nutrient imbalance and cycles	Restored peatlands can regain nutrient retention and water purification functions (Succow and Joosten, 2001).
Soil salinization and alkalinization	In coastal regions: peatland restoration stops subsidence, and could prevent saltwater intrusion and/or flooding (Hooijer *et al.*, 2015).
Soil contamination/pollution	Peatlands can act as water filtering systems and may retain nutrients and pollutants.
Soil acidification	Depending on peatland type: ombrotrophic peatlands already are acidic (Succow and Joosten, 2001). Rewetting of acid sulphate peat soil reduces the risk of acidification (Wösten *et al.*, 2006).
Soil biodiversity loss	Loss of non-peatland soil organisms (earthworms, moles, rodents) is a positive sign of successful peatland restoration.
Soil compaction	Compaction caused by soil decomposition and mineralization is halted, wetland plant establishment recreates original topsoil conditions with low bulk density, high hydraulic conductivity, etc. (Wichtmann, Schröder and Joosten, 2016).

Soil threats	
Soil water management	Restored peatlands with wetland vegetation self-regulate water run-off and provide a hydrological buffer function (Dommain, Couwenberg and Joosten, 2010; Wallage and Holden, 2011).

4.3 Increases in production (e.g. food/fuel/feed/timber)

Rewetting establishes conditions for the growth of wetland plants that can be used for a range of purposes: energy production, fibre for artisanal and construction materials, fodder, horticultural substrates (Wichtmann, Schröder and Joosten, 2016), and also for food (Giesen and Nirmala, 2018). Productivity depends on species and site characteristics but can be high compared to conventional drainage-based agriculture (Zerbe *et al.*, 2013).

4.4 Mitigation of and adaptation to climate change

Rewetting of peatlands minimizes CO_2 emissions immediately (Carlson *et al.*, 2015; Couwenberg, 2018). Methane emissions may increase in the first years after rewetting (Günther *et al.*, 2020). Models of global warming show the potential trade-offs between the CO_2 and N_2O (long-lived GHGs) emissions of drained peatlands versus increased CH_4 (a short-lived GHG) emissions of rewetted peatlands and suggest that rewetting would greatly reduce the global warming potential of these ecosystems in decades to come (Günther *et al.*, 2020; Wilson *et al.*, 2016; Figure 6). Thus, rewetting all or half of global peatlands now would decrease global heating more than rewetting later, or never rewetting peatlands (Günther *et al.*, 2020).

If bringing the water table back to surface level entirely is not practicable (e.g. due to economic constraints) partial rewetting, as applied on drained peatlands in Indonesia and Malaysia, reduces SOC losses and fire risk, and thus, can partially help but not solve issues related to subsidence, fire and degradation (Carlson, Goodman and May-Tobin, 2015; Wakhid *et al.*, 2017, see figure 1 in factsheet n°13 “Paludiculture”, this volume). However, blocking only a part of drainage system only slows down soil degradation, and should be regarded as an intermediary step towards full rewetting.

Additionally, drained soils are at increased risk of catastrophic peat fire and losses of carbon stores (Turetsky *et al.*, 2015). Peatland restoration thus decreases the risk of dramatic negative impacts on human livelihoods as well as on the global carbon cycle from peat fires (Page and Hooijer, 2016).

4.5 Socio-economic benefits

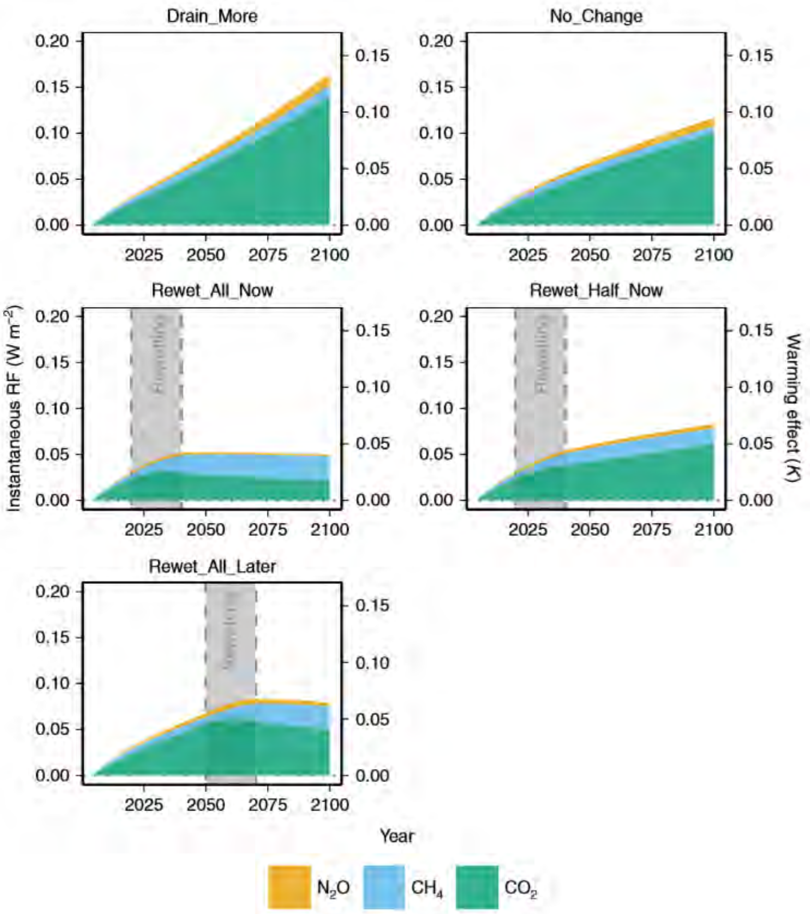

Figure 6. Effects of greenhouse gas emissions from peatlands following global rewetting scenarios, and related radiative forcing (RF) and global warming effect. The gray area shows the period of rewetting. (Nature, Günther *et al.*, 2020)

Restoration halts peatland subsidence – lowering of the peat surface – and related loss of coastal and riverine areas and helps to avoid peatlands becoming undrainable over time (Hooijer *et al.*, 2015), therefore also protecting lives, infrastructure and livelihoods. Negative health and economic impacts of fire and haze are prevented with restoration measures. Restored peatlands can provide sustainable livelihoods with a range of options for wet biomass production and other economic activities (see factsheet n° 13 on paludiculture, this volume).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

None are known.

5.2 Increases in greenhouse gas emissions

Whilst rewetting of peatlands immediately reduces CO_2 emissions, the CH_4 emissions can increase depending on the water level for a few years (Wilson *et al.*, 2016; Jauhiainen, Page and Vasander, 2016). As a consequence, restored peatlands should not be flooded, but it is recommended to manage and maintain the medium annual ground water table close to or at the surface (Wichtmann, Schröder and Joosten, 2016). Furthermore, methane emissions are not as important as CO_2 emissions for long term climate impact, as methane breaks down quicker in the atmosphere compared to long-lived CO_2 (Günther *et al.*, 2020; Figure 6). Wetland plants (e.g. reed or cattail) in the temperate zone lead to decreased CH_4 emissions compared to rewetted bare peat soils (Swenson *et al.*, 2019; Vroom *et al.*, 2018), highlighting the importance of revegetation for preserving SOC and reducing the global warming potential.

5.3 Conflict with other practice(s)

Invasive activities (e.g. intensive agriculture or plantations), which would include tillage, drainage, annual crops, or high livestock density with a significant trampling impact, cannot be aligned with restoration of peatlands.

5.4 Decreases in production (e.g. food/fuel/feed/timber)

Restoration of peatlands is not compatible with conventional drainage-based land-use. There are established traditional peatland uses for various biomass extraction, and similarly alternative productive systems based on biomass production in wet peatland condition (Paludiculture, see factsheet n°13 of this volume) are being studied through pilot projects (see case studies).

6. Recommendations before implementation of the practice

- Peatland restoration should involve and consult all concerned stakeholders, e.g. farmers, landowners, users of waterways, and neighbouring communities.
- It is necessary to properly plan the restoration activity suited to the local characteristics of sites, including hydrological balance. Proper preparation requires also a verified map of the peat extent and the whole hydrological unit, where the peatland is embedded.
- Policy makers should explore potential incentives and pricing instruments (i.e. subsidies) that compensate for lost drainage-based productivity. Some schemes may reward practices that transition from conventional peatland use to peatland restoration.

7. Potential barriers to adoption

Table 44. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Restoration may not lead to original pre-drainage peatland conditions, depending on the duration and intensity of peatland degradation.
Cultural	Yes	Peatlands are wetlands that often are difficult to access, and sometimes host insects that may spread malaria, and may be perceived as unpleasant or threatening environments.
Social	Yes	Resistance to restoration is often present as it requires abandonment of conventional drainage-based land-uses.
Economic	Yes	Peatland restoration may not be compatible with drainage-based production practices, resulting in short-term economic losses if no policies are in place to incentivize restoration (see chapter 4.1.3 Paludiculture) and reduced economical viability of the land utilization.
Institutional	Yes	Drainage-based utilization is institutionalized in many countries as the most familiar use, which sometimes also benefits of incentives.
Legal (Right to soil)	Yes	Landowners need to agree to change hydrological conditions of the area, which may stop conventional land-use and thus reduce the lands economical potential.
Knowledge	Yes	Restoration requires peatland-specific knowledge, that is lacking in many regions of the world.

Note: The authors of this chapter acknowledge the contributions from Beth A. Middleton, Eric Ward and Lorenzo Menichetti in section 4.4 on climate change mitigation and adaptation.

Table 45. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Biomass from reeds as a substitute for peat in energy production in Lida region, Grodno oblast, Belarus	Eurasia	Unknown	6	20
Sphagnum farming for replacing peat as horticultural growing media, Lower Saxony, Germany	Europe	10	6	21

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13. Paludiculture

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1. Description of the practice

Paludiculture produces biomass from wet or rewetted peatlands under conditions that maintain the peat integrity, facilitating peat accumulation and ensuring the provision of peatland ecosystem services (FAO, 2014). Spontaneous vegetation or artificially establishing crops on rewetted sites (see chapter 4.1.2 on Peatland Restoration) require adapted machinery for harvesting. Aside from producing traditional agricultural products such as food, feed, fiber and fuel, the biomass can be used as a raw material for industrial biochemistry, for production of construction materials, substrate for horticulture, high-quality liquid or gaseous biofuels, extraction and synthesizing of pharmaceuticals and cosmetics, among other purposes.

Under paludiculture, water tables need to be established at or close to the surface throughout the year (Jauhiainen *et al.,* 2016; Tanneberger *et al.*, 2020; Wichtmann, Schröder and Joosten, 2016). Paludiculture is currently considered the most efficient solution to balance restored peatland use for livelihood purposes and to increase or to allow SOC maintenance and sequestration in the long term.

When considering wet peatland use, management methods must be well adapted to site-specific conditions. Peatland type, natural vegetation, soil conditions, such as nutrient availability and acidity and hydrology are some factors that need to be considered (Wichtmann, Schröder and Joosten,2016). In temperate zones, herbaceous *Sphagnum* sp, *Drosera* sp., *Typha latifolia*, *Phragmites australis*, *Alnus glutinosa*, are some of the potential paludiculture species (Abel, Couwenberg and Joosten, 2012). In Southeast Asia, the palm *Metroxylon sagu* and trees *Dyera polyphylla, Shorea* ssp., are amongst a whole range of known species used for food, timber and other non-timber products (Van der Meer and Karyanto, 2013; Giesen and Nirmala, 2018). Species that are not able to cope with permanent wet soil conditions with high water tables, cannot be considered for paludiculture purposes (Giesen and Nirmala, 2018; FAO, 2014; Tata, 2019).

2. Range of applicability

The need for a shift from drainage to wet-based peatland utilization is being promoted internationally (e.g. Crump, 2017; FAO, 2014), and it is urgent due the recognition of the severe impacts of drainage-based peatland use on climate, biodiversity loss (Dohong, Aziz and Dargusch, 2017), sustainable livelihoods, and other important ecosystem services they provide (see chapter 4.1.1 and 4.1.2). Paludiculture approaches and techniques with different plant species, and for a range of purposes are being explored in temperate regions (Wichtmann, Schröder and Joosten,2016) as well as in tropical Southeast Asia (Giesen and Nirmala, 2018).

3. Impact on soil organic carbon stocks

Paludiculture can reduce the loss of SOC as GHG emissions (**Figure 7**) and/or dissolved organic carbon (DOC), and in the long-term the carbon sequestration function will potentially be restored. The magnitude of the effect of paludiculture[5] practices on the carbon sequestration potential and the temporal scale needed are not well understood and further long-term research is needed.

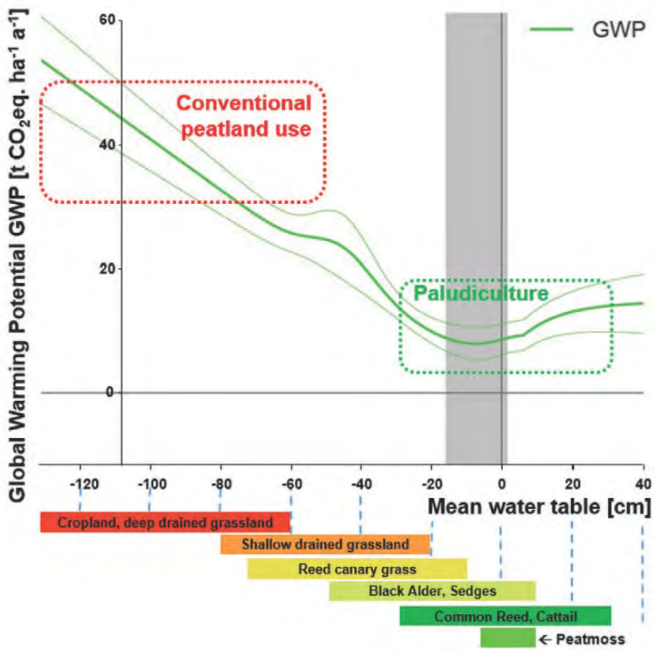

Figure 7. Relationship between Global Warming Potential of greenhouse gas emissions, mean annual ground water table and land-use practices for temperate peatlands. Adapted from Tanneberger *et al.*, 2020

[5] Also see factsheet n° 13 on Paludicutlture, this volume.

4. Other benefits of the practice

4.1. Improvement of soil properties

See factsheet n° 12 on Peatland Restoration, this volume.

4.2 Minimization of threats to soil functions

Table 46. Soil threats

Soil threats	
Soil erosion	Surface lowering (land loss or subsidence) is prevented by avoiding drainage (Joosten, 2015). Wind erosion which is a severe problem of ploughed, dry organic soils is eliminated (Succow and Joosten, 2001).
Nutrient imbalance and cycles	Intact peatlands retain nutrients and help purify surface waters (Succow and Joosten, 2001).
Soil salinization and alkalization	Coastal peat swamps act as a buffer between salt- and freshwater systems, preventing saline intrusion into coastal lands (Silvius, Joosten and Opdam, 2008).
Soil contamination / pollution	Intact peatlands retain pollutants and help purify surface water.
Soil acidification	Contingent on peatland type: ombrotrophic peatland are already acidic (Succow and Joosten, 2001).
Soil compaction	Compaction caused by soil decomposition and mineralization is halted (Schröder *et al.*, 2015).
Soil water management	Intact peatlands with wetland vegetation self-regulate water run-off and release, thus having a buffer function (Dommain, Couwenberg and Joosten, 2010).

4.3 Increases in production (e.g. food/fuel/feed/timber)

Biomass production on rewetted peatlands can be equal or even exceeding that of drainage-based agriculture and other biomass production (Wichtmann, Schröder and Joosten, 2016). Bioenergy production from paludiculture can replace fossil fuels (Wichtmann *et al.,* 2014; Wichtmann *et al.,* 2019).

4.4 Mitigation of and adaptation to climate change

Paludiculture is the only climate-neutral, SOC preserving land-use option for peatland use. Transforming drainage-based peatland-use such as oil palm plantations into paludiculture (e.g. sago or illipe nut) cultivation can reduce anthropic greenhouse gas emissions between 70 to 117 tonnes CO_2eq per ha per year with CO_2 responsible for approximately 60 percent and N_2O responsible for approximately 40 percent of the emissions (Cooper *et al.*, 2020; IPCC, 2014). In Central Europe, reported anthropic GHG emission reduction potential is 20–60 tonnes CO_2eq per hectare per year (Couwenberg *et al.*, 2011; Tanneberger *et al.*, 2020; Figure 7). Furthermore, paludiculture has the potential to reduce CH_4 emissions – which increase after peatland rewetting – through improved topsoil properties by wetland plant root chemistry (Vroom *et al.*, 2018). In the tropics however, methane dynamics in drained and rewetted areas are yet poorly understood (Sakabe *et al.*, 2018; Deshmukh *et al.*, 2020). If biomass from paludiculture is used to replace fossil fuel energy production, the climate change mitigation potential is further increased (Wichtmann, Couwenberg and Kowatsch, 2009). Climate change adaptation benefits include a net cooling effect on regional climate by keeping water in the landscape, thus playing an important role in water availability for communities, in arid regions in particular. Furthermore, benefits for disaster risk reduction include inhibited future flooding risks by preventing subsidence (Hooijer *et al.*, 2015) and reducing fire risks (Page and Hooijer, 2016).

4.5 Socio-economic benefits

Paludiculture provides long-term sustainable livelihoods for people, by halting subsidence and thus avoiding loss of riverine and coastal land through subsequent flooding (see "Hotspots: Peatlands"), by preventing hazardous environments as well as substantial economic losses due to fires and haze (Page and Hooijer, 2016). Paludiculture also has the potential to offer viable income sources from production on wet peatlands, particularly when viable value chains are established (Giesen and Nirmala, 2018; Wichtmann, Schröder and Joosten, 2016; Uda, 2019).

4.6 Additional benefits to the practice

Further benefits include nutrient retention, water regulation and purification (FAO, 2014; Walton *et al.*, 2020), and maintenance and improvement of biodiversity (Wichtmann, Schröder and Joosten, 2016).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

None are known.

5.2 Increases in greenhouse gas emissions

See factsheet n° 12 of this volume and Peatlands chapter of volume 2.

5.3 Conflict with other practice(s)

Raising the water table can lead to land use conflicts with conventional land-use practices such as grasslands in Europe or oil palm plantations in Southeast Asia, causing negative short-term economic effects (Wichtmann, Schröder and Joosten, 2016). Furthermore, land-use conflicts between paludiculture and conservation may occur after peatland restoration. Adaptive guidelines and spatial land-use planning is required to guide and manage transition from traditional drainage-based land-use towards sustainable paludiculture on peatlands (Joosten *et al.*, 2015; Tanneberger *et al.*, 2018).

5.4 Decreases in production (e.g. food/fuel/feed/timber)

At present, paludiculture is not as economically viable as drainage-based agriculture. Moreover, it cannot compete with food production in terms of productivity (Leifeld and Menichetti, 2018). That said, conventional cost benefit analysis needs to account for the long-term, highly negative impacts of drainage-based production systems.

5.5 Other conflicts

In Central Kalimantan, Indonesia, the establishment of paludiculture practices has encountered several challenges, such as low local community participation (Syahaza, Bakce and Irianti, 2019). The concept of wet agriculture or forestry in peatlands is often a novel concept to local communities and requires guidance and education for general acceptance and successful implementation. There are, however, also traditional, wet production systems that are non-disruptive to peatland ecosystems, such as traditional fishing using 'beje', or collection of rattan - both within the limits of natural regeneration.

6. Recommendations before implementation of the practice

- Study available cases of paludiculture (e.g. FAO, 2014; Giesen and Nirmala, 2018; Wichtmann, Schröder and Joosten,2016) and connect with the community or practitioners,
- Prepare the land for wet cultivation, including restoration of high water tables,

- Assessing suitable plant species in participatory processes with local communities, according to climate, peat type, hydrology and regional market conditions,
- Establish or strengthen value chains and develop a market for paludiculture products,
- Establish (plant) nurseries to ensure sufficient plant provision,
- Depending on the production system, investment into new machinery suitable for soft and wet peat soils may be needed (Schröder *et al.*, 2015); and
- Promote the simplification of spatial planning processes to speed up rewetting procedures and the establishment of paludiculture.

7. Potential barriers to adoption

Table 47. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Access to rewetted peat soils is reduced and requires adapted machinery (Wichtmann, Schröder and Joosten, 2016).
Cultural	Yes	Practitioners and consumers' unfamiliarity with wetland products.
Social	Yes	Paludiculture may reduce the number of workers per ha (e.g. if biomass for energy is produced instead of meat) but this could also be increased if more labour-intensive management of paludiculture sites and/or products (e.g. sago) requires new processing chains, which results in increased added value.
Economic	Yes	Limited paludiculture establishment, market development and participation in Payment for Ecosystem Services mechanisms reduces competitiveness against drainage-based production systems.
Institutional	Yes	Slow planning processes may hinder the implementation (Peters *et al.*, 2020).
Legal (Right to soil)	Yes	In the EU, wetland plants are not yet considered agricultural crops - nor are there agricultural subsidies available for paludiculture (Peters *et al.*, 2020). Political frameworks that support the development of paludiculture are not known from any part of the world.
Knowledge	Yes	Worldwide, productive use of wet peatlands is a nascent field of peatland management. Hence, knowledge about paludiculture and its potential for ecosystem services is still limited. Finding profitable commodities suitable for rewetted peatland cultivation that can compete with conventional commodities is the research challenge. Salient solutions would allow for more voluntary implementation of paludiculture systems.

Photo of the practice

Photo 16. Peatland Restoration and reforestation site in Sumatra Kayuagung, South Sumatra

Table 48. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Biomass from reeds as a substitute for peat in energy production in Lida region, Grodno oblast, Belarus	Eurasia	Unknown	6	20
Sphagnum farming for replacing peat as horticultural growing media, Lower Saxony, Germany	Europe	10	6	21

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14. Restoration of mangrove forest

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1. Description of the practice

Mangrove forests occur worldwide along tropical coasts in inundated soils where primary production and anaerobic conditions contribute to the building of soil organic matter (Also see Mangroves Hot-spot, Volume 2). Note that peat may accumulate in certain coastal mangrove (Middleton and McKee, 2001). The actual amount of soil organic matter stored in these wetlands depends on the balance between primary production and decomposition processes (Middleton and McKee, 2001; Kolka *et al.*, 2018; Middleton, 2020). The restoration of mangroves can increase carbon stocks both in soil and aboveground biomass (Wickland *et al.*, 2013; Chimner *et al.*, 2017; Friess *et al.*, 2019). While tropical inland peatland forests may have higher carbon sequestration rates than mangrove swamps, methane (CH_4) emissions are generally lower in mangrove swamp, so that these wetlands have greater carbon sequestration potential (Kolka *et al.*, 2018; Al-Haj and Fulweiler, 2020). Mangroves and forested boreal and temperate peatlands tend to store more carbon than non-forested peatlands (Kolka *et al.*, 2018).

Mangrove forest regeneration will occur naturally if the environment suits their life history requirements and propagules are deposited at appropriate elevations (Lewis, 1994; Middleton, 1999). Furthermore, the overall functional equivalence of restored wetlands to natural wetlands is a matter of debate (Kolka *et al.*, 2018).

In mangroves, if the soils of the restored mangrove swamps were not originally damaged, total carbon stocks of restored and pristine forests are similar. CO_2 sequestration is substantial after restoration (Sharma *et al.*, 2020). Restored mangrove swamps hold an amount of carbon equivalent to undisturbed ones after 25-30 years (e.g. Cambodia; Sharma *et al.*, 2020) so that restoration programs have a high potential for carbon sequestration (Hutchison *et al.*, 2014; Chimner *et al.*, 2017). The success of mangrove restoration also depends on the environmental context, e.g. former aquaculture ponds often have higher success rates than seafront plantings (Friess *et al.*, 2019).

2. Range of applicability

Mangroves are restricted primarily to the tropics, with poleward ranges set by freezing-induced damage and mortality (Osland *et al.*, 2017).

3. Impact on soil organic carbon stocks

Table 49. Changes in soil organic carbon stocks reported for restoration of mangrove forests

Location	Context	Cseq/Additional C storage	More information	Reference
Mangrove; northern Cambodia	Restored ecosystem C storage after 25 years	949.4 ± 64.4 tC/ha	Restored and pristine carbon stocks similar	Sharma *et al.* (2020)
Worldwide	Mangrove Restoration Potential Map	0.069 Gt carbon in above ground biomass; 0.296 Gt carbon in top m of soil (over an area of 8 120 km^2 of restorable mangroves worldwide)	Potential increase in carbon after restoration	Worthington and Spalding (2018)

4. Other benefits of the practice

4.1. Improvement of soil properties

Peat accretion via organic matter from roots (Middleton and McKee, 2001) is an important component of maintaining coastal elevation with sea level rise (McKee *et al.*, 2007). Peat elevation is most likely to be stable if the vegetation is healthy, and collapse can occur if vegetation is damaged (Chambers, Steinmuller and Breithaupt, 2019).

4.2 Minimization of threats to soil functions

Table 50. Soil threats

Soil threats	
Soil erosion	Restored mangroves protect from coastal erosion from tsunamis and tropical storms (Alongi, 2012; Hutchison *et al.*, 2014).
Soil salinization and alkalinization	Restored mangroves in high salinity environments have decreased CO_2 flux and root productivity (Troxler *et al.*, 2015).
Soil contamination / pollution	Restored mangroves purify water (Hutchison *et al.*, 2014).
Soil acidification	Restoration reduces acid sulfate soils in aquaculture ponds (Alongi, 2002).
Soil biodiversity loss	Plant-soil-microbial relationships are understudied (Alongi, 2002).
Soil water management	Raising water table for restoration will decrease GHG loss by slowing decomposition of soil organic matter (Kolka *et al.*, 2018).

4.3 Increases in production (e.g. food/fuel/feed/timber)

Mangrove restoration increase production of materials from natural forests, as opposed to conventional agriculture and silviculture (Middleton, 1999). Mangroves are invaluable for supporting both primary and secondary production related to biodiversity and wildlife (Alongi, 2002).

4.4 Mitigation of and adaptation to climate change

Mangrove swamps have high productivity and contribute to coastal geomorphology and sediment deposition (Barbier *et al.*, 2011) through peat accumulation via roots (Middleton and McKee, 2001; McKee *et al.*, 2007).

4.5 Socio-economic benefits

Carbon sink function is maximized and atmospheric GHG concentrations are reduced, which minimize global shifts in climate temperature and precipitation (Moomaw *et al.*, 2018).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Table 51. Soil threats

Soil threats	
Soil erosion	Aquaculture and other practices less useful in coastal protection (Alongi, 2002).
Soil contamination/ pollution	Shrimp farming results in nitrogen eutrophication of the pond (Burford and Longmore, 2001).
Soil acidification	Pond aquaculture can lead to acid sulfate soils (Alongi, 2002).
Soil biodiversity loss	Seed banks of natural swamp lost during farming (Middleton, 1999).
Soil water management	Lowering water table for farming or forestry will increase CO_2 loss through decomposition of soil organic matter (Kolka *et al.*, 2018).

5.2 Increases in greenhouse gas emissions

The rehabilitation of mangrove forests ultimately reduces net CO_2 emissions (Wickland *et al.,* 2013; Cameron *et al.*, 2019). Freshwater wetland ecosystems are a noted source of methane (CH_4) emissions worldwide, though uncertainty is high given the rarity of long-term, continuous flux data, especially in tropical regions (Knox *et al.*, 2019; Zhang *et al.*, 2017), with emissions generally decreasing with salinity in coastal ecosystems (Poffenbarger, Needelman and Megonigal, 2011), including mangroves (Al-Haj and Fulweiler, 2020). Thus, CH_4 emission rates are often lower in mangroves than inland peatland forests, which, coupled with lower restoration costs, makes mangrove restoration generally a more cost-effective natural carbon sequestration method (Taillardat *et al.*, 2020).

5.3 Conflict with other practice(s)

Restoration conflicts with agriculture (rice, oil palm), infrastructure development (Friess *et al.*, 2019), and aquaculture ponds, which are productive for only 3-10 years (Cameron *et al.*, 2019).

5.4 Decreases in production (e.g. food/fuel/feed/timber)

Restoration negatively impacts production from agriculture and aquaculture (Cameron *et al.*, 2019) and other extractive activities (e.g., peat mining, logging). At the same time restoration provides opportunities for the development of useful products from the natural swamp e.g. food and medicine (Alongi, 2012).

6. Recommendations before implementation of the practice

Secure buy-in from local stakeholders through their participation in the restoration activities, as well as their long-term usage of restored mangrove resources.

7. Potential barriers to adoption

Table 52. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Environment may have shifted in the region, and physical conditions at the site may no longer be suitable for the desired target vegetation type (Middleton, 1999).
Cultural	Yes	Stakeholders who have used the land previously may object to a change in the land-use.
Social	Yes	See "Cultural Barrier" above.
Economic	Yes	Management of freshwater for coastal wetland restoration could potentially flood coastal communities (Meeder *et al.*, 2018); Harvest from aquaculture ponds supplies fish resources and reduces the chance of overharvesting from wild caught fisheries (Cameron *et al.*, 2019). Economic loss from land uses such as agriculture and other infrastructure.
Institutional	Yes	Legal barriers to restoration. Land tenure issues may direct restoration efforts to sub-optimal locations (Friess *et al.*, 2019).

Barrier	YES/NO	
Legal (Right to soil)		Securing land for the rehabilitation of mangroves may be difficult when the target property spans several human communities under various legal and land tenure restrictions (Cameron *et al.*, 2019).
Knowledge	No	Restoration approaches are generally well known (Middleton, 1999).
Other	Yes	Rewetting of former agricultural land results in net annual removals of CO_2 in the majority of organic soil classes and a lowering of net GHG emissions (Wilson *et al.*, 2016).

Table 53. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Mangrove restoration in abandoned ponds in Bali, Indonesia	Asia	10	6	17

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15. Restoration of organic coastal and inland freshwater forests

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Note: This document provides a specific perspective with regards to chapter 4.2.1 on peatlands restoration. For a complete overview of the practice, readers may also refer to chapter 4.2.1.

1. Description of the practice

Peatland forests occur worldwide in inundated soils where primary production and anaerobic conditions contribute to the building of soil organic matter (Günther *et al.*, 2020). Greenhouse gas emissions (GHG) can be substantial from drained freshwater forests with organic soils. Therefore, rewetting peat via hydrologic restoration (see factsheet n° 12 on Peatland restoration, this volume) can restore the function of these forests as carbon sinks and reduce their emission of certain components of GHG (Wilson *et al.*, 2016). While the drainage of forests with organic soil is often a part of the process of agriculture, forestry, and peat harvesting, drying of peat can contribute to GHG emissions (Wilson *et al.*, 2016; Günther *et al.*, 2020). Reflooding of organic forest soils to restore hydrology can lead to an increase in tree health, production and organic matter accumulation (Middleton, 1999, 2020a), and a considerable overall reduction in CO_2 and N_2O emissions (Wilson *et al.*, 2016). Depending on the duration and nature of the previous land-use, forested peatland restoration can be successful from seeds remaining in the seed bank or deposited via flood-pulsed dispersal (Middleton 1999, 2000, 2003). It is important to consider the nutrient status, hydrology and salinity of disturbed inland peat soils in peatland forest restoration (Chimner *et al.*, 2017). Furthermore, the overall functional equivalence of restored wetlands to natural wetlands is a matter of debate (Kolka *et al.*, 2018).

2. Range of applicability

Peatland forests are widely distributed across continents and exist in at least 180 countries (Parish *et al.*, 2008) in coastal, inland and high mountain settings. These wetlands are highly variable in hydrology and fertility. Forests include riparian, groundwater-fed and precipitation-fed types such as tropical freshwater hardwood and palm forests on floodplains (Dargie *et al.*, 2017), temperate swamp forest (Middleton, 2020a, 2020b), boreal bog forests, temperate bottomland forests and others (Kolka *et al.*, 2018). Though widespread on the landscape, peatlands cover only three percent of the land surface with an overall land area of approximately 4.23 km^2 (Xu *et al.*, 2018).

The rewetting of degraded worldwide peatland forests has a positive effect on GHG mitigation (Kolka *et al.*, 2018) in restored agricultural, forestry and excavated peatland. Natural forested peatlands vs. drained peatlands have lower levels of CO_2 emission, acidity, and compaction (Warren *et al.*, 2016; Silvius, 2014; Runkle and Kutzbach, 2014, respectively). Carbon losses associated with land clearing and cultivation are often very high in these ecosystems and recovery is slow (Warren *et al.*, 2016). Model results predict only about one third of the carbon lost during a single 25-year oil palm rotation is recovered 75 years after the restoration of tropical peatland forests (Warren *et al.*, 2016).

Recovery of forests can be difficult following farming. In freshwater bald cypress (*Taxodium distichum*) swamps of the southeastern United States, bald cypress does not rapidly reestablish in abandoned farmland because of the loss of soil seed banks and microbes during farming, and lack of dispersal in highly fragmented landscapes (Middleton, 1999, 2003). After bald cypress forest removal, recalcitrant cypress wood does not readily build carbon in the soil if forests are slow to reestablish (Middleton, 2020a). These considerations may be the main reasons that certain wet forested soils can be slow to build carbon after disturbance removes the trees.

3. Impact on soil organic carbon stocks

The rewetting of forested peatland has the potential of increasing carbon storage in the soil, and above ground plant biomass in tropical, temperate and boreal settings (Warren *et al.*, 2015; Wilson *et al.*, 2016, Table 54). Above ground carbon stocks represented by trees and above ground roots (e.g. knees) can contribute to large amounts of "teal carbon" soil organic matter, especially in anaerobic conditions with tree species with slow decomposing wood (Middleton 2020a, 2020b).

Table 54. Potential avoided emissions and changes in soil organic carbon stocks reported for restored degraded peatlands during the duration of study or model run

Baseline emissions of these systems are rarely known

Location	Context[1]	Avoided emissions and/or C sequestration	Duration	Methodology	More information	Reference
Tropical degraded peatlands	All peatland (forested and not forested) Restored vs. Degraded	Avoided emissions*: 16.7 (0.5–31.5) tC/ha/yr CO_2eq	Based on 2005–2006 land use	Modeled emissions from global mapping of degraded peatlands	Avoided emissions include CO_2, CH_4, N_2O and DOC	Leifeld and Meni-chetti (2018)
Temperate degraded peatlands		Avoided emissions*: 4.1 (2.6–5.4) tC/ha/yr CO_2eq				
Boreal degraded peatlands		Avoided emissions*: 4.6 (2.8–6.3) tC/ha/yr CO_2eq				
Tropical peat swamp forest; Southeast Asia (region-wide)	Reflooding of oil palm forests to restore swamp forest	Avoided emissions: 45, 110 and 140 tC/ha in dry, moderate and wet climate models	75 years	Modeled CO_2 emissions only	One year of oil palm cultivation offset by 6-20 years of restoration	Warren *et al.* (2016)
	Eliminating surface fires between oil palm rotations	Avoided emissions: 545.8 tC/ha	100 years		Estimate based on lack of fire only	
Tropical peatland forest; Southeast Asia (region-wide)	Restored swamp after 25-year oil palm rotation	C sequestration rate: 0.6 to 1.7 tC/ha/yr;	25 years		Biomass not included; model simulation	

Location	Context[1]	Avoided emissions and/or C sequestration	Duration	Methodology	More information	Reference
		Avoided emissions: 12 tC/ha/yr				
Tropical peat swamp forest; Southeast Asia (Indonesian Borneo)	Simulated reflooding of degraded peatland forest	Avoided emissions: 2.6 tC/ha/yr CO_2eq	2004-2006	Model	Model based on chamber measurements of CO_2 and CH_4	Page *et al.* (2009)
Tropical peat swamp forest; Malaysia, Brunei Darussalam & Indonesia	Converted oil palm plantation	Avoided emissions: 19–32 tC/ha/yr CO_2eq	1 year	Field study	Chamber measurements of CO_2, CH_4 and N_2O	Cooper *et al.* (2020)
Boreal forest, organic soil; global	Forest rewetting	Avoided emissions: 0.33-0.42 tC/ha/yr CO_2eq	Various	Review	Emission avoided, CO_2, CH_4, N_2O & DOC	Wilson *et al.* (2016)
	Cropland rewetting	Avoided emissions: 9.13 t CO_2 tC/ha/yr CO_2eq				
Temperate forest, organic soil; global	Forest rewetting; nutrient poor soils[1]	Avoided emissions: 2.31 tC/ha/yr CO_2eq				
	Cropland rewetting	Avoided emissions: 6.99 tC/ha/yr CO_2eq				
Tropical forest, organic soil; global		Avoided emissions: 14.97 tC/ha/yr CO_2eq				

Nutrient "poor" refers to rainwater fed settings with little external input of nutrients. *Area-weighted mean (Range)

4. Other benefits of the practice

4.1. Improvement of soil properties

After the snowmelt period, runoff on restored black spruce peatland is 25 percent of unrestored sites (Shantz and Price, 2006).

4.2 Minimization of threats to soil functions

Table 55. Soil threats

Soil threats	
Soil erosion	Water runoff reduced after restoration (Shantz and Price, 2006).
Soil salinization and alkalinization	Very low levels of soil salinity can cause a reduction in production, survival and regeneration with sea level rise, storm surge and freshwater extraction. Minimizing freshwater over-usage helps prevent vegetation damage (Middleton and Montagna, 2018).
Soil acidification	Reflooding likely reduces the soil acidification that occurs in drained organic forest soils (Silvius, 2014).
Soil biodiversity loss	Improve biodiversity with seed bank transfer, dispersal via flood pulsing or planting (Photo 17; Middleton, 1999)
Soil compaction	Reflooding may decrease compaction (Runkle and Kutzbach, 2014).
Soil water management	Reflooding can reduce regional flooding, particularly on floodplains at least partially managed for nature conservation (Middleton, 1999). Wet peatlands often regulate water flow. Evapotranspiration by trees on tropical floodplains may reduce water (Joosten, Tapio-Biström and Tol, 2012). Above-ground root systems of certain tropical and temperate floodplain trees maintain hydrology (Joosten, Tapio-Biström and Tol, 2012, Miroslaw-Swiatek and Amataya, 2017, Middleton, 2020b).
Soil organic matter accretion	Reflooding reduces organic matter decomposition (Wilson *et al.*, 2016a; Middleton, 2020a).

4.3 Increases in production (e.g. food/fuel/feed/timber)

Rewetting drained forests with organic soils increases production of trees and associated forest products (Middleton, 1999).

4.4 Mitigation of and adaptation to climate change

Reflooding drained forested peatland decreases CO_2 emission, but may increase CH_4 emission (Haddaway *et al.*, 2014), particularly in the years immediately following rewetting (Wilson *et al.*, 2016). It is important to note here that recent eddy covariance studies conclude that natural peatland forests may have higher CH_4 emissions than previously thought (Deshmukh *et al.*, 2020; Wong *et al.*, 2020), possibly because previous chamber studies did not capture the efflux of CH_4 via tree stems (Pangala *et al.*, 2013; Covey and Megonigal, 2019). On the other hand, N_2O emissions tend to decrease with rewetting (Wilson *et al.*, 2016).

4.5 Socio-economic benefits

Carbon sink function is maximized, so that rewetted organic forests can reduce global heating (Moomaw *et al.*, 2018; Günther *et al.*, 2019), and emissions, air quality impacts and risk to property from fire (Turetsky *et al.*, 2015; Warren *et al.*, 2016).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Table 56. Soil threats

Soil threats	
Soil biodiversity loss	Suitable plant material may not be available because forested wetland seeds die after drainage for farming (Photo 17; Middleton, 1999).
Soil water management	Reflooding reduces carbon release to soil water and surface water (Wilson *et al.*, 2016).

5.2 Increases in greenhouse gas emissions

Previous land use likely has a large influence on GHG emissions after rewetting. For example, CH_4 emissions following the reflooding of abandoned farmland can be very high (Hendriks *et al.*, 2007; Harpenslager *et al.*, 2015). In contrast, rewetted boreal peatlands after tree removal have much lower levels of CH_4 emission (Tuittila *et al.*, 2000; Waddington, Tóth and Bourbonniere, 2008), but more studies are needed on different land use transitions (Wilson *et al.*, 2016). See 4.4 above regarding trade-offs between CO_2, CH_4 and N_2O emissions.

5.3 Conflict with other practice(s)

Raised water levels decrease value for agriculture, forestry and peat extraction (Haddaway *et al.*, 2014), and obstruct canals and ditches sometimes used as waterways for transportation. In particular, adjacent land usage that relies on drainage may be threatened regionally (Joosten, 2014) including the entire peatland and surrounding areas. Only species adapted to high water tables would thrive after thorough rewetting.

5.4 Decreases in production (e.g. food/fuel/feed/timber)

Tree species less adapted to flooding will have lower productive capacity after reflooding (Middleton, 1999), with a subsequent decrease in soil organic matter accumulation depending on the decomposition rate or organic matter (Middleton, 2020a). However, the productive capacity of wetland species increases (see e.g., the high productive capacity of mangrove forests that often generate peat soils).

5.5 Other conflicts

Soil carbon is a balance of production levels and decomposition rates. Higher primary production occurs in low levels of flooding, even though the lowest rates of decomposition occur in more deeply flooded sites (Middleton, 2020a). While deeper water may be easier to maintain in impoundments, natural water regimes including at least occasional periods of drawdown during the growing season maximizes the production of most wetland tree species (Middleton, 1999, 2020a).

6. Recommendations before implementation of the practice

Subsidence in farmed tropical peatland eventually may make farming impossible (e.g. drained peatland in Southeast Asia (Hooijer *et al.*, 2014)), so that restoration may become an attractive land use alternative for carbon mitigation (see factsheet n° 12 on Peatland restoration, this volume). The assumption is that hydrologic restoration restores the function of the abandoned farmland as a carbon sink over time, and that CO_2 removal in rewetted organic soil is higher than in drained soils (Wilson *et al.*, 2016). However, carbon loss from peat during farming can greatly exceed the capacity of this type of peatland to recapture atmospheric carbon after restoration. Models suggest that 75 years after hydrologic restoration, only as little as 1/3 of carbon is recaptured that was released to the atmosphere during 25 years of oil palm cultivation (Warren *et al.*, 2016). For example, over 100 years of cultivation consisting of four oil palm rotations with burning, more carbon was lost from peat than is naturally accumulated by these peatlands over 3000 years (Warren *et al.*, 2016). Furthermore, drained and degraded peatlands that are not hydrologically restored continue to lose soil carbon and have an increased fire risk for many years. This situation is particularly problematic if the abandoned farm site is not actively managed (Turetsky *et al.*, 2015; Warren *et al.,* 2016; FAO, 2012).

7. Potential barriers to adoption

Table 57. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Permanent changes in hydrology e.g. downcutting of channel feeding floodplain (Middleton, 1999).
Cultural	Yes	Farmers, peat miners, and other land managers may object to a change in land use to restored wetland because of traditional, drainage-based usage.
Social	Yes	A change to one part of the watershed may affect the entire region (Joosten, 2014). Stakeholders may view rewetting as irreversible although it would be possible to re-dig ditches or canals, or remove dams or drainage obstacles.
Economic	Yes	Flooding in urban areas (Meeder *et al.*, 2018). Decreased flooding downstream impacts agriculture (Silvius, 2014).
Institutional	Yes	May require public funding and planning (Joosten, 2014).

Barrier	YES/NO	
Legal (Right to soil)	Yes	Legal barriers to restoration e.g. property flooding; land regulations; traditional land use may not be possible after restoration so that land is no longer as valuable.
Knowledge	Yes	Success of vegetative restoration is elusive (Middleton, 1999). The capacity of rewetted peatlands to act as carbon sinks is not well known for tropical peat swamps (Wilson *et al.*, 2016).
Other	Yes	Rewetting of former agricultural land can produce high levels of CH_4 emission (Wilson *et al.*, 2016).

Photo of the practice

Photo 17. Regeneration of vegetation in drained organic forest can occur after short-term farming from seeds in the seed bank, dispersal or replanting (Cache River, Southern Illinois)

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16. Water level management in rice paddies

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1. Description of the practice

Water level (or water sheet) management is a key to allow SOC sequestration and the GHG emissions balance in wetland-cultivated areas. The water sheet level along the soil profile governs the emissions of CH_4, N_2O and CO_2: during the flooding period, the anaerobic soil conditions lead to increased CH_4 emissions (due to the activity of anaerobic soil bacteria) and decreased CO_2 emissions (due to the limited mineralization from soil aerobic bacteria). This also generally leads to increased SOC sequestration rates. On the contrary, during dry periods, the aerobic soil conditions lead to decreased CH_4 emissions and increased CO_2 and N_2O emissions.

In the case of rice fields, the water sheet position in the soil profile can also determine crop yield. For example, a submergence or high water sheet allows a better weed control and a higher efficiency of fertilizer use. On the other hand, drainage is important to a correct soil-plant-atmosphere exchange of substances such as ammonia-N, hydrogen sulphide and other harmful substances that are produced under reductive conditions during submergence. The global impact of the management practices chosen by rice farmers in each agricultural region is by no means a minor issue, since rice is the major cereal crop for more than half of the world's population and about 750 million tons of rice are grown worldwide each year. Rice fields cover 1.29 million km^2, just over 10 percent of the world's wetlands (Ramsar, 2018).

Rice fields (paddies) can be cultivated in several water regime conditions:

- Continuous waterlogging (CW): flooding is permanent, except when soil conditioning, fertilization, and rice harvesting tasks are performed, and;
- Alternate wetting and drying (AWD): the most commonly used water regime that consists of alternation of flooding with drainage until the soil is dry, or even only damp, and flooding again

(Photo 18, Photo 19 and Photo 20). It aims to control the water sheet height, raising and lowering it to different depths, but always at levels that ensure an adequate humidity for the development of rice plant. Different actions can be undertaken to manage the water table height, such as:

- Mid-season drainage (MSD): waterlogged soil is drained for 30 days just 21 days after transplanting; during this time the soil is dry or damp, and remains flooded the rest of the time;
- Alternative wetting (AW): waterlogged soil is drained several times throughout the rice cycle, shortening times intervals between drying acts;
- Intermittent irrigation (II): repetition of drainage and irrigation that decreases the amount of water used and ensures longer aerobic conditions;
- Controlled irrigation (CI): during the crop cycle the soil remains more dry or damp than waterlogged, i.e. higher deep flooding of 10 cm or more in height of water layer, leaving paddy soil dry at 60–80 percent during the growing season (without flooding after the re-greening of rice seed).

♦ Rainfed irrigation with sprinkler (S): soil is not flooded at any time during the rice growing cycle but irrigated to ensure that soil moisture is maintained at the levels required by the crop.

In a context of climate change and increasing water scarcity in many wetlands around the world, the irrigation system used becomes very relevant. The use of sprinklers in rice as it can be a smart water management that contributes effectively to reduce water consumption. Both systems (AWD and S) also reduce costs, another of the great challenges of modern agriculture.

2. Range of applicability

Water sheet management can be applied in several climate conditions, from tropical and sub-humid to Mediterranean areas. The key for establishing in a simple way the suitability of the AWD strategy is to take into consideration basic environmental aspects such as the climatic parameters (temperature, rain and potential evapotranspiration) and some soil properties (percolation potential and texture). Based on their combination, the farmer must select the best strategy to drain while maintaining a level that does not reduce crop yield and allow agricultural work. In the Philippines, for example, 96.7 percent of paddy areas are moderately or highly adequate to AWD during the dry season; however during the wet season the abundant rains typical of the monsoon period and a moderate to high clay content leads to excessive water amounts that can be difficult to drain in wide rice areas (Sander *et al.*, 2017). In drier climates (e.g. Mediterranean climate) AWD can be applied successfully when the rice fields conditions are adequate, reducing methane emissions and water consumption without decreasing rice yield. An important factor is that if there is a scarcity of water due to low rainfall, the management regime significantly influences the physicochemical characteristics of the soil and therefore increases the importance of observing their long-term effects (Fangueiro *et al.*, 2017; Meijide *et al.*, 2017).

3. Impact on soil organic carbon stocks

Studies in areas with a traditional rotation of rice management show that long waterlogging enhances a slow accumulation of organic carbon in wet soil conditions due to the organic-mineral associations developed during the first 100 years of soil development. A meta-analysis carried out in China associates the highest storage capacity of C with the pedogenic subgroups of Fe-accumulating and Fe-leaching paddy soils in the long term (Pan *et al.*, 2003).

This thesis explains why cultivation in traditional flood conditions shows a greater contribution of organic carbon than non-flooded highland crops. For example, values in eastern China for A horizons range from 17.8 to 30 g/kg C in paddy rice under cultivation for 50 to 2000 years, while they only reach 11 g/kg C in non-paddy soils (Wissing *et al.*, 2014); or 15-16 g/kg C in recent paddy soils (Xiong *et al.*, 2015; Chen *et al.*, 2017). It would appear, then, that in comparison with rainfed cultivation, irrigated rice cultivation in China led to increased SOC storage in paddy soils. Pan *et al.* (2003) reported that paddy cultivation induced total C sequestration to half of China's total annual CO_2 emission in the 1990s.

AWD can have negative effects on carbon sequestration as the time that soil is dry increases, but these can be counteracted by implementing different cultivation techniques (e.g. no-till, biomass incorporation, improved varieties). Table 58 shows SOC values in rice fields cultivated in CW and AWD; during various periods of time, with incorporation of biomass (CB) or rice straw (RS) and without incorporation (conventional tillage, CT). As a general idea, values in the Table 58 indicate:

- Lower OC accumulation in short duration experiences regardless of water level management (CW vs. AWD) (Huque *et al.*, 2017 *vs* Zhang *et al.*, 2010, Chen *et al.*, 2017 and Xionghu *et al.*, 2011);
- Importance of time on OC accumulation in relation to the change in water management. Haque *et al.* (2017) reported minimal differences for a one-year duration while Chen *et al.* (2017) showed an evolution of carbon stocks associated with drying cycles in paddy during more than 100 years
- Positive effect of rice straw incorporation (RS). Zhang *et al.* (2010) and Xionghu *et al.* (2011) showed similar records of stocks even though the number of crop years are very different (several thousand years *vs* 28 years)
- Positive and similar effect of cover crop biomass (B) incorporation between WC or AWD management (Haque *et al.*, 2017)

Table 58. SOC stocks and changes in soil organic carbon stocks reported for water level management

Location (Reference)	Climate (MAR, MAP)	Soil type	Crop and soil management	Water management	SOC stock or concentration	Depth (cm)	Duration	Fertilizer additions (t/ha)
Tai Lake plain, Yiangsu, Eastern China (Zhang *et al.*, 2010, data recalculated)	Subtropical monsoon (1177 mm, 15.7 °C)	Entic Hapludept 39% clay content 24 g C/kg 1 g/cm^3 pH: 6.5	Rice variety: *cv. Wuyunjing 7*	CW	34.4 tC/ha	0–15	paddy for several thousand years	Inorganic fertilizer + urea
Wnagcheng Hunan Province, Southern China (Xionghui *et al.*, 2012)	1370 mm, 17°C	Reddish Yellow Paddy Soil pH: 6.6	- Rice variety: *Conventional and Hybrid (Xiang 67);* - Double rice (DR) - Typical local management: rice straw incorporation (RS) and conventional tillage (CT)	CW	33.2 tC/ha Additional C storage: 0.075 tC/ha/yr	NA	28 years	RS: 2 x 2.63 t/ha NPK : *Early + late* N: 0.15+0.18 t/ha P: 2 x 0.038 t/ha k: 2 x 0.0996 t/ha
Taoyuan County Hunan Province,	Tropical Humid monsoon	Soils derived from		CW	42.9 tC/ha	0–20	paddy for over 100 years	

<table>
<tr><th>Location (Reference)</th><th>Climate (MAR, MAP)</th><th>Soil type</th><th>Crop and soil management</th><th colspan="2">Water management</th><th>SOC stock or concentration</th><th>Depth (cm)</th><th>Duration</th><th>Fertilizer additions (t/ha)</th></tr>
<tr><td>Southern China, Asia
(Chen et al., 2017)</td><td>(1450 mm, 16.5 °C)</td><td>quaternary red clay</td><td></td><td colspan="2">AWD
Wet season: wetting and dry cycles
Dry season: dryness</td><td>35.5 tC/ha</td><td></td><td>Double rice under conventional tillage weeds during 8 years abandonment</td><td></td></tr>
<tr><td rowspan="2">Jinju, South Democratic People's Republic of Korea
(Haque et al., 2017)</td><td rowspan="2">Continental monsoon
(1528 mm, 13.5 °C)</td><td rowspan="2">Silty loam
20.4 g OC/kg</td><td rowspan="2">- Rice variety:
v. Japonica
- Rice under different cover crop biomass incorporation (CB0, CB1, CB2 and CB3)</td><td rowspan="2">5–7 cm water when flooded</td><td>CW</td><td>CB0: 4.7 gC/kg
CB1: 5.99 gC/kg
CB2: 7.67 gC/kg
CB3: 5.78 gC/kg</td><td rowspan="2">NA</td><td rowspan="2">1 year</td><td rowspan="2">Rates of cover crop biomass incorporation and NPK (t/ha):
CB0: 0; CB1: 3; CB2: 6; CB3: 12
NPK: N:0.09; P:0.02; K: 0.048</td></tr>
<tr><td>AWD (MSD): drain for 30 days,
21 days after transplanting</td><td>CB0: 4.59 gC/kg
CB1: 5.87 gC/kg
CB2: 7.61 gC/kg
CB3: 5.72 gC/kg</td></tr>
</table>

CW: Continuous Waterlogging; **AWD**: Alternate wetting and drying; **MSD** Midseason Drainage; **DR**: Double rice; **CT**: Conventional Tillage; **RS:** rice straw incorporation

4. Other benefits of the practice

4.1. Improvement of soil properties

The main improvement derived from lowering water sheet through drainage and increase the drying periods is that translate into aerobic conditions that improve the oxidative condition of the soil and reduce the redox processes of Fe, improving the permeability of the soil in the next fallow season (Chen *et al.*, 2017; Shiratori *et al.*, 2007).

Mid-season drainage is important to cut-off the supply of ammonia-N and to secure some desirable plant characteristics in relation, for example, to upper leaves, lower inter-node, ear formation, and healthy root growth. In addition, this action removes hydrogen sulphide and other harmful substances, which are produced by microbial action under reductive conditions of submergence (Amin, Rowshon and Aimrun, 2011).

During the time that the water sheet is elevated and paddy rice is flooded, anaerobic conditions increase the total and hot water extractable carbon in soil, decrease bulk density and induce soil aggregates stabilization (Shiratori *et al.*, 2007; Wissing *et al.*, 2014).

4.2 Minimization of threats to soil functions

Table 59. Soil threats

Soil threats	
Soil salinization and alkalization	In coastal zones, after flooding, salinization is reduced by dilution effect; in Thailand, for example, water drainage reduced soil salinity by 50 percent (Maeght *et al.*, 2005).
Soil pollution / contamination	Maintenance of the water sheet below the topsoil reduces absorption of contaminants in soils watered with water enriched or pollution, i.e. arsenic (Spanu *et al.*, 2012).

4.3 Increases in production (e.g. food/fuel/feed/timber)

Management of the water sheet can condition rice yield. In Spain, Fangueiro *et al.* (2017) found that an increase in drying time leads to a decrease in production that can reach up to 50 percent when CW is replaced by rainfed irrigation and conventional tillage. With abundant rainfall and under NT such as in subtropical Chinese rice fields, the substitution of CW for AWD implies a drop in production of only 10 percent (from around 7.5 to about 6.5 t/ha) (Xu *et al.*, 2015). On the contrary, when NT conditions are used in semi-arid areas of Spain, the change of the irrigation system even implies a slight increase in production.

4.4 Mitigation of and adaptation to climate change

In general, implementing AWD management translates into lower emissions as compared to CW practices. Collected data in different climatic conditions corroborate that widen soil dry periods reduces CH_4 and in consequence the Global Warming Potential (GWP) and the intensity of greenhouse gases (Sriphirom *et al.*, 2019; Nelson, 2015). In addition, Shiratori *et al.* (2007) showed that installing subsurface drains beneath poorly drained clayey soil rice fields (0.6–0.8 m below) also aid in reducing in wet season methane emissions up to 70 percent. In Haque *et al.* (2017), biomass incorporation is translated in increased emissions due to mineralization, affecting the GWP (Table 60).

Table 60. GHG emissions and Global Warming Potential (GWP) according to water level management CW vs AWD or Sprinkler

Location (Reference)	Climate (MAP, MAT)	Soil type	Duration (Years)	Crop and soil management	Fertilizer additions	Water management		GHG emissions (t/ha/yr)			GWP $tCO_2eq/ha/yr$
								CH_4	N_2O	CO_2	
Po Valley, Italy, (Meijide *et al.*, 2017)	Mediterranean (650 mm, 12.3 °C (2009))	Loam to clay loam, Calcic Gleysol	2 years CW (2009) AWD/MSD (2010)	Non cultivated after harvest	N (inorganic fertilizer) 2009: 0.13 t/ha 2010: 0.12 t/ha	CW		0.37	0.0008	NA	11.48
						Midseason drainage (AWD/MSD)		0.21	0.0011		2.89
Extremadura region, Spain (Fangueiro *et al.*, 2017)	Mediterranean (< 480 mm, 16.8 ^{o}C)	Loam Hydragic Anthrosol	3	Conventional tillage	N inorganic fertilizer: 0.14 x 2 = 0.28 t/ha	10 cm water when flooded	CW	0.35	0.011	6.7	19.4
							Sprinkler	0.004	0.008	10.2	12.4
				No-till			CW	0.12	0.014	5.3	12.5
							Sprinkler	0.001	0.006	5.8	7.39
Lucknow, India (Tyagi, Kumari and Singh,2010)	Humid Subtropical (1001 mm, 25.2 ^{o}C)	Sandy loam	1	Rice variety: *v. Somali 4006*	NA	CW vs. Midseason drainage (AWD/MSD) In all cases 4-8 cm water when flooded Drainage: MSD1: once at vegetative cycle MSD2: 7 days, 70 DAP AW: 3 days, 21 and 77 DAP		*Daily average (kg/ha/day)* CW: 3.5 MSD1: 3.15 MSD2: 2.2 AW: 2.04	NA	NA	CW: 8.15 MSD1: 7.41 MSD2: 5.1 AW: 4.82

Location (Reference)	Climate (MAP, MAT)	Soil type	Duration (Years)	Crop and soil management	Fertilizer additions	Water management	GHG emissions (t/ha/yr) CH$_4$	N$_2$O	CO$_2$	GWP tCO$_2$eq/ha/yr
Hubei, lowland Central China (Xu *et al.*, 2015)	Subtropical monsoon (1500 mm, 17.5 ^{o}C)	Silty clay loam	1 (drought year in study area)	No-till Rice varieties: *v HY3* (Drought-resistant) and *v HYY299* (Typical)	NA	CW	*v. HY3:* 0.96 *vHYY299:* 0.92	NA	*v. HY3*: 9.25 *vHYY299:* 7.45	*v. HY3:* 35.55 *vHYY299:* 32.26
Hubei, lowland Central China (Xu *et al.*, 2015)	Subtropical monsoon (1500 mm, 17.5 ^{o}C)	Silty clay loam	1 (drought year in study area)	No-till Rice varieties: *v HY3* (Drought-resistant) and *v HYY299* (Typical)	NA	Alternative wetting (AWD/AW) Drainage: AW1: soil always damp AW2: let soil dry	AW1 *v. HY3:* 0.37 *vHYY299:* 0.38 AW2: *v. HY3:* 0.18 *vHYY299:* 0.15	NA	AW1: *v. HY3*: 12.14 *vHYY299*: 15.44 AW2: *v. HY3:* 18.05 *vHYY299*: 17.83	AW1 *v. HY3:* 23.99 *vHYY299:* 27.99 AW2 *v. HY3:* 25.50 *vHYY299:* 24.49
Jinju, South Democratic People's Republic of Korea (Haque *et al.* 2017)	Continental monsoon (1 528 mm, 13.5 ^{o}C)	Silty loam	1	Rice variety : *v. Japonica*	N : 0.09 t/ha P : 0.02 t/ha K : 0.048 t/ha Cover crop biomass incorporation (t/ha): CB0: 0 CB1: 3 CB2: 6 CB4: 12	5–7 cm water when flooded: CW	NA	NA	CB0: -1 CB1: 5 CB2: 6 CB4: 12	CB0: 5 CB1: 8 CB2: 23 CB4: 40
Jinju, South Democratic People's Republic of Korea (Haque *et al.* 2017)	Continental monsoon (1 528 mm, 13.5 ^{o}C)	Silty loam	1	Rice variety : *v. Japonica*	N : 0.09 t/ha P : 0.02 t/ha K : 0.048 t/ha Cover crop biomass incorporation (t/ha): CB0: 0 CB1: 3 CB2: 6 CB4: 12	5–7 cm water when flooded: Midseason drainage (MSD/AWD) Drainage for 30 days, 21 DAP	NA	NA	CB0: -1.8 CB1: 2 CB2: 5 CB4: 11	CB0: 4 CB1: 6 CB2: 16 CB4: 21

CW: continuous waterlogging; AWD: alternate wetting and drying; MSD: midseason drainage; AW: alternative wetting; DAP: days after planting; CB: cover biomass incorporation

4.5 Socio-economic benefits

On one hand, an efficient water sheet management is a good measure for saving resources in areas with water shortages where it is necessary to purify it, or extract from wells, and therefore the economic cost is higher. For example, for wetter weather conditions as in the Philippines, the generalized application of AWD would save 30 percent of water (Sander *et al.*, 2017; Nelson *et al.*, 2015). On other hand, fertilizer consumption can be reduced because the midseason drainage of the otherwise flooded field can outcompete the effect of organic fertilizer application regarding CH_4. In addition, in most cases implies lower evapotranspiration rate (Meijide *et al.*, 2017).

4.6 Additional benefits to the practice

Studies of water level fluctuations in rice fields show that increases in anaerobiosis time can favors the growth of other plant species. An example is explained by Nishio *et al.*, (2006) who established that the fluctuations of water level in flood plains and the artificial management of water level in the rivers around rice paddies influenced the spatial distribution of *Ud. nipponensis*.

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Table 61. Soil threats

Soil threats	
Soil salinization and alkalization	In sensible coastal wetlands, shortening flood periods or decreasing the height of the sheet of water above the ground can exacerbate soil salinization due to marine intrusion (Moreno-Ramón *et al.*, 2015).
Soil compaction	In aerobic conditions, the faster decomposition of OM tends to limit soil aggregates stabilization (Wissing *et al.*, 2014).
Soil biodiversity loss	With minimum water use the soil condition are very different and changes in biota and weed can be important (Nielsen *et al.*, 2009).

5.2 Increases in greenhouse gas emissions

AWD management often results in an increment, sometimes important, of CO_2 and N_2O emissions. As in concerned Mediterranean areas, drainage of the water table in the middle may not significantly increase fluxes of N_2O with respect to the absence of drainage (Table 60).

5.3 Decreases in production (e.g. food/fuel/feed/timber)

In general, in any weather and soil conditions, mid-season drainage, rain-fed and other water-saving techniques resulting in reduced the yield grain (Meijide *et al.* 2017). For avoiding the decrease of yield under AWD management, some complementary measures can be used, such as timely irrigation, coordinating irrigation with fertilization and weed control (Zhi, 2000). For example, field experiences of Fangueiro *et al.* (2017) indicate that plots with NT and S irrigation obtain a grain yield (8.2 t/ha), which is similar to the yield of fields with CW and CT (between 6.7 and 8.9 t/ha)

6. Recommendations before and during the practice implementation

The success of the implementation of water saving practices is based in simultaneous improving agronomic practices. It is therefore necessary to know soil properties such as EC, texture, bulk density and pH, and climatic conditions, which determine water movement in soil and its fertility (Amin *et al.*, 2011; Fangueiro *et al.*, 2017). The main available strategies include fertilization and other agronomic management, developing improved varieties, changing the crop planting date, optimum use of rainfall, supplementary irrigation in rain-fed fields, improving water distribution, and water reuse or recycling. For example, mid-summer drainage, intermittent irrigation, and subsurface drainage systems can accelerate the leaching of reducible iron, to results in degradation of paddy fields, so application of iron materials is a possible countermeasure for increasing soil oxidation capacity.

In general, a good option could be the incorporation of organic matter amendments, although always without exceeding a certain threshold level from which production decreases again (Haque *et al.*, 2017). In any case, we must not forget that more amendment implies more SOC and usually more emission.

A common recommended practice for many regions of Asia and that is working well in Philippines, Vietnam, and Bangladesh (Nelson *et al.*, 2015) consists in starting the dry phase during the growing season, about 2–3 weeks after transplanting (or 3–4 weeks after sowing). Generally, the field dries until the saturated soil zone reaches a level of approximately 10-15 cm below the soil surface. At that time, it is irrigated again until the standing water reaches 3-5 cm on the field. The whole process can be controlled by piezometers (Photo 21). Depending on the soil and rain regime of the area, soil-drying time can take up to 10 days. In any case, the choice of the time of irrigation, with the soil dry or still somewhat humid, is relevant because grain yield and aboveground biomass are less if you wait for is fully dry (Xu *et al.*, 2015). One of the things to keep in mind with these practices is that high levels of water are required after transplanting and proper rooting, as well as during the flowering stage. On the other hand, low water levels are required in tillering, panicle development and maturation stages. About 5 cm water depth is needed at milk stage for translocation of nutrients stored in plant body to ear or panicle for healthy development of developing grain or spikelet (Amin *et al.*, 2010).

A recommended practice applied in the South Democratic People's Republic of Korea (Haque *et al.,* 2017) consists of incorporation of cover crop biomass (3 t/ha), which maintained an optimal and increasing level of rice paddy yield compared to CW. The yield differences between both systems (AWD and CW) did not show significant differences between them, but AWD reduced water inputs and greenhouse gas emissions. Therefore, in rainfed areas, the use of complementary methodologies such as the application of a thin transparent polyethylene film, or the incorporation of straw, can increase the efficiency of this management; especially if what is sought is to apply irrigation to keep the soil moist without standing water.

7. Potential barriers to adoption

Table 62. Potential barriers to adoption

Barrier	YES/NO	
Cultural	Yes	Tendency among farmers to flood the field as much as possible (Howell, Shrestha and Dodd, 2015).
Economic	No	Cost of supplementary measures (drainage in the wet season, weed control, preliminary tests of management viability, etc.)
Knowledge	Yes	Individual farmer may not be aware that AWD practice can save water for his or her fellow farmers' field without compromising the yield on his/her field. Some farmers perceived that AWD will increase weed infestation (Howell, Shrestha and Dodd, 2015). Hence, building awareness will be necessary.

Photos of the practice

Photo 18. Removing water in AWD (Valencia-Spain)

Photo 19. Flooding conditions in rice fields (Albufera of Valencia - Spain)

Photo 20. Dried fields in Mediterranean wetland (Abufera of Valencia - Spain)

Photo 21. Piezometer

Table 63. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Water regimes in rainfed rice-paddies in Indonesia and Thailand	Asia	1 and 3	6	16

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17. Straw residue management

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1. Description of the practice

Possible alternatives for the management of rice residues are: i) incorporation ii) burning, or iii) elimination. Stubble burning has been widely used in the past, and refers to the deliberate act of applying fire to the straw stubble that remains after the rice is harvested. Burning straw can show positive and negative effects on soil and crop yield in the short and long term, depending on the burning technique, the regional climatic and topographic conditions. It is necessary to be careful with a high intensity fires, high slope topographies or torrential rains. In general, burning a crop in the field is not the best recommended choice as it results in air pollution and high CO_2 emissions in nearby areas with people, and may damage soils (Whitbread *et al.,* 2003). However, the scientific community and the public acknowledge that burning residues through the use of slash-and-burn agriculture, some communities do not rely on external inputs based on fertilizers, pesticides and irrigation with the use of fossil energies. Those agroecosystems are irreplaceable and ecologically acceptable for that type of communities that generally are located in developing countries (Kleinman *et al.*, 1995). Another straw management that is used by farmers, is to cut and remove rice straw for another use outside the field (e.g. for livestock feeding, biosolids).

Residues can also be left on the soil after harvest, and incorporated into the upper soil layer, leading to a slow plant biochemical degradation. Specific machinery used in rice paddies in rotation allows the cutting and removal of rice straw, in order to sow the next crop (wheat or another cereal) on the bare soil, and after planting the deposition of straw as mulch. Rice straw can also be used in compost preparations and, be returned to the soil as a source of stabilized organic matter. Both options can also be carried out with other plant residues such as corn or wheat, in areas where rice is grown in rotation. Another recent possibility is the use of biochar, which is a C-rich by-product obtained by biomass pyrolysis under limited oxygen conditions (Also see factsheet n° 17 on Biochar, volume 3). In any of the cases (fresh residue incorporation, compost or biochar), its incorporation can be carried out-with or without tillage or accompanied or not by inorganic N or NPK fertilizers applications. Rice straw incorporation is considered as a good agronomic practice as long as site-specific management practices are implemented to improve soil properties and increase rice production (Asai *et al.*, 2009; Jiang *et* *al.*, 2019). Regardless of how the organic matter is applied in the paddy fields (with or without N or NPK fertilizer application, with or without tillage, or simple deposition as mulch), it is widely documented that incorporating rice straw to the soil seems to increase SOC stocks but it negatively affects GHG emissions. Recent studies conclude that the intensity of the effects (positives and negatives) will depend on: i) the climate; ii) the type and the amount of fertilizer; iii) the amendment (type, amount and the frequency); iv) the initial content of nitrogen; v) organic carbon and other soil properties; and vi) duration of the period of soil saturation (see sections 4 and 5).

2. Range of applicability

The agronomic benefits derived from the incorporation practices have shown to be important in all type of soils and rainfall situations, as is evidenced by different meta-analyzes carried out in many paddy rice areas with different environmental conditions in China and India (Huang *et al.*, 2013; Srinivasarao *et al.*, 2014). Specifically in terms of soils, positive effects have been registered from depleted soils (oxisols) to poorly developed soils (inceptisols), including also very heavy soils with silt layer (vertisols) and from low-nitrogen content soils (<1) to high N content soils (> 1.5 kgN per kg of soil). In reference to the climatic conditions, these studies focus on paddy areas with annual mean precipitation from < 1 000 mm to more than 1 500 mm and from cool annual mean temperatures (< 10 ºC) to high temperatures (> 20 ºC).

Incorporation of rice straw is already common practice in large areas of the world's most important rice growing areas. In this sense, alternatives that are more efficient are being explored recently in relation to the specific environmental conditions of the crop. For example, critical C inputs requirements for maintaining SOC stock are highest in coarse-textured soils and highest rates of rainfall. Consequently, in the cold northeast China region the return of soil fertility due to the addition of straw is very slow, and biochar is a possible solution for a much faster restitution of SOC (Sui *et al.*, 2016).

In addition, in semi-arid tropics (where rainfall exceeds potential evapotranspiration only two or four months of year), if rice-wheat residues are pyrolized before soil addition, about 50 percent of the C in biomass can be returned to soil (Srinivasarao *et al.,* 2014). The straw addition technique is also applied in arid and semi-arid Mediterranean environments where traditional burning was replaced by the incorporation of the fresh residue (Jégou and Sanchis-Ibor, 2019).

3. Impact on soil organic carbon stocks

As indicated before, compared to the removal option, crop residue retention is always a good option for increasing soil C storage. And as for the type of residue, in principle biochar would be the best alternative, since it can be known as an indicative value that the carbon content of the rice straw biochar can double the total C content of the initial vegetable residue. In traditional paddy rice fields in northern China, biochar addition of 1.78 t/ha increases soil C by up to between 6 and 12 g/kg. In the case of straw rice incorporation (5 t/ha), increases reaches 4 g/kg of C. While it may sometimes seem that the rice straw amendment causes minimal changes in the total C content of the soil, a meta-analysis from China suggests that soil C saturation would occur after 12 years of straw C input (Zhu *et al.*, 2014).

In Chinese clay loam soils, straw reintroduction led to lower SOC concentration increases than in sandy loam and silty soils (Liu *et al.*, 2014). In addition, rotary tillage rice straw (6 t/ha) would lead to higher SOC increases than conventional tillage. The net increase was higher at 14-21 cm depth (18.4 percent) than 7-14 cm (8.7 percent) and 0-7 cm depths (7 percent) (Zhu *et al.*, 2014).At last, the application of biochar doses above 30 t/ha triggers carbon immobilization well above the increase that can be achieved with the addition of non-composted straw (50-90 percent for biochar versus 10-20 percent for straw), regardless of whether it has been used nitrogen fertilization (usually urea). Lower doses of biochar do not seem to enhance carbon sequestration much more than that obtained by incorporating straw residue rates of 3-6 t/ha (Table 64).

Table 64. Changes in soil organic carbon stocks reported for straw residue management

Location	Climate (MAR, MAP)	Soil type	SOC stocks (tC/ha)		Annual Cseq (tC/ha/yr)	Depth (cm)	Duration (Years)	Crop and soil management	NPK Additions (t/ha)	Biomass incorporation (t/ha)	Reference
Shenyang, Liaoning Province, Northeast China	Semi-humid temperate, continental monsoon (500 mm, 8.3 °C)	pH: 6.7 BD: 1.31 g/cm^3 C/N: 9.17	*C: 40.7 *RS:44.7 *B1: 46 *B2:48.9 *B3: 62.7	*C: 39.7* *RS:44.3* *B1: 44* *B2:44.4* *B3:76.5*	NA	0–20	2	- Rice variety: *Japonica Shennong 265* - Upland crop-single rice	With urea : N: 0.21 P_2O_5: 0.62 K_2O: 0.2	RS: 5.05 (370 gC/kg; 7 gN/kg) incorporated before transplanting at 5 cm; B1: 1.78 B2:14.8 B3: 29.6 Biochar composition: 671 gC/kg; 8.1 gN/kg	Sui *et al.* (2016) data recalculated
			**C: 23.6* **RS:28.2* **B1: 29* **B2: 30* **B3: 33.7*	*C: 21.93* *RS: 27.5* *B1: 26* *B2: 27.7* *B3: 35.9*		20–40					
Tai Lake plain, Yiangsu, Eastern China	Subtropical monsoon (1 177 mm; 15.7 °C)	Entic Hapludept 24 gC/kg BD= 1 g/cm^3	C: 34.4 B1: 39.02 B2: 48.06		NA	0–15	1	- Rice variety: *cv. Wuyunjing 7* - Water management: AWD Flooding-Drainage-re-Flooding -Moist	With urea: N: 0.3 P_2O_5: 0.13 K_2O: 0.13	B1: 10 B2: 40	Zhang *et al.* (2010) data recalculated
Wnagcheng and Changsha County, Hunan Province,	1 370 mm, 17°C	pH: 6.6 20.03 gC/kg 2.1 gTN/kg	*C: 33 NPK: 34.6 RS+NPK:36.3		*C: -0.07 NPK: 0.08 RS+NPK:0.1		28	- 28-year trial: Conventional and Hybrid *(Xiang 67)*, conventional tillage	N: 0.15 (*v.early*) + (0.18 *v.late)* P: 38.7 x 2 K: 99.6 x 2	RS: 2.63	Xionghui *et al.* (2012)

Location	Climate (MAR, MAP)	Soil type	SOC stocks (tC/ha)	Annual Cseq (tC/ha/yr)	Depth (cm)	Duration (Years)	Crop and soil management	NPK Additions (t/ha)	Biomass incorporation (t/ha)	Reference
Southern China	1 500 mm, 17.1°C	pH: 5.5 12.7 gC/kg 2.1 gTN/kg	C: 20.8 RS: 21.3	C: -0.02 RS: 0.16		1	- 1-year trial: Early and late rice, no-till - Double rice in both trials	N: 0.15 (*v. early*) + 0.18 (*v. late*) P: 0.9 + 0.45 K: 0.9 + 0.112	RS: 4.5	
Yangtze Delta Plain, Yiangsu, Eastern China	Subtropical monsoon (1 050 mm, 15.7 °C)	CEC: 15 cmol/kg BD: 1.28 g/cm³	NA	C: 0.08 RS1: 0.94 RS2: 1.77	0–25	3	Upland crop-single rice: wheat-rice; Short flood	With urea: N: 0.45 P_2O_5: 0.06 K_2O: 0.12	RS1: 3 RS2: 6	Xiong *et al.* (2015)
				C: 0.47 RS1: 1.43 RS2: 2.42			Double rice (oil-rape-rice-Rice); Long Flood			
Cuttack, Odisha, NE India	Tropical (1 500 mm)	Sandy clay loam (Aeric Endoaquept) TC: 4.9 g/kg TN: 0.5 g/kg BD: 1.41 g/cm³	*C: 8.08 U: 8.88 RS+U: 9.42 RS+GM: 9.35	*C: - 0.41 U: 0.07 RS+U: 0.35 RS+GM: 0.22	0–15	4	- Rice variety: *Cv Gayatri* - Continuous flooding and conventional tillage	0.6 tN/ha	RS: 383 gC/kg GM: 370 gC/kg U: 0.6 tN/ha RS+U: 0.3+0.3 RS+GM: 0.3+0.3	Bhattacharyya *et al.* (2012)

*Without fertilizer nor organic amendment

+ According to the dose of urea

C: control (straw removal); RS: rice straw (incorporation); B: biochar (incorporation); GM: green manure incorporation; U: urea; CW: continuous flooding; AWD: alternate wetting and drying; BD: bulk density; NA: non available

4. Other benefits of the practice

4.1. Improvement of soil properties

Regardless of their form of application (fresh, compost or biochar) and nitrogen fertilization (with or without), applying straw residues on the first centimeters of soil helps to improve nitrogen contents, porosity, capillary porosity, and air-filled porosity (Sui *et al.*, 2016). In addition, long-term incorporation of straw increases rice root biomass (Jiang *et al.*, 2019). Soils treated with rice husk biochar generally show a higher cation exchange capacity (Wang *et al.*, 2011), and in most studies, it has been found that microbial biomass increases, with significant changes in the microbial community component (Lehmann *et al.* 2011).

4.2 Minimization of threats to soil functions

Table 65. Soil threats

Soil threats	
Nutrient imbalance and cycles	The addition of biochar increases nutrient retention and decreases its leaching. It increases fertility, especially on tropical soils (Noguera *et al.*, 2010). In addition, it is estimated that rice straw contains approximately 40 percent of the nitrogen absorbed by rice, 30 percent of phosphorus and 80 percent of potassium, and can therefore be reincorporated into the soil and can be reintroduced into soil (Chivenge *et al.*, 2020).
Soil acidification	Biochar addition increases by 6 to 12 percent in acid (4.6 pH) and neutral soils (6.2 pH) respectively (Liu *et al.*, 2012)
Soil biodiversity loss and soil compaction	Straw addition increases the amount of soil macro-aggregates because: i) its decomposition increases colloids to binding micro-aggregates to macro, and ii) increases microbial biomass and therefore the production of microbial-derived binding agents (Liu *et al.*, 2014) Long-term straw incorporation increases soil methanotrophic abundance and root size (Jiang *et al.*, 2019).
Soil water management	Biochar improves the saturated hydraulic conductivity of topsoil and xylem sap flow of the rice plant (Asai *et al.*, 2009).

4.3 Increases in production (e.g. food/fuel/feed/timber)

Benefits on productivity resulting from the incorporation to the soil of any form of organic matter varies with the temperature and rainfall of each area. This is because in hot and rainy places processes of decomposition of organic matter and the washing of salts from the soil are rapid, while the drop in temperature and rainfall slows them down.

For example, for Indian semiarid tropics rice paddy areas, each t/ha increase in SOC stock in the root zone lead to an increase of 0.16 t/ha in grain rice yield. In the semi-arid and sub-humid areas with low rainfall (average annual rainfall < 1 000 mm) and therefore poorly developed soils (Inceptisols), an increase in rice production of between 15 to 20 percent as response to additional return of rice straw about 5 to 9 t/ha/yr was noted. When the rain reaches 1 000 mm, leaching increases and with it the impoverishment of the soil. In these conditions adding organic matter means an increase of yield about 40 percent (in Vertisols) and 50 percent (in Oxisols) (Srinivasarao *et al.*, 2014).

In China, rice straw incorporation or biochar show more discreet outcomes (Liu *et al.*, 2012; Xiong *et al.*, 2015). Specifically, Huang *et al.*, 2013 in their meta-analysis show that, in general and regardless of the soil N content, crop residues retention increased rice yields by 5.2 percent as average value. The magnitude of the increase was significant at any average annual temperature but the highest increase was 7.2 percent for an area with average temperatures <10ºC and the smallest was by 3.3 percent for 10-15 ºC. Finally, the meta-analysis show that straw incorporation increases rice yield with an increase in the duration of the application (4.7 percent for 3 years and 9.7 percent for > 10 years).

4.4 Mitigation of and adaptation to climate change

In general, when rice residue has been removed, addition of biochar does not increase GHG emissions. For the management conditions analysed in China (with plots where nitrogen fertilization has been applied), incorporate low rates of biochar (about 2 t/ha) decrease CH_4 emissions while medium rates (around 10 or 30 t/ha) produce little changes. However, high rates (above 40 t/ha) increase emissions (Sui *et al.* 2016; Zhang *et al.* 2010). Regardless, experiences in field trials in a hot and humid tropic of the Philippines (Knoblauch *et al.*, 2010) show that the incorporation of large amounts of charred rice husks, only significant increased methane emissions in the first season, since in the following years it decreases rapidly.

On one hand, in the climatic and management rice conditions of China, Sui *et al.* (2016) report that biochar addition decreases CO_2 emissions only if rates of application are medium or large (above 10 t/ha in the cases analysed). On the other hand, the studies related by Liu *et al.* (2012) and Zhang *et al.* (2010) confirm that there is a significant reduction of N_2O emissions, which will be greater as more biochar is incorporated into the soil. For example, with rates of about 40 t/ha, nitrous oxide emissions fell by about half. By reducing the dose of biochar, the drop is approximately 30 percent.

Data from China indicate that rice straw management only reduces nitrous oxide and CO_2 emissions if low amounts of residues are incorporated. In the eastern (Xiong *et al.*, 2015) with rates below 6 t/ha, the recorded reductions of nitrous oxide do not exceed 10 percent but if the contribution of the straw is made for the second time in the year (double rice management), then emissions increase. As for CO_2, for the conditions analyzed by Sui *et al.* (2016) in northeast the addition of 5 t/ha of straw reduces the emissions by half compared to the control if urea is not supplied. On contrary, if nitrogen fertilizer is applied, the emissions are very similar to the control.

As to global warming, to date there is limited knowledge regarding biochar incorporation. Study conducted in northeast China (Sui *et al.*, 2016) in paddy rice fields without nitrogen fertilization indicate that straw incorporation had a global warming potential nearly 1.5 times more that of 29.6 t/ha biochar amendment. In the same vein, they also found that a large biochar application with N fertilization markedly decreased GWP when compared with rice straw incorporation, nevertheless the improvement in relation to the complete removal of the straw from the fields and the role that nitrogen incorporation plays in all this was not clear. Prendergast-Miller *et al.* (2014) attribute the positive effect of biochar to increased oxygen released by rice roots, which stimulates methanotrophs and can suppress methanogens in the rhizosphere.

If, on the contrary, the net GWP is considered, the effect of applying straw does not seem so negative. In a two-year study in the south Democratic People's Republic of Korea (Lee *et al.*, 2020) showed that the sum of methane (CH_4) and nitrous oxide (N_2O) fluxes and net ecosystem carbon budget (difference between C input and output) is greater in plots with straw removal. The reported values were 12.65 t CO_2eq/ha the first year and 6.92 the second in plots without straw; 6.02 tCO_2eq/ha the first year and 3.46 tCO_2eq/ha the second when straw is mixed with soil; and finally, 13.36 and 7.38 tCO_2eq/ha respectively where straw is left over soil. To do the GWP net balance calculation, C input source is the straw incorporated plus net primary production, and harvest C removal and heterotrophic respiration C loss are counted as C output.

4.5 Socio-economic benefits

Of the three possible alternatives for the management of rice residues, incorporation is the most beneficial alternative for the soil and the environment (as supported by the data provided in the previous sections 4.1, 4.2 and 4.3 and the later: 5.4 and 5.5). For example, in arid and semi-arid Mediterranean environments, mulch use maintains soil moisture during dry season and can be an important complement as water sheet management is implement when soil surfaces are dry.

Regarding fertilizer, the most widespread practice in paddy management is to apply inorganic NPK fertilization, at rates that can reach around 200 kgN/ha (with urea), 125 kgP_2O_5/ha and 125 kgK_2O/ha. However, it is possible that crop residues application can substitute a part of these inorganic fertilizers. In reality, rice yield in China and the Lao People's Democratic Republic is not adversely affected as a consequence of the rates reduction of inorganic N, P, and K fertilizers by averages between 8.3 and 29.4 percent. (Wang *et al.*, 2011; Asai *et al.*, 2009). In the case of tropical areas, rice straw incorporation or biochar show more limited outcomes (Liu *et al.*, 2012; Xiong *et al.*, 2015), but even so a cover without tillage is associated with a high-efficiency practice because it allows decrease tillage and inorganic fertilizers input.

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Table 66. Soil threats

Soil threats	
Nutrient imbalance and cycles	Biochar addition produces N immobilization and a decrease in plant N uptake due to the effect of C in the C/N ratio (Lehmann *et al.*, 2002; Asai *et al.*, 2009).
Soil water management	Prolonging the waterlogging time causes increase on global warming. On at same subtropical monsoon climate, the increase due to rice straw incorporation can range values from 6-12 percent by upland crop single rice to 107-180 percent by double rice in flooding conventional (Xiong *et al.* 2015).

5.2 Increases in greenhouse gas emissions

Under the most common rice growing conditions, incorporation of straw worsens forecasts on global warming with respect to the option of disposing of rice straw. However, depending on agricultural management conditions important differences can be observed in the magnitude of these increases In general, the studies realized in China by Sui *et al.* (2016) show that even when rates are low or moderate (beneath 6 t/ha/yr), to complete with nitrogen fertilization is recommended because adding only rice straw increase GWP by 60 percent (Sui *et al.*, 2016).

In general, in rice production area in Asia (China and India), methane emission increases with increasing amounts of rice straw incorporation regardless both nitrogen fertilization and water management. The increases in emissions are also biggest when rice is flooded during long periods, and the magnitude of increases can even reach 900 percent. However, the long-term addition of straw rice makes CH_4 and CO_2 emissions stabilize. The maximum values are registered after the first year of application and went on decreasing season after season during three years (Philippines, Knoblauch *et al.*, 2010; the South Democratic People's Republic of Korea, Luyima *et al.*, 2019). In this aspect, emissions in double and single rice system in Chinese rice production areas (with a straw rate of approximately 6.6 and 4.2 Mt/ha respectively) by plowing during 20 years were, on average, 48 percent lower than CH_4 emissions IPCC estimates (Jiang *et al.*, 2019).

About biochar, as seen in section 4.4, the impact of its incorporation depends largely on the agricultural practices carried out. In China (Sui, *et al.,* 2016), low rates incorporation produces higher CO_2 emissions than if the residue were completely removed Furthermore, when there is no N fertilization, methane emissions are higher than when there is, regardless of the amount of biochar incorporated. Lastly, despite the few available studies on N_2O emissions, it is indicated the possibility of a threshold value of incorporated stubble from which the emissions go from being negative increases (-30 percent) to positive (+10 percent), so more GWP is produced (Xiong *et al.*, 2015; Das and Adhya, 2014).

Table 67. GHG emissions and GWP according to residue type incorporated

Location and reference	Climate and soil type	More information	GHG emissions (t/ha/yr)				GWP (tCO_2eq/ha/yr)		Duration (Years)	Reference
			CH_4		CO_2		CH_4 + CO_2			
Shenyang, Liaoning Province, Northeast China	See Table 64	UR only; For biomass and fertilizer additions (Table 64)	*C: 0.045 *RS: 0.424 B1*: 0.102 B2*: 0.073 B3*: 0.05	C: 0.075 RS: 0.237 B1: 0.031 B2: 0.066 B3: 0.052	*C: 8.46 *RS:4,81 *B1: 8.78 *B2: 7.99 *B3: 9.28	C: 20.9 RS: 19.04 B1: 21.45 B2: 29.43 B3: 19.85	*C: 9.6 *RS: 15.4 *B1: 11.3 *B2: 9.8 *B3: 10.5	C: 22.8 RS: 25 B1: 22.2 B2: 31 B3: 21.2	2	Sui *et al.* (2016)
			CH_4		N_2O					
Tai Lake plain, Yiangsu, Eastern China		For biomass and fertilizer additions (Table 64)	C: 0.069 B1: 0.067 B2: 0.107		C: 0.002 B1: 0.0012 B2: 0.00098		NA		1	Zhang *et al.* (2010)
Wnagcheng and Changsha County, Hunan Province, Southern China		GHG emissions and GWP: CH_4 only For biomass and fertilizer additions (Table 64)	*C: 0.64 NPK: 0.49 RS+NPK:1.23				*C: 15.96 NPK: 12.16 RS+NPK:30.87		28	Xionghui *et al.* (2012)
			C: 0.21 RS: 0.32				C: 5.31 RS: 8.05		1	

Location and reference	Climate and soil type	More information	GHG emissions (t/ha/yr)		GWP ($tCO_2eq/ha/yr$)	Duration (Years)	Reference
			CH_4	N_2O	CH_4 + N_2O		
Yangtze Delta Plain, Yiangsu, Eastern China		All cases: NPK (with U) UR: wheat-rice; short flood DR: oil-rape-rice-rice; CW For biomass and fertilizer additions (Table 64)	UR and DR C: 0.104 RS1: 0.208 RS2: 0.302	UR C: 0.00226 RS1: 0.00208 RS2: 0.00219 DR C: 0.00277 RS1: 0.00313 RS2: 0.00304	UR C: 4.32 RS1: 4.6 RS2: 4.8 DR C: 5.87 RS 1: 12.2 RS 2: 16	3	Xiong *et al.* (2015)
Yangtze river, Jiangsu Province, China	Monsoon (MAT: 23.7 °C)	GHG emissions CH_4 only; rice-wheat rotation WC, CT; NPK additions (t/ha): N: 0.3; P_2O_5: 0.15; K_2O: 0.24	C: 0.12 WS: 0.66 RS: 0.13 RS+ WS: 0.64		NA	2	Hou *et al.* (2013)
			N_2O			1	Liu *et al.* (2012)

Location and reference	Climate and soil type	More information	GHG emissions (t/ha/yr)		GWP (tCO_2eq/ha/yr)	Duration (Years)	Reference
Sichuan, Hunan and Jiangxi, South China	humid to semihumid climate; acid to neutral soils	B (wheat straw 450-550 °C): B1: 20 t/ha B2: 40 t/ha Urea: 0.24(1) to 0.3 (2) t/ha	C+: 0.00146(1) - 0.00188(2) B1+: 0.00079(1) - 0.00133(2) B2+: 0.00068(1) - 0.00087(2) +according to the dose of urea				
Cuttack, Odisha, Northeast India	See table 64	For biomass and fertilizer additions (Table 64)	CH_4	N_2O	CH_4 + N_2O	4	Bhattacharyya *et al.* (2012)
			*C: 0.069 U: 0.093 RS+U: 0.115 RS+GM: 0.127	*C: 0.0002 U: 0.001 RS+U: 0.0008 RS+GM:0.00072	*C: 5.86 U: 8.08 RS+U: 9.42 RS+GM: 10.2		
Cuttack, Odisha, NE India	Tropical (MAP: 1500 mm) Sandy clay loam (Aeric Endoaquept)	Rice variety: *v IR 36* CW, CT Urea additions: 120 kgN/ha	CH_4	N_2O	CH_4 + N_2O	1	Das and Adhya (2014)
			*C: 0.11 U: 0.15 RS+U: 0.21	*C: 0.00016 U: 0.00076 RS+U: 0.00057	*C: 2.9 U: 3.96 RS + U: 5.34		

*without fertilizer or organic amendment

C: control (straw removal); RS: rice straw (incorporation); B: biochar (incorporation); WS: wheat straw (incorporation); U: urea; GM: green manure (incorporation); NT: no tillage; CT: conventional tillage; CW: continuous flooding

5.3 Conflict with other practice(s)

Rice vegetative cycle and the tillage operations carried out in each phase of the crop largely determine the GHG emission. The CH_4 flux for example, increases after rice transplantation and dropped quickly during the midseason drainage; after reflooding, it increases again to an emission peak and then decreases gradually to a negligible amount until harvest. In addition, the peak emission at the end of the reproductive stage has been observed in all fields receiving rice straw (Naser *et al.*, 2007).

5.4 Decreases in production (e.g. food/fuel/feed/timber)

A medium rate of biochar application (about 16 t/ha) increases N immobilization and so can leading to low grain yields (Asai *et al.*, 2009). The addition of straw also results in effective nitrogen immobilization in the soil, which can amount to up to 2-4 percent of the applied nitrogen (Said-Pullicino *et al.*, 2014)

5.5 Other conflicts

From an environmental point of view, there is a fundamental difference between the three possible alternatives for the management of rice residues. Burning residues in the field results in air pollution affecting nearby populated areas and high CO_2 emissions and can damage soil. Cao *et al.* (2006) estimated that in China, emissions from agriculture due to burning in field during 2000 were 100.0 Gg of black carbon and 395.8 Gg of organic carbon, contributing to the 6.8 percent and 9.8 percent respectively of total country emissions.

At this point, other economic considerations come in relation to the other two options of removing or incorporating crop residues. In fields with difficult-to-drain dammed soils, the withdrawal implies the use of machinery adapted to the soil waterlogging conditions, which greatly increases costs. In the case of transforming the stubble into biochar before returning it to the field, the cost of its subsequent incineration must be included (Srinivasarao *et al.*, 2014).

6. Recommendations before implementation of the practice

Soil water management also plays a relevant role in the potential for greenhouse gas emissions that straw residue incorporations represent (Xiong *et al.*, 2016). According to the meta-analysis on the Chinese rice paddy, a good soil and water management strategy to decrease the production of greenhouse gases is to avoid excessive C accumulation in soil, increase P availability, and decrease available of Fe content (Wang *et al.*, 2017).

In this sense, an increasingly common practice in many rice regions is to change the traditional water level management, replacing the permanent flooding (Continuous Waterlogging, CW) with the alternation of flooding with drainage until the soil is dried, or even only damp wetting (Alternate Wetting and Drying, AWD).

This system, in addition to reducing water consumption and saving on the costs associated with its handling, reduces CH_4 and in consequence the Global Warming Potential (GWP) and the intensity of greenhouse gases (Sriphirom *et al.*, 2019; Tiag et al, 2010; Xu et al, 2017). Sprinkler irrigation experiences carried out in the Mediterranean regions of Spain and Italy corroborate this trend and show very positive results (Fangueiro et al, 2017; Mejide et al, 2017)

For last, a common recommended practice for China (Huang *et al.*, 2013), is to apply inorganic N fertilizer in the early vegetative stage. SOC increase is independently by residue types (legume and non-legume) or labor type (tillage and non-tillage)

7. Potential barriers to adoption

Table 68. Potential barriers to adoption

Barrier	
Economic	See section 5.5

Table 69. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Management of Rice straw in Mediterranean wetlands, Spain	Europe	7	6	11
Conservation Agriculture in intensive rice-based cropping systems in the Eastern Gangetic Plain	Asia	5	6	12
Organic rice cultivation with internal nutrient cycling in Japanese Andosols	Asia	4, 8 and 12	6	14

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18. Selection of rice varieties adapted to salinity

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1. Description of the practice

Rice is classified as a relatively salt-sensitive crop (FAO, 1988) and several studies have estimated that rice yield decreases by 50 percent in salt-affected soil (when the electrical conductivity (EC) reaches 6.9 dS/m) (Grattan *et al.*, 2002). When the water-soluble salt content of the soil exceeds a certain threshold value, salinity degrades the soil physical and chemical properties, affecting plant growth. The effect occurs at the osmotic, oxidative and ionic levels, and results in the interruption of the metabolic functions of the cell due to ionic toxicity, as well as the appearance of psychopathologies in roots, leaf and fruit, among others (Flowers, 2004; Singh *et al.*, 2016).

Decrease in yield entails a lower biomass production and in theory a lower carbon immobilization in the soil. Faced with this situation, farmers tend to increase the inputs in chemical fertilizers and organic amendments, generally increasing Global Warming potential (GWP) (Wang *et al.,* 2012; Yuan *et al.,* 2018). A good alternative to maintain soil-plant-atmosphere balance are rice varieties that are more resistant to salinity conditions, making it one of the most important lines in the fight against the loss of rice production in areas affected by this type of land degradation. The new rice cultivars developed show a wide range of variations in salt tolerance.

Pokkali, Getu, Nona Brokra, Cherireruppu, FL478, CSR13, CSR43, PSBRC50, BRRI Dhan 54, SR86, IR65192-4B-10-13 are some of the most tolerant cultivars for white rice, whereas for black rice, Niewdam Gs.no.00621, Niewdam Gs.no.21629 and KKU-LLR-065 are the most salt-tolerant varieties. Some of these cultivars have been developed by genetic engineering, as in the case of Pokkali, a traditional variety that has been used as a donor in many breeding programs (e.g. FL478 is a cross between Pokkali and IR 29).

2. Range of applicability

Coastal wetlands in arid and semiarid environments are the main areas at risk of soil salinization due to water scarcity, poor quality water inputs or the presence of a saline water table near the soil surface. These situations cause a great concentration of salts in root area of soils and therefore can affect rice production. Although this kind of issue is mainly related to arid and semiarid areas, it can appear in other areas.

In coastal areas (deltas, plains coast, coastline, and lagoons), the interaction between freshwater and seawater intrusion can be a source of salts in soil. Farmers in general should avoid bare soil in paddy fields, because evapotranspiration can produce salts that rise from deep zones and appear on soil surface, affecting plants. Frequently, management of the sheet of water over the rice fields triggers intrusion of seawater since the loading of the sheet of water is non-existent. Due to these situations, the water table rises and causes losses in rice production, affecting it in the vegetative development. Germination stage is considered as the most tolerant period, while seedling and reproductive stages are the periods where the greatest saline stress occurs (Castillo *et al.*, 2007; Ebrahimi *et al.*, 2011).

In general, salinization is related to geographical position or climate, but land management can end up salinizing fields that initially did not present such problems. Water scarcity is one of those situations and has to do with poor quality water inputs. Irrigation time, electrical conductivity and the amount of water are factors that must be controlled by farmers. The accumulation of salts in soil profile has different treatments in rainfed agriculture if we compare with rice in flooding conditions. To avoid soil salinization and loss of rice yield, a higher dose of water irrigation can be applied to wash salts and extract them through drains from the root area.

Finally, salinization problems may be associated with soil type or original rock of rice fields. If salinity problems are defined by geological material the problem can be complicated to manage because the problem is in the origin of the soils. In addition, rice paddies are also normally found in places with low gradient or low transmissivity aquifers, so artificial underground drainage is not possible. At the end, a recurring option nowadays is the use of resistant varieties, or those genetically modified based on local species.

3. Impact on soil organic carbon stocks

General data of SOC and soil salinity can be found in several studies, but there is no study about the SOC production according to paddy varieties (resistant and non-resistant salinity).

Although no direct relationship can be evidenced, it is known that salinity affects the biogeochemical cycles that support C storage and therefore usually leads to a reduction of SOC sequestration due a low microbiological activity. For example, Morrissey *et al.* (2014) concluded that organic matter content decreased 21 percent when salinity changed from fresh to brackish (0.03-1.85 ppt) in the Chesapeake Bay in Virginia (United States of America). From this point of view, it is therefore reasonable to assume that using varieties that are less affected by soil salinity should help prevent this decline.

However, in contrast, sometimes microbial activities help to counteract this negative behavior of saline soils. Various studies conducted in coastal rice paddies have also revealed that increasing soil salinity has an inhibitory effect on organic carbon mineralization rates (Rao and Pathak, 1996; Weston, Dixon and Joye, 2006; Weston *et al.*, 2011, Vepraskas and Lindbo, 2012; Moreno-Ramón *et al.*, 2015; Luo *et al.*, 2019). In this way, this positive effect would be enhanced to the use of resistant varieties, resulting in an increase in SOC in the medium and long term.

Overall, environmental conditions and the degree of salinity of soils are the main factors that determine these behaviors.

4. Other benefits of the practice

4.1 Minimization of threats to soil functions

In South Asia (alluvial Indo-Gangetic plains Ganges delta), sodic soils can be used for rice cultivation with salt-tolerant varieties (Singh *et al.*, 2016). It is therefore possible to use degraded soils with a combining cost-effective crop and nutrient management and maximize the productivity and profitability of sodic soils.

4.2 Increases in production (e.g. food/fuel/feed/timber)

The use of salinity-tolerant varieties improve yield in rice fields. Islam and Gregorio (2013) reveals that BRRI Dhan 54 in wet season in Bangladesh showed the highest yield (6 t/ha) versus IR77674-B-25-1-2-1-3-12-4-AJY, a tolerant variety, which registered the lowest value (4 t/ha). BRRI Dhan 41 developed the highest plant development (height) in wet season, although the yield was quite low (4.2 t/ha). In addition, for a dry season BRRI dhan47 (IR63307-4B-4-3) and BINA dhan8 (IR669463R-149-1-1) can tolerate soil EC 12 to 14 dS/m during the seedling stage and EC 6 dS/m during all the cycle. In that situation, tolerant varieties obtained a yield potential between 2.8 to 8.1 t/ha.

In the same trend, Singh *et al.* (2016) concluded that CSR43 variety produced about 0.5 t/ha additional grain yields over current varieties in sodic soils located in the alluvial plains of India and in the salt-affected areas of the Ganges delta.

4.3 Mitigation of and adaptation to climate change

Currently, there are no specific data on how the implementation of salinity-resistant varieties affects greenhouse gas emissions. There is no specific study comparing tolerant and non-tolerant varieties under salinity and in the same conditions (soil, climate, management, etc.). In this sense, to determine the effect of salt-tolerant and non-tolerant rice varieties on GWP associated only with the plant and its metabolic processes it would be necessary more specific studies. Rice paddy greenhouse gas emission data exist and it is evident in the other sections of this technical document, but there is a clear lack of data in the case of the emission of tolerant varieties versus non-tolerant ones.

In general, a multitude of evidence indicates that the addition of chemical fertilizers and rice straw with traditional varieties can double the gas emissions (CH_4 and N_2O). In that regard, the farmer's aim will be to obtain the minimum incidence of toxicities or nutritional deficiencies in rice plants, in order to avoid the use of fertilizers or the incorporation of carbon (straw) because both worsen the forecasts on global warming. The magnitude of these increases can range depending climatic and management conditions. For example, in the case of rice flooding conditions with incorporation of straw and conventional tillage, emissions increase between 108 to 180 percent compared to no straw incorporation and tilling (Xiong *et al.*, 2015). On the other hand, Sui *et al.* (2016) found 60 percent more emissions with the incorporation of straw only, being around 34 percent when only urea is added (Bhattacharyya *et al.*, 2012).

The use of salinity-resistant varieties as a way of maintaining the yield of rice at optimum levels without the need to increase fertilization or the incorporation of straw, therefore seems a good option to avoid increasing the emission of greenhouse gases

5. Potential drawbacks to the practice

The use of genetic modified varieties or other salt-tolerant varieties can displace local varieties and the biodiversity can be reduced.

6. Potential barriers to adoption

Table 70. Potential barriers to adoption

Barrier	YES/NO	
Cultural	Yes	Reticence to abandon the use of traditional varieties in the area.
Economic	Yes	The increase of production costs due to the rise in the price of rice seeds of new varieties.
Knowledge	Yes	Ignorance of the most appropriate management for new varieties.

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19. Integrated rice-based farming systems

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1. Description of the practice

Rice is the staple food of about 50 percent of the global population and is cultivated on 160 million ha (Mha) mainly concentrated in Asia (China, India, Indonesia, Bangladesh, Vietnam, Thailand, Myanmar, Pakistan, Philippines, the Democratic People's Republic of Korea and Japan). Rice cultivation constitutes the vital source of income of about 140 million of rice-farming households and rural poor (Pathak, Samal and Shahid, 2018). Rice production is intricately linked with water and land ecosystems, and its intensification in the last decades led to soil, water and environmental degradations and increased greenhouse gases emissions (GHG), reducing its societal benefits (Pathak, Samal and Shahid, 2018; Kopittke *et al.*, 2019). As a response, rice-based integrated farming systems (RIFS) aim to combine rice cultivation with diverse practices such as livestock, aquaculture, agroforestry, agri-horticulture, beekeeping, mushroom, vermicomposting and/or other crops including pulses and cereals. In such systems, the synergies, mutualism and by-products generated from one component become potential inputs for others (Figure 8) (Hu *et al.*, 2016; Bashir *et al.*, 2020; Nayak *et al.*, 2020a).

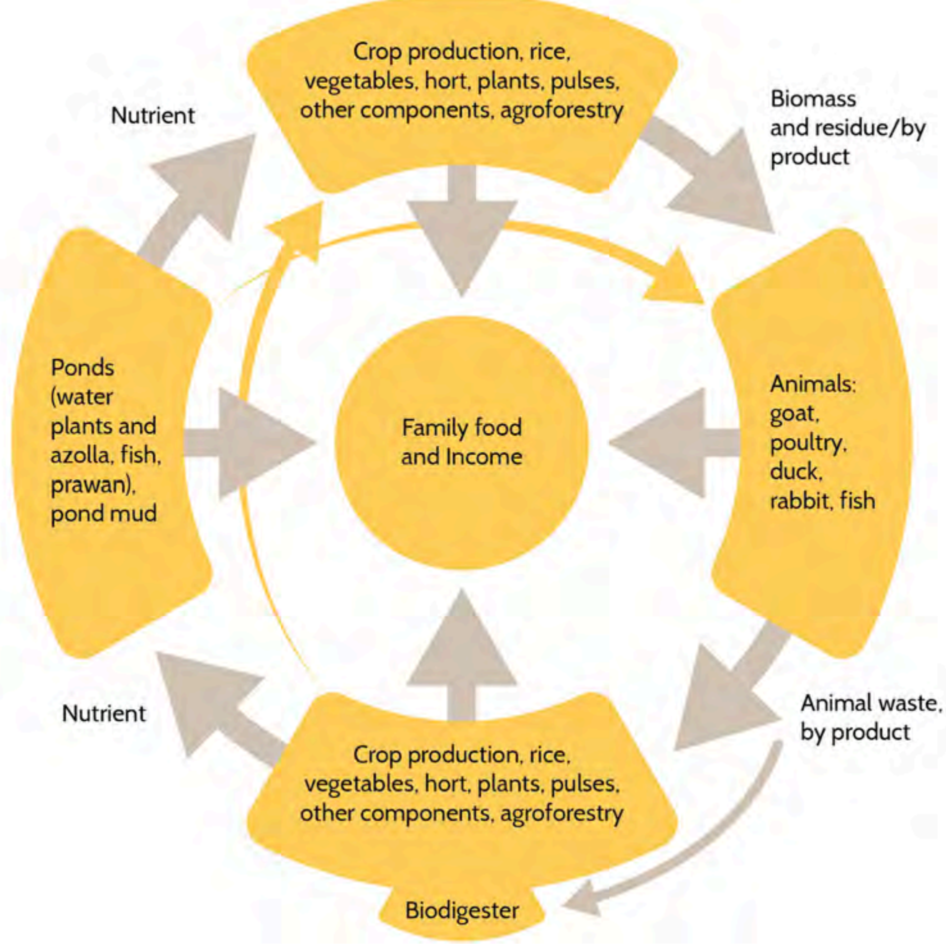

Figure 8. Pathways contributing to resource flows (i.e. from one enterprise to other) portraying their synergy and mutualisms conducive to enhance family foods and income

RIFS include a large diversity of combinations of practices that mostly depend on various factors, such as the size of land holding, the facilities of obtaining credits, potential marketing, awareness among the stakeholders including their level of training and knowledge, among others. The various combinations of RIFS are categorised under different schemes:

A. Rice-crop-animal husbandry systems

These systems include crops as cereals or pulses. In conventional rice-wheat and/or rice-pulses cropping systems, crop residues are usually dumped, thrown or burnt, leading in the long-term to soil degradation such as nutrient imbalances or the increased use of fertilizers. The inclusion of an animal component allows to make use of the crop residues that serve as animal feed, while the animals can also be used for traction. The animal waste (e.g. dung and slurries) is returned to fields leading to improvements in soil health and fertility (Bhatt *et al.*, 2016; Adarsh, Jacob, and Giffy, 2019). In between fallow periods, fodder crops can be taken up. In addition, some parts of the land (preferably the irrigated ones) can be permanently assigned for fodder and forage crop cultivation. The introduction of pulses (such as green gram (*Vigna radiata*), black gram (*Vigna mungo*), lentil

(*Lens culinaris*), horse gram (*Macrotyloma uniflorum*) or chickpea (*Cicer arietinum*), either as rotational, intercrops, relay crops or cover crops in a rice-pulse cropping system supports the improvement of soil health and provides a better feed for animals.

B. Rice-aquaculture systems

These systems support the means of livelihoods of millions of smallholder farmers worldwide (Halwart and Gupta, 2004; FAO, 2019a). Many forms of integrated rice-aquaculture systems exist (Nayak *et al.*, 2020a; Bashir *et al.*, 2020). These are categorised as:

B1. Rice-fish systems (e.g. rice-cum-fish, rice-fish–vegetables, rice-fish-livestock, Photo 22), where fish is grown together with or in alternance with rice cultivation. Depending on the regional availability and consumers demand, the main fish species used are the Indian major carp, the exotic carp, the common carp, tilapia, silver carp, minor carps, nutritionally important small fish and also crabs, shrimps, crayfishes or catfishes (Halwart and Gupta, 2004; Hu *et al.,* 2016; FAO, 2019a; Nayak *et al.*, 2020a). Besides fish, various vegetables (dyke or main fields) and animal components (livestock or poultry) can be introduced. In these systems, fish is placed in the rice fields only after the establishment of rice plants (in dry seeding rice: after attaining six inches growth of rice plants, and transplanted rice: mostly after 20 days of planting to avoid damage to rice plants); and for a period of 7-10 months or more, in the case where ridges and water refuges are created for water storage. The harvested pond water can also be used for irrigation in addition of fish culture. The application of organic manures (cow dung and fertilizer) stimulates the natural growth of fish food organisms (e.g. planktons, benthos) that support fish growth. The introduction of *Azolla* causes twin benefits, as it can be used as feed for fish and as a nitrogen source for rice.

Photo 22. Depicting an improved version of rice-aquaculture system including a livestock component. This was enabled after a land reshaping to create wide bunds (or dyke) of 2-4 m wide all around the site. The pond (or water refuge) is connected with two sides trenches. Rice cultivation covers 65 percent of the total area

B2. Rice-fish-agroforestry-horticultural-duck/poultry systems (multitier farming systems): these systems include the use of improved rice varieties, vegetables, tuber crops (A*morphophallus, Yam, Colocasia, Turmeric, Ginger*), fruit crops (e.g. papaya, coconut, arecanut, banana, guava, mango), fodder (e.g. napier, gunia grass, legume fodder, cowpea/lobia), agroforestry (e.g. *Acacia mangium*, *A. auriculiformis*, *Eucalyptus globulus*), floriculture, apiculture along with animal components (e.g. fish, prawn, poultry, duckery, goatry) and additional activities (e.g. beekeeping, mushroom cultivation) (FAO, 2019a; Nayak *et al.*, 2020a) (Figure 9).

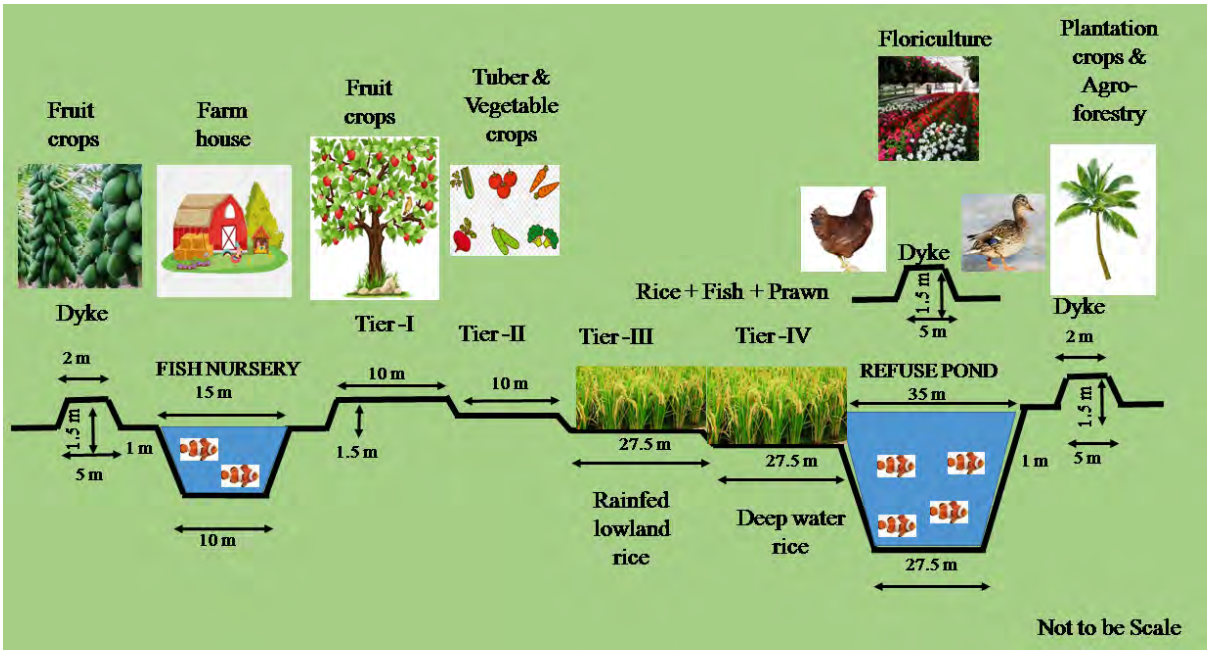

Figure 9. Multitier rice based integrated farming system. The land shaped creates the rice ecosystem of an upland (Tier I and Tier II, 15 percent of the field area), rainfed lowland (Tier III, 20 percent, up to 50 cm water depth), deep water (Tier IV, 20 percent, up to 50 – 100 cm water depth), micro-watershed (20 percent area) and raised wide bund (25 percent) surrounding the entire fields area. Different components as rice, fish and prawn, dry season crops, horticultural plants and agroforestry components need to be suitably cultivated in different tiers of land. The duck and poultry houses are constructed on the bund having projection to facilitate dropping fall directly in the pond water. The goat house is constructed on the bund using bamboo, wood and wire net with straw thatching or asbestos top. Source: Authors' own compilation and analysis

B3. Rice-cum duck farming/poultry farming (Photo 23; Figure 10, Figure 11 and Figure 12) (Nayak *et al.*, 2018; Nayak *et al.*, 2020a; Li *et al.*, 2019). The main used duck species are Khaki campbell (egg layer) or White pekin (meat type), but the use of local species more adapted to the diversity of regions is encouraged.

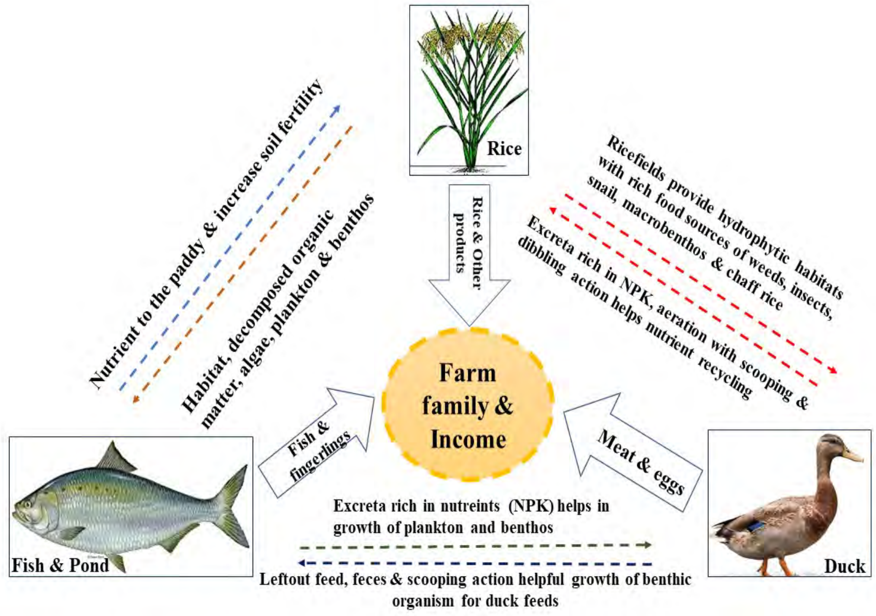

Figure 10. Symbiotic relationship between rice-fish-duck yielding maximum mutual benefits to all the entities. Source: Authors' own compilation and analysis

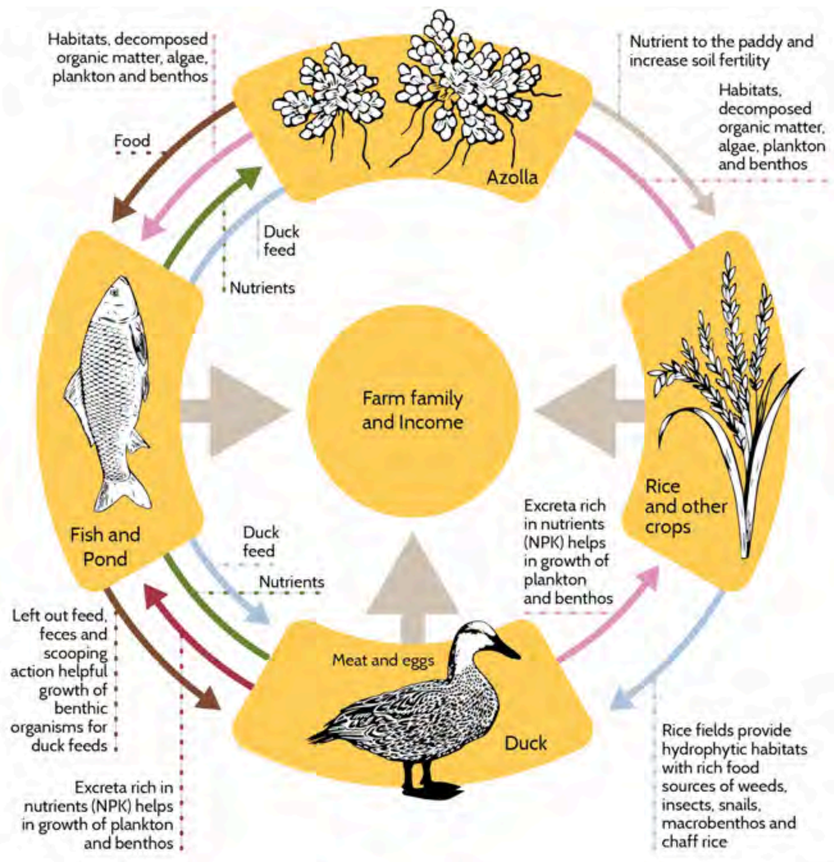

Figure 11. Framework of potential mutualism and synergies among rice-fish-duck-Azolla based co-culture. Azolla is a free-floating aquatic fern widely distributed in tropical India, and has nitrogen fixing capabilities through symbiotic cyanobionts (around 1 100 kg N/ha/year) considered as one of the best bio-fertilizer for rice plants and feed for livestock (Nayak *et al.*, 2020a)

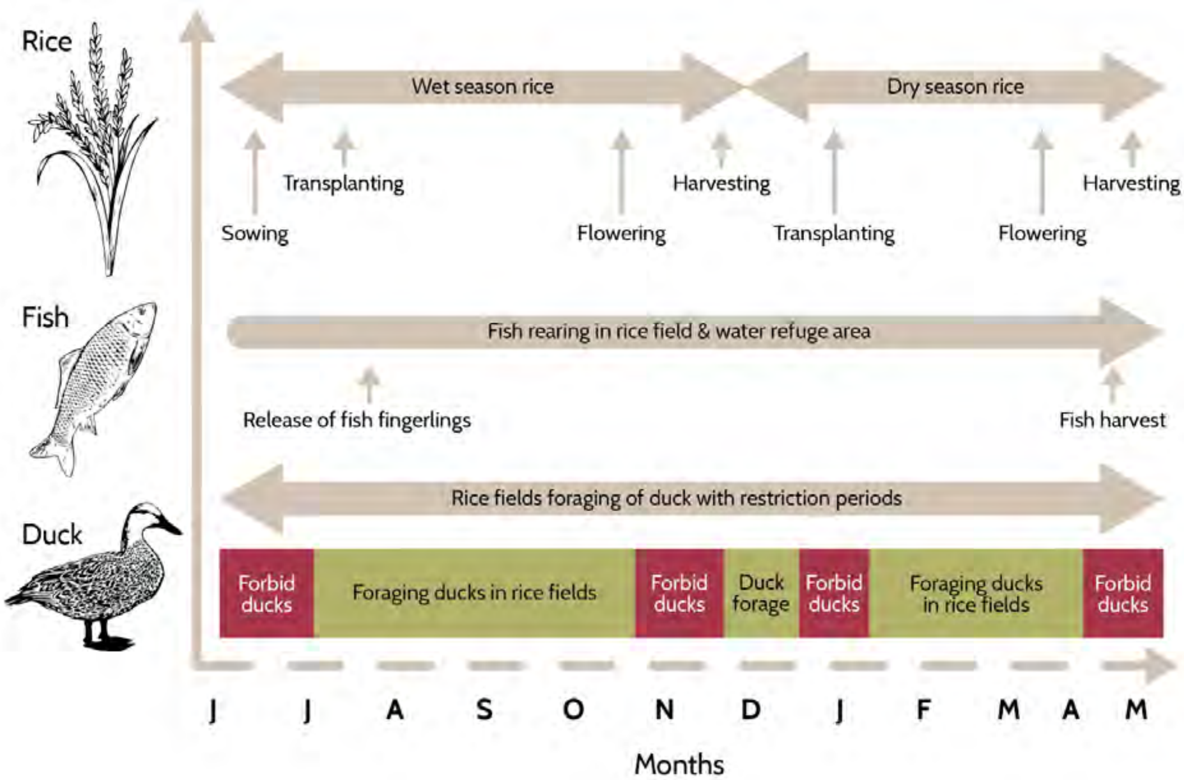

Figure 12. Pictorial views of operational periodicities of rice, fish and ducks in rice-rice cropping system. The figure indicates periods of foraging and restrictions in foraging of fish and ducks within the rice fields to protect rice plants from potential damage during foraging i.e. 20 days after rice plants establishments, during rice flowering to harvest (30 days only for ducks) in rice-rice cropping systems. Strict implementation of foraging restrictions is one of the characteristic features for rice-fish-duck co-culture technology

Photo 23. (Left) Duck foraging in dry seeded rice fields creating a conducive environment for initial (after 20 days of rice plant establishment) control of weeds and pests, duck dropping for fertilization and loosening of soil for better root growth and subsequent growth of rice plants. (Right) Rice-fish-duck co-culture system with luxuriant growth of rice

B4. Pond-dike farming systems (Photo 24): Their type mostly depends on the type of dyke planted crops (e.g. mulberry dyke-fishpond, sugar cane dyke-fishpond, banana dyke-fishpond, vegetable dyke fishpond). Input and output materials are suitably recycled, and the system energy flows are fundamentally balanced. For example, in integrated mulberry dyke-fishpond systems, the mulberry leaves are used to feed silkworms, the generated by-products from silkworm are used as feed for fish and the fertile mud (bottom of pond) fertilizes the mulberry plants at dyke. The systems can be suitably integrated with raising of vegetables, livestock, and other components (Karim and Little*,* 2018; Babu *et al.*, 2019).

Photo 24. Pond-dyke farming system

B5. Plant/fish rotations (forage or compost): where rice grows in rotation with fish. After the rice harvest (i.e. during fallow periods), fish are introduced for three to six-month. For example, in Indonesia *sawah tambak* (rice field pond brackish water) is very popular in the coastal Java region. Fish are grown as if they were a second crop *(palawija ikan system)* during the fallow-season after rice harvest, by using a hoe (water depth of 30-40 cm). In the western coastal region of India (low-lying coastal rice lands), after rice harvest (in September) rice fields are flooded with tidal water and used to raise shrimps (Halwart and Gupta, 2004).

B6. New emerging systems (e.g. rice-crayfish, rice-snail or rice-crab): In brackish and freshwater rice- rice systems shrimp cultivation (e.g. Penaeus *monodon, Penaeus merguiensis, Penaeus indicus*, *Metapenaeus ensis; Macrobrachium rosenbergii* in freshwater) are gaining importance. Among the rice-crab co-culture, freshwater crabs *(Oziotelphusa senex senex* or *Parathelphusa hydrodromus),* brackish water carb like the chinese mitten crab *(Eriocheir sinensis)* and mud crab *(Scylla serrata*, cultured throughout the world) are used. In rice-crayfish co-culture the two common species are *Procambarus clarkii* and *Cherax quadricarinatus*. During the rice harvest, the rice field water is drained or receded, to help crayfish to make burrows for shelter. After harvest, the re-growth of rice stubbles serves as food for crayfish. Further, watering rice stubbles decomposes and facilitates the growth of fish food organisms (i.e. planktons, benthos, insects, worms and mollusks) which helps the growth of crayfish (Si *et al.*, 2017; FAO, 2019a. Although the Golden apple snail (*Pomacea canaliculate*) is considered as a major pest to rice, rice-snail co-culture is used, as snail meat is used as a cheap source of feed in the culture of prawn, shrimp and rabbitfish (Visca and Palla, 2018).

2. Range of applicability

Table 71. Rice based integrated farming system prevalent in different part of the world

Type of RIFS	Sub-type of RIFS
Rice-crop-animal husbandry systems	**Rice-wheat-animal husbandry**: Applicable in South and East Asian countries, Sub Saharan Africa and South America but mostly concentrated in tropical and sub-tropical climatic conditions (Dixon, Gibbon and Gulliver, 2001).
	Rice-pulses-Animal husbandry: Prominent farming system in rainfed areas across the tropical and sub-tropical regions. The rice-pulses and animal husbandry systems are prevalent in rainfed upland and lowland areas (Dixon, Gibbon and Gulliver, 2001; Erenstein *et al.*, 2007).
Rice-aquaculture integration: Traditionally in Asian countries but adapted worldwide, even in deserts and arid lands like Egypt or Oman, thanks to the efficient use of water.	**Rice-Fish integration:** Already practised in the ancient India and China, and later adopted in most of the Asian countries (Bangladesh, Vietnam, Indonesia, Philippines, Malaysia, Thialand and Japan). These systems are mostly suitable in lowland rice area including coastal areas, where water retain in the fields even after rice harvest (Halwart and Gupta, 2004; Lu and Li, 2006; Hu *et al.*, 2016). Under plain, medium lowland and rainfed conditions, rice and fish can be grown at the same time, while in deep water and coastal lowland areas, fish either grown simultaneously with rice and/or also with off season fish rearing and seed raising (FAO, 2019a; FAO-SHOU, 2020). In hilly regions, rice-fish integration depends on suitable designs of counter bunding and water storage ponds. The traditional system of rice-fish co-culture has evolved with time, with the introduction of higher economically important aquatic species or to integrated-aquaculture and agriculture systems.
	Rice-fish-livestock-horticultural-duck/poultry multitier system: Suitable for all kinds of rice ecologies (i.e. upland, medium lowland and deep-water ecologies).

Type of RIFS	Sub-type of RIFS
	Rice-livestock-horticultural and **agroforestry-based IFS**: Mostly prevalent in Asia, Europe, South America and Africa. Suitable to medium deep or deep-water lowlands, free from heavy flooding having clay soil and with prolonged water retention capacity (Nayak *et al.*, 2020a)
	Rice-fish-duck integration: Mostly prevalent in Asia and African countries. Well adapted to medium deep- or deep-water lowland rice ecologies, free from heavy flooding and having clay soil and with prolonged water retention capacity (Pernollet *et al.,* 2015; Nayak *et al.*, 2018; Nayak *et al.*, 2020a; Li *et al.*, 2020).
	Pond-dyke farming system: Mostly prevalent in Asia, Africa and South America region (Gongfu, 1990; Babu *et al.*, 2019).
	Plant/fish rotations (forage or compost): Popular in South Asia including China, India, Indonesia and Cambodia. This type of rice-fish systems is very popular in low-laying coastal areas of Indonesia and India (Halwart and Gupta, 2004).
	New emerging systems (e.g rice-crayfish, rice-snail or rice-crab): Practised in Asia, Australia and the United States of America.

3. Impact on soil organic carbon stocks

Suitable combination of system components (crop, aquaculture, livestock, agroforestry, and horticultural components) in a rice-based integrated system contribute positively to SOC sequestration (Oliveira *et al.* 2018; Nayak *et al.* 2018; Li *et al.* 2019). Some examples of rice based integrated systems that show an enhancement of soil carbon stocks are mentioned in Table 72.

Table 72. Changes in soil organic carbon stocks reported for rice based integrated farming systems

Rice-based IFS	Location	Climate Zone	Soil type	Baseline C stocks (tC/ha)	Additional C storage (tC/ha/year)	Duration (Years)	Depth (cm)	More information	References
ICLF tree-based farming system	South Brazil 10°38'13'' S, 55°42'32'' W	Tropical Moist	Kaolinitic oxisol	Pasture = 16.5	Pasture = 1.37 ICLF = 1.91	12	0–30	IFS with trees promotes SOC accumulation, even on short periods (3 yrs), if there is no soil fertility constraint (N deficiency).	Oliveira *et al.* (2018)
ICL and ICLF	South-East Brazil 21°57'42'' S 47°50'28'' W	Tropical Moist	Oxisol	Extensive grazing = 1.45 ICL= 1.48 ICLF = 1.55	Extensive grazing = 1.68 ICL = 1.96 ICLF = 1.74	6	0–40	Land intensification increases C stocks however, converting pasture (extensive grazing) to ICL and ICLF increases soil C stocks at the rate of 0.28 MgC/ha/yr	Bieluczyk *et al.* (2020)
Rice-fish IFS	Cuttack, Odisha, India	Tropical Moist	Aeric Endoaquept sandy clay loam	Rice monoculture = 0.15	0.18	4	0–15	Increase in C stock over rice monocropping.	Nayak *et al.* (2018)
Rice duck IFS					0.23				

Rice-based IFS	Location	Climate Zone	Soil type	Baseline C stocks (tC/ha)	Additional C storage (tC/ha/year)	Duration (Years)	Depth (cm)	More information	References
Rice-fish-duck IFS					0.30			Increase in C stock over rice monocropping. Enhanced soil fertility and biodiversity	
Rice duck IFS	South China, 23⁰14'N, 113⁰38'E	Tropical Wet	Sandy loam soil	0.18	0.25	1	0–15	Rice-duck system enhanced carbon stocks	Li *et al.* (2019)
Rice-crayfish IFS	Hubei Province, China	Tropical Moist		Rice monoculture = 0.20	0.27	10	0–10	The C fractions like Microbial biomass carbon, dissolve organic carbon and particulate organic carbon are also increased	Si *et al.* (2017)
Rice-carb co-culture	Liaoning province, China 40⁰51'N, 122⁰13'E	Tropical Moist	Heavy clay of alluvial origin	0.25 (conventional rice monoculture)	conventional rice crab culture = 0.29 organic manure rice-crab culture = 0.35	5	0–20	Enhancement of bacteria contribution to SOM turnover	Yan *et al.* (2014)

ICLF: integrated crop-livestock-forestry system; ICL: integrated crop-livestock system; IFS: integrated farming system; SOC (t/ha) = SOC x BD x Depth cm x 10^{-1}, where BD, bulk density (g/cm^3), SOC (g/kg), soil organic carbon;

The SOC sequestration rate was calculated by dividing the changes/accumulation of SOC stock respect to treatment/establishment by the number of years i.e. SOC sequestration rate (t/ha/year) = SOC stock change/ accumulation/ storage (tC/ha) / nos. of year.

4. Other benefits of the practice

4.1 Improvement of soil properties

Improvements in land use managements include applying compost, conservation agriculture including cover crop, crop rotation, perennial crops, minimum or zero tillage and inclusion agroforestry and fodder crops with practices of rice-animal co-culture. These practices potentially lead to increased build up of soil organic matter, SOC and available nitrogen, phosphorus and potassium contents, which subsequently influence physico-chemical properties: for instance, in the long term, an increase of clay content of soil, possibly due to augmentation of rate of biological weathering (Bot and Benites, 2005; Teng *et al.*, 2016; Nayak *et al.*, 2018; Li *et al.*, 2019). Higher SOM reduces the bulk density and readily dispersible clay content and increases microbial activity (Gajda, Czyż and Dexter, 2016). The continuous addition of excreta of livestock components enhances SOC and available nitrogen, phosphorus and potassium content, and additionally, introduction of *Azolla* contributes substantial amount of nitrogen fertilizers for rice growth (Nayak *et al.*, 2020a) Long-term rotation of rice-shrimp farming leading to improvements in soil's physical and chemical properties including the upsurging of soil nutrients (Cai *et al.*, 2019). Duck activities enhances aeration (bioturbations). The use of leguminous crops increases available nitrogen and soil organic carbon content (Erenstein *et al.*, 2007; Adarsh, Jacob, and Giffy, 2019).

4.2 Minimization of threats to soil functions

Table 73. Soil threats

Soil threats	
Soil erosion	The diversity of practices included in RIFS reduces and protects soil from erosion, through increased cover crops areas and time periods, cultivation of perennial forage and other perennial crops, and site-specific inclusion of components as e.g. hedges, ponds, ditches, trees, agroforestry, suitable management of livestock and farm residues, addition of organic manure; management practices as e.g. terrace management, increased use of conservation agriculture and reduced tillage including living-plant windbreaks. Further, the enhancement of SOC increases soil aggregation stability which prevents erosion processes (Bots and Benites, 2005; FAO, 2019b).
Nutrient imbalance and cycles	Rice-fish, rice-duck, rice-fish-duck, crop-livestock-agroforestry and horticultural systems improve soil nutrients (NPK), enhance SOC build up and enhance nutrient recycling with augmentation of microbial diversities (Nayak *et al.*, 2018; Li *et al.*, 2019; Masciandaro *et al.*, 2018). Crops rotation and diversification, use of *Azolla*, and the addition of organic manures increase the soil available nitrogen levels and enhance the N use efficiency. In lowland ecology, rice-fish-duck co-culture leads to enhancement of total nitrogen (121 percent), available nitrogen (50 percent), available phosphorus (67 percent) and potassium (150 percent) respectively, compared to conventional rice farming (Nayak *et al.*, 2018).

Soil threats	
Soil salinization and alkalization	The maintenance of a suitable soil moisture, adequate drainage and increased SOC and aggregate stability prevent from salinization and alkalization. RIFS provisioned for increased use of organics and FYM leading to reduction in the root zone accumulation of salt (Kaledhonkar, Meena and Sharma, 2019).
Soil contamination / pollution	RIFS lead to optimized and reduced application of synthetic fertilizers, pesticides and herbicides which ultimately prevents soil and water contamination and enhanced the water quality (Long *et al.*, 2013; Nayak *et al.*, 2020a).
Soil acidification	Judicious and reduced rates of application of agricultural chemicals and high soil organic matter prevent soil acidification (Long *et al.*, 2013). Amendments like lime, organic and farmyard manure mostly used in RIFS resulting increase in pH leading to amelioration of acid sulphate soil (Halim *et al.*, 2018).
Soil biodiversity loss	Higher SOC stocks enhance soil fauna, flora and microbial population. RIFS increase soil biodiversity and biological soil quality index (SQI_{Biol}) as compared to conventional farming (Kremen, Iles and Bacon, 2012; Nayak *et al.*, 2020a).
Soil water management	The retention of available moisture and an effective water drainage system when saturated leads to higher water use efficiency (Ahmed, Ward and Saint, 2014).

4.3. Increases in production (e.g. food/fuel/feed/ timber)

RIFS increase water use efficiency and system productivity (Nayak *et al.* 2018; Li *et al.* 2019; FAO, 2019a), since it is a multi-enterprising farming system where the substantial part of food, fuel and fibre requirements are provisioned from the system itself. The potential enhancements of productivity from co-culture have been reported from many countries (Bashir *et al.*, 2020), such as China where rice with fish, turtle, crayfish and crabs produce higher average production (9.3-12.0 t/ha of rice and 1.9-2.5 t/ha of fish) (Zhang *et al.*, 2016); Bangladesh where rice-fish (shrimp, prawn, fish) yielded 3.8-5.0 t/ha rice and 1.8 t/ha of fish (Islam, Barman and Murshed-e-Jahan, 2015; Ahmed, Ward and Saint, 2014); India, rice-fish (fish, prawn, crabs) integration produces 3.0-5.0 t/ha rice and 0.7-2.0 t/ha fish (Das, Sarkar and Prasad, 2014; Nayak *et al.*, 2018; Nayak *et al.*, 2020a); Indonesia where rice-fish integration produces 6.5-7.8 t/ha rice and 0.3-0.89 t/ha fish (Dwiyana and Mendoza, 2006); Vietnam yielded 4.2-5.7 t/ha rice and 2.2 t/ha fish (Berg *et al.*, 2017); and the African continent (Melaku and Natarajan, 2019).

4.4 Mitigation of and adaptation to climate change

CH_4 is a potent greenhouse gas (28 times more than CO_2) emitted from submerged rice ecosystems due to the anaerobic degradation of organic matter by the soil microorganism (IPCC, 2007). When rice cultivation is integrated with aquatic animals (e.g. fish, shrimp, shellfish, crayfish, crab, turtle, frog and ducks), bioturbations (paddling, scooping, trampling of soil and water) and foraging activities (resulting in reduction of weeds and aquatic organisms leading to a decline in oxygen demand for their respiration) occur. These activities enhance the available dissolved oxygen levels, which leads to better soil and water aerations, and results in an acceleration of CH_4 oxidative processes (by methanotrophic bacteria) and a reduction of CH_4 emissions (Nayak *et al.*, 2018; Nayak *et al.*, 2020a; Nayak *et al.*, 2020b; Xu *et al.*, 2017; Zhang *et al.*, 2017). However, seasonal cumulative CH_4 emissions are potentially higher in co-culture systems (rice-fish, rice-duck and rice-frog) (Frei *et al.*, 2007; Datta *et al.*, 2009; Bhattacharyya *et al.*, 2013; Xu *et al.*, 2017; Fang *et al.*, 2019; Wang *et al.*, 2019); nevertheless, the GHG intensity estimated per unit output (expressed in rice equivalent yield) is much less. In rice monoculture, at some point of growing period the crop is subjected to alternate wetting and drying causing wet spell and dry spell but in rice -fish system the water remains standing throughout the growing period which explains the differential CH_4 emission. GWP potential is reduced in rice-fish-duck integrated farming, possibly, due to the reduced agri-chemicals/fertilization and better aerated environments within the paddy ecosystem (Nayak *et al.*, 2020a).

Paddy cultivation is also an important anthropogenic source of N_2O (with a global warming potential 298 times higher than CO_2). N_2O emissions depend mostly on the intensity and methods of N fertilization (synthetic and organic), water management and drainage, and temperature. In addition to submerged anaerobic conditions, microbial functions as nitrification and denitrification can potentially be disturbed (IPCC, 2014; Wu *et al.*, 2018). Significant reduction of N_2O emissions (Figure 13) from rice-animal co-culture (rice-fish, rice-crab, rice-crayfish, rice-duck, and rice-fish-duck) have been reported (Frei *et al.*, 2007; Datta *et al.*, 2009; Bhattacharyya *et al.*, 2013; Xu *et al.*, 2017; Fang *et al.*, 2019; Wang *et al.*, 2019; Nayak *et al.*, 2020a).

RIFS enhances the system's resilience and adaptive capacity and provides mutualism and flexibility to reduce trade-offs and competitiveness among the system's components, all of which offer adaptation options to overcome the vulnerability to climate-induced disturbances. In a four-year experiment on rice fish-duck co-culture, the SOC stocks doubled (+106 percent) and the GWP decreased by 11 percent compared to conventional rice farming at Cuttack, India (Nayak *et al.*, 2018; Nayak *et al.*, 2020 a). Different adaptation and mitigation strategies of RIFS are discussed in Table 74 and Figure 13.

Table 74. Different climate change adaptation and mitigation options in rice-fish based integrated farming systems

Integrated system	Component	Climate change adaptation and mitigation option	Emission potentials
Integrated Rice-fish system	Crops	• Reduction in synthetic fertilizers, pesticides and herbicides	Reduction in GHG emission
	Fish production	• Lesser feed requirements • More efficient water use • Ducks and fish bioturbation (rapid movement) and presence of *Azolla* in rice ecosystems enhance the concentration of dissolved oxygen in water, resulting in aerobic conditions, which decrease methanogens	

<table>
<tr><th>Integrated system</th><th>Component</th><th>Climate change adaptation and mitigation option</th><th>Emission potentials</th></tr>
<tr><td></td><td></td><td>bacterial activity and subsequently decreases the GHG emissions.</td><td></td></tr>
<tr><td rowspan="3">Crop-livestock-agroforestry based IFS</td><td>Crops</td><td>Organic manure used for plant growth.
Reduction of synthetic chemicals (fertilizer and pesticides), their production, transport and application.
Reduced area for feed crops with efficient land use.
Recycling of crop residue, manure and nutrient recycling.</td><td rowspan="3">Adoption and mitigation of global warming</td></tr>
<tr><td>Livestock</td><td>Quality feed for livestock (ruminants, pig, duck and poultry can eat crop residue and by-products) lower enteric methane emissions.
Sound manure management reduces GHG emissions.
Efficient use of land area management.</td></tr>
<tr><td>Agro-forestry</td><td>Inclusion of agroforestry component in RIFS significantly reduces the effects of global warming potential. Higher carbon sequestration in biomass and soil.
Improved soil health and water infiltration and retention capacity.
Fodder availability throughout the year.
Improved thermal comfort, welfare, health and production of animals</td></tr>
</table>

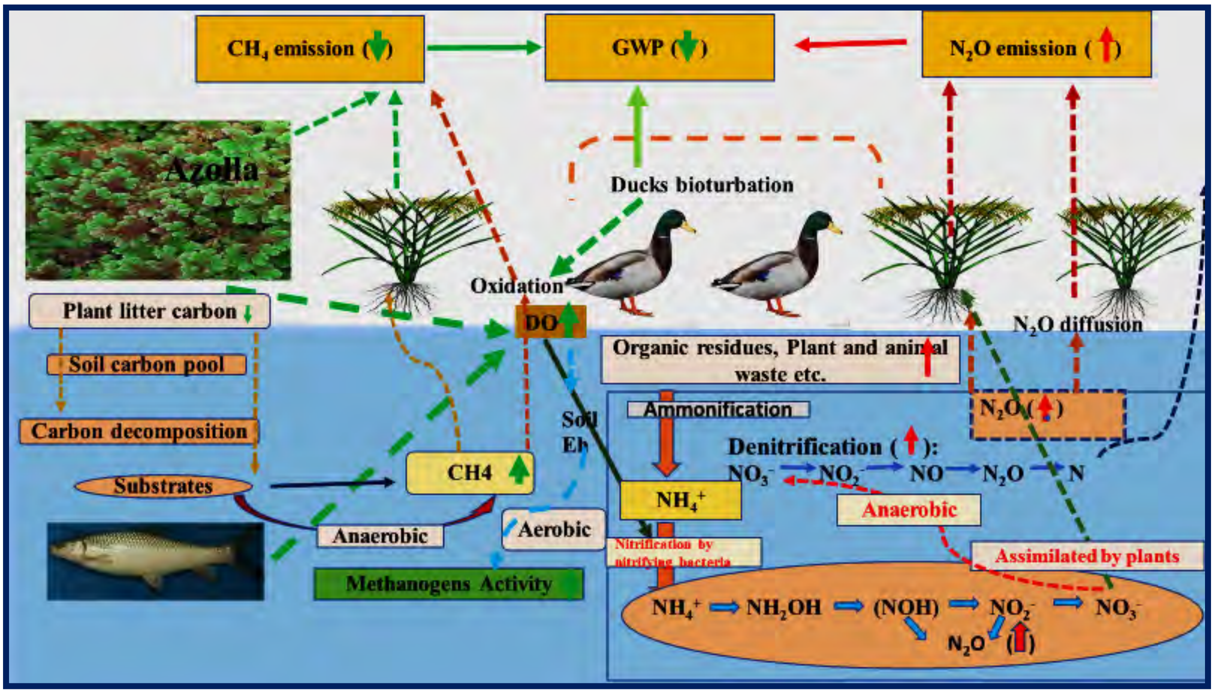

Figure 13. Schematic view of CH_4 and N_2O emissions from rice-based integrated farming systems and synergistic effects of duck and fish contributions towards reduction of global warming potentials (modified and reconstructed from Xu *et al.,* 2017 and Nayak *et al.,* 2020a)

4.5 Socio-economic benefits

RIFS offer higher socio-economic benefits leading to improved livelihoods and socio-economic standing of the rural farmers (FAO, 2019a; FAO-SHOU, 2020).

Integration with animals (fish, shrimp, ducks, etc.) supports productivity and economic returns (Bashir *et al.*, 2020; Nayak *et al.*, 2018; Nayak *et al.*, 2020a). In the Lowland ecology of India, higher economic returns (expressed in terms of benefit cost ratio) were observed in rice-ornamental fish culture (2.5), crop-livestock-agroforestry system (2.9-3.4), rice-fish-duck (2.5-2.8), rice-fish-duck-*Azolla* (2.7-3.0), multitier rice-fish horticultural system (2.0-2.5). The profit margins are mostly dependent on the integrated components and their effective managements (Nayak *et al.*, 2020a). Integrated aquaculture-agriculture (IAA) systems significantly increase farming households incomes (Ahmed, Ward and Saint, 2014), and rice-animal (rice-fish, rice-crab and rice-crayfish) co-culture plays a significant role in promoting rice ecosystem efficiency and enhancing farmers' incomes in many countries (FAO, 2019a). The rainwater harvesting model (land shaping IAA) and paddy-cum-fish culture contribute to enhanced livelihood security levels of farm families in terms of creation asset (i.e. farm pond, adoption of multiple cropping and aquaculture). It increases their resilience, productivity and income, generates employment, facilitates access to market price, extension services and institutions making them self-reliant and enhancing their social status (Kumaran *et al.*, 2020). Pond dyke integration provides better economic return as compared to rice monoculture (Karim and Little, 2018; Babu *et al.*, 2019). Eventually, the introduction of high value aquatic species generates higher economic return and profits, which in turn generates additional employment and is helpful in improving national economy.

Enhancing the quality of life (food security, balanced nutrition, employment generation and gender equity) along with preserving people's social and cultural needs is a challenging task in most developing countries. Pond and rice fields can help achieving several social benefits (Halwart and Gupta, 2004). Indeed, pond based IAA produce a year-round food production, deliver diversified healthy foods, generate employability, and addresss gender issues (women have equal level of resources access) (Halwart and Gupta, 2004; FAO, 2019a). Accumulation of farm wastes can create environmental problems, but RIFS relies on recycling of the waste generated in the system, thereby, helping in maintaining sanitation and environment safety in farm families and its surroundings. Crop-livestock-agroforestry based integrated farming generate additional employments (400-500-man days/ha/year) depending upon the extend and type of integration (Nayak *et al.*, 2020a). The two important public health vectors such as mosquitos (malaria and dengue fever) and snails (Schistosomiasis and liver cirrhosis or common liver fluke caused by *Fasciola hepatica*) used rice fields as a breeding ground which potentially caused health hazards for humans are naturally controlled by the adoption of rice-fish/animal co-culture (rice-fish, rice-duck) (Halwart and Gupta, 2004; Singh, 2011).

4.6 Other benefits of the practice

Increased water use efficiency: The rice-aquatic animal co-culture enhances blue water use efficiency with intensifying and diversifying cropping pattern (Ahmed, Ward and Saint, 2014). In rainfed areas, IAA and rice-fish-duck integrated farming with rainwater harvesting and storage facilities allow to reuse water in dyke-farming or emergency lifesaving irrigation for other crops (Ahmed, Ward and Saint, 2014; Nayak *et al.*, 2020a).

Increased nutrient recycling and biodiversity: Rice-animal (fish, duck or fish-duck) co-culture improve water quality. The addition of faecal matter, continuous movement and activities (scooping, churning and trampling of soil and water) of fish and ducks increase dissolved oxygen levels and increase the aquatic biological diversity, including planktons (phyto- and zoo-plankton), soil benthic fauna's and microbial populations (Halwart, 2008; Nayak *et al.*, 2018). The biodiversity index scoring of rice-fish, rice-fish-duck and crop-livestock-agroforestry integration system (in respect to planned vegetative richness, intensity of cropping, richness of land scape elements, microbial, plankton and benthic richness and livestock richness etc.) is significantly higher than in conventional systems (Nayak *et al.*, 2020a).

Bio-control prospecting of weed and pests: Significant reduction in weed density and weed biomass with increase in weed control efficiency was observed in rice-fish, rice-duck and rice-fish-duck integrated farming. The weed biodiversity (species richness; Simpson's index) and species diversity (Shannon-Weiner index) declined significantly with increase in Pielou evenness community index[6] in rice-fish-duck integration, signifying highly diversified weed community composition with reduction of formerly dominant weeds (Nayak *et al.*, 2020b). The presence of fish and ducks enhance bio-control efficiency of rice-insect pest (leaf roller, brown plant hopper, zig zag leaf hopper and stem borer etc.), and thus application of pesticides/herbicides can be reduced or avoided (Li *et al.*, 2019; Nayak *et al.*, 2020a).

Energy efficient system: Intensification of agriculture with extensive use of chemical fertilizers and pesticides/herbicides and large-scale use of mechanized farming operations are progressively making modern agricultural practices becoming less energy efficient. The co-culture and/or mixed farming (crop-dairy-fish-

[6] Species evenness refers to how close in numbers each species in an environment is. Mathematically it is defined as a diversity index, a measure of biodiversity which quantifies how equal the community is numerically.

poultry) is more energy efficient and uses more renewable energy as compared to conventional rice farming (Paramesh *et al.*, 2019; Nayak *et al.*, 2020a).

Conservation of natural resources: The potential reduction in application of agri-chemicals (fertilizers and pesticides lesser 24 percent and 68 percent, respectively) in China (Xie *et al.*, 2015; Long *et al.*, 2013), supports greater diversity of aquatic flora and fauna that supports resources and ecosystems conservation (Halwart, 2008). Additionally, rice ecosystems provide habitat and breeding ground for many other aquatic species (Halwart and Gupta, 2004; Halwart, 2008).

Enhancing knowledge and skills: RIFS supports the development of skills and aims to exploit the available resources (crops, livestock and genetic potentials) to make more resilient systems. Multi-enterprising systems require improved knowledge and skills in respect to their specific managements. Gender empowerment and capacity building are the main framework in RIFS and demands gender specific managements and knowledge skills (Halwart and Gupta, 2004).

5. Potential drawbacks to the practice

5.1 Trade-offs with other threats to soil functions

RIFS minimizes and prevent the soil threat, however, if not adopted and implemented properly sometimes leading to trade-offs to other soil threats.

Table 75. Soil threats

Soil threats	
Soil erosion	In heavy rainfall areas, possible risks of flooding leading to water flows-based soil erosion. In arid and semi-arid areas, there can be risks of wind-based soil erosion. Cover crops in RIFS to be suitably managed with plantation of annual and perennial crops to avoid erosion (Zhang *et al.*, 2011).
Nutrient imbalance and cycles	In case of unsustainable management, possible negative effect on depletion of SOM, nutrients (mainly N, P, K, S and micronutrients) resulting in severe limitations to nutrient available forms and Cation exchange capacity (CEC) which reduce the water and nutrient use efficiency (Nayak *et al.*, 2018; Li *et al.*, 2019).
Soil contamination / pollution	Excessive use of inputs like water and agricultural chemicals may lead to soil and water contamination and pollution which degrades soil and water quality and biodiversity (Ongley, 1996). In RIFS, any chemical (insecticides,

Soil threats	
	pesticides, herbicides, growth hormones, antibiotics) used in any component of the system, naturally reaches the other components and enhances bioconcentrations.
Soil water management	In RIFS, if soil and water are not correctly managed, there is a risk of soil erosion, leaching and waterlogging leading to soil acidification and loss of biodiversity.

5.2 Increases in greenhouse gas emissions

Adopting suitable manure management in crop-livestock integration reduces nutrient requirements (lesser import of agri-chemicals) and improves animal health and herd management (with efficient digestible feeds) ultimately, resulting in reduced GHG emissions (Soussana *et al.,* 2015; Mottet *et al.,* 2017.

5.3 Conflict with other practice(s)

RIFS needs to be properly implemented to avoid conflict with other agricultural practices:

- Selection of components, extent of integration and management practices should be carefully planned (compatibility and synergy), to avoid damage to other cultivated components of the RIFS. For example, in rice-animal (ducks) co-culture food shortage or scarcity in rice ecosystem, sometimes leads to predation and foraging of planted vegetables and other plants.
- Intensification of rice cultivation with high degree of mechanisation is affected due to implementation of RIFS.

5.4 Decreases in production (e.g. food/fuel/feed/timber)

RIFS has no negative impact on production and productivity, however, without judicious introduction of the different components, there is a risk of disturbing the balance and synergies of the system that may lead to competition between the different system's components.

5.5 Other conflicts

RIFS involving animals' co-culture may cause accumulation of pathogenic bacteria that may trigger human health hazards (Singh, 2011).

6. Recommendations before implementation of the practice

- The system's components selection must be based on the existence of mutualism and synergies. The choice of fish, livestock, poultry introduction or selection of crops must not contradict or be competitive.
- Rice-animal co-culture needs to be suitably fenced to protect animals from predation and prevent them to reach adjacent fields (may cause damage to other field crops) and escape from the field.
- Farmers need to understand the concept and management practices of each combination (crops, animals). The system should be planed before shaping land. It is necessary to understand precisely the amount of manure needed in order to improve yields, while preserving water quality and avoiding water contaminations that may be harmful.
- The implementation of RIFS initially needs higher investments, but provides labour intensive cultivation with diversified products and incomes, which in turn need special marketing skills, otherwise products may be perished and wasted. Thus, special governmental incentives might be helpful for achieving overall sustainability and environmental safety with involving farmer's active participations.
- The multi-enterprising rice based integrated system compete with the intensive mono-cropping practices as well as limited farm resources.

7. Potential barriers to adoption

Table 76. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Animals and crops can be damaged during extreme events, such as flooding, where fish, livestock and plants may sweep away or hurt, and drought and/or hot spells (water deficiency).
Cultural	Yes	Practice of some components like pig husbandry may compete with socio-economic and religious belief.
Economic	Yes	Implementation costs: The initial costs of implementation like land shaping and cost of various types of inputs (seeds, planting material, fish fingerlings), animal components (duckling, chicks, etc.) require higher investments. Lack of suitable financial incentive, availability and subsidized inputs for farmers make RIFS less attractive for small and marginal farmers. Poor access to insurance, credit and markets: Limited access of farmers to credit and finance and heavy insurance procedures and gain access to markets largely undermines the economic viability and motivation towards adoption of RIFS. Lack of suitable added-value of product and long-term storage and market transport facilities limits economic successes of the systems.
Institutional	Yes	Lack of coordination among sectors and producers: In many countries, RIFS fall under agriculture, environment and forestry departments and animal husbandry, thereby no single institutional mechanism taking a lead role in the advancement of adoption of IFS.
Legal (Right to soil and water)	Yes	Insecure tenure and small holding: Many small farmers have landless or holding very small size of land which act as major barriers towards adoption. Without formal land title, farmers are not interested to plant trees and horticultural plants.
Knowledge	Yes	RIFS include combination of different components and the production systems are knowledge intensive. This requires suitable access to information and technical support (e.g. extension services). Farmers are mostly reluctant to take risks and fear production losses. Sustained efforts are needed to strengthen extension and institutional functionaries to create adequate awareness and knowledge by organising training and demonstration for farmers/entrepreneur's motivation.
Natural resource	Yes	Fragmentation of land holding size may restrict farmer to implement profit-oriented farming systems.
Other	Yes	Theft, poaching and presence of predators can threaten the good implementation of RIFS.

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Urban soils and infrastructures

20. Management of gardens, parks, and lawns

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1. Description of the practice/concept

Urban planners, landscape architects, and other urban designers create and maintain gardens, parks, plazas, forest- and ecological -preserves, recreation areas (such as golf courses), streetscapes, and other landscapes in cities. The ground surfaces are composed of mosaics of trees, woodlands, shrubs, grasses, perennials, edibles, lawns, and waterbodies as well as paved areas, such as sidewalks, roads, and parking areas. Landscapes, including pleasure gardens, edible gardens, lawns, and park settings, are also designed and maintained on private land. Locations range from small residential lots to larger homesites and institutional properties, such as campuses and corporations. Taking this variety into account for purposes of this practice, gardens and parks are urban areas created and managed for public space, recreation, and ecological diversity. Together, gardens and parks cover a range of site scale and vegetation complexity and therefore soil extents.

2. Range of applicability

The history of development of public gardens and parks in cities all over the world varies by ecological and cultural context. Historical land-uses are drivers for the potential to accumulate soil organic carbon (SOC). The content of SOC varies widely. The highest SOC storage was recorded in wetland soils followed by forest soils and then lawn soils (Bae and Ryu, 2015). The range of accumulation is affected by local climate; geological and topographical formation; landscape typologies deployed within these features; native soil types; urban and infrastructural development; cultural attitudes; and landscape management and maintenance practices. Pouyat *et al.* (2002) compared carbon densities in different urban land types and found that low-density residential and institutional areas had 44 and 38 percent higher organic carbon densities, respectively, than soils of commercial areas.

3. Impact on soil organic carbon stocks

The potential organic carbon sequestration in urban areas varies by land use and distance from the urban core (Table 77). The quantity of SOC varies based on vegetative cover, maintenance intensity and history, and regulations for topsoil restoration (Brown, Miltner, and Cogger, 2012). Studies reporting carbon sequestration must be carefully read to distinguish the source (aboveground, root, or soil) and type (organic, inorganic, or total carbon). Eighty-three percent of carbon stocks are stored in soil, 16 percent in trees and shrubs, and 0.6 percent in herbal vegetation (Jo and McPherson, 1995). In most studies, organic carbon is reported. However, increases in soil inorganic carbon have been documented in arid regions as well along a rural-urban transect (Koerner and Klopatek, 2010).

The highest density of soil carbon is in the topsoil and in leaf litter in woodlands. Construction, excavation and mass grading, and infrastructure development in urban areas usually result in clearing of trees and removal or burial of topsoil (Logsdon, Sauer and Cambardella, 2017). In mass grading, topsoil is stripped and either stockpiled for re-spreading, where it oxidizes SOC, or sold by developers as an income source (Pouyat *et al.*, 2010). Compaction by foot and vehicle traffic results in dense soil and lower plant production. All of these factors may lead to lower soil organic matter, carbon stocks, and carbon inputs.

In more arid regions, the content of soil carbon in urban areas is often elevated compared to nearby natural landscapes. This difference is likely due to human importation of water to supplement the local climate and grow vegetation, which places more vegetative litter into the soil system (Trammel *et al.*, 2020). The greatest increases in urban soil carbon have been observed in the most highly managed soils, such as golf courses and lawns (Pataki *et al.*, 2006). Urban areas are also a source of organic matter that can be added to the soil or to containers. Urban sources include fresh and composted food waste, yard and park maintenance waste, and hay and manure from zoos and horse stables. Bio-solids and liquids are produced from disposal and recycling of human waste products. Organic matter in all forms may be used to increase the storage of carbon in the soil up to the point of equilibrium with microbial decomposition and harvesting removals. This point is reached in turfgrass after 25 to 40 years (Shi, Bowman and Rufty, 2012). Tree density in a park in Almada, Portugal, was positively correlated with total carbon stocks. Grasslands with high tree density areas and the forest had 228 and 262 tC/ha of total carbon from soil to overstory. Soil carbon was highest under forest and lawns with low tree density (Mexia *et al.*, 2018).

Table 77. Changes in soil organic carbon stocks reported for management of parks, gardens and lawns

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Years)	Depth (cm)	More information	Reference
Nebraska City, NE, United States of America	udic, mesic = humid continental	Typic Argiudoll in Soil Taxonomy; Vertic Luvisols in World Resource Base		0.52 for unirrigated fine fescue 0.74 for irrigated fine fescue 0.32 for Kentucky bluegrass 0.78 for creeping bentgrass	4	20		Qian and Follet (2002)
East Lansing, MI, United States of America	udic, mesic = Humid Continental Mild Summer	container soil	8	0.5	2	6		Getter *et al.* (2009)
Seoul, Democratic People's Republic of Korea	Tropical	Clay loam	20-140	SOC concentrations in topsoil increased 256 percent	10	100	Seoul Forest park; SOC stock dependent on land use	Bae and Ryu (2015)
Helsinki and Lahti, Finland	Boreal	Spodosol	149	0.6	100	50	10 parks of contrasting age	Setälä *et al.* (2016)
Urumqui city, NW China	Arid	Solonetz, Castanozem	55	1.25	20	80	11 urban greenspaces; SOC storage due to irrigation and fertilisation	Yan *et al.* (2016)

4. Other benefits of the practice

4.1. Improvement of soil properties

Adding organic matter to landscape and garden soils on a regular basis is a common practice to improve some of the physical and chemical properties of the soil. The additions should not exceed the long-term equilibrium organic carbon content of reference topsoil levels. Adding organic matter (which eventually decomposes into humus and other stable carbon compounds) to topsoil in sites that have lost topsoil to erosion or removal, should help in improving uptake of nutrients, providing N, P and S following mineralization, reducing the toxicity of certain metals such as Al at low pH, capturing pollutants such as Atrazine, increasing the cation exchange capacity of the soil, buffering against change in pH, lowering the density, improving soil structure, increasing infiltration and water-holding capacity, and decreasing surface evaporation losses and crusting. Adding mulch also helps to keep weeds from growing and provides long-term organic carbon and nutrient supply which eventually results in amended soil fertility and pH. Therefore, soil properties may be enhanced by adding organic residues following lawn mowing, shrub pruning, and tree-branch trimming; adding decaying perennial plant matter from the ground surface; and after composting (as practiced in Seoul Forest Park (Bae and Ryu, 2015)). Adequate garden management may decrease salinization and alkalinity.

4.2 Minimization of threats to soil functions

Properly designed and managed planting areas can increase infiltration and minimize soil exposure to wind, runoff, and erosion. For example, gardens on sloping ground should be terraced on the contour to decrease runoff and erosion. In parks and lawns, pavements can be made of pervious materials that create a firm surface for walking or driving but also allow infiltration and decrease runoff.

Table 78. Soil threats

Soil threats	
Soil erosion	Soil erosion can be prevented by adequate management; i.e. maintaining continuous soil cover.
Nutrient imbalance and cycles	Nutrient cycles may be fostered by careful fertilization practices and organic matter additions.
Soil salinization and alkalinization	Soil salinization and alkalinization may be amended through adequate garden management (see below).
Soil biodiversity loss	Soil biodiversity loss may be avoided by the establishment of gardens, which vary widely in structures that may promote plant biodiversity, such as ponds, moss, ground cover, and varied plant species (Tresch *et al.,* 2019).

Soil threats	
Soil sealing	Soil sealing will be avoided because parks and gardens generally contain few impervious surfaces.
Soil water management	Gardens, parks, and lawns allow for water infiltration, thus helping to prevent stormwater runoff.

4.3 Increases in production (e.g. food/fuel/feed/timber)

Gardens, parks, lawns, and school gardens produce significant amounts of vegetation and organic matter. Many of these are perennial plants with long-lived root systems that add organic matter beneath the soil surface. Production can be harvested and used for domestic consumption and commercial sales.

4.4 Mitigation of and adaptation to climate change

Parks, gardens, and lawns contribute to climate change mitigation through carbon sequestration in biomass as well as soil. This contribution, however, is mostly omitted in national and regional carbon stock estimates, especially in the global south, where it can be substantial. For example, in the metropolitan area of Kumasi, Ghana, a total of 1676 Gg or 66 t/ha of carbon is stored in above- and below-ground vegetation and soil of parks, gardens, and lawns (Nero *et al.,* 2017). Like other green infrastructures, parks that have high tree density are cooling islands. They thereby help to alleviate high temperatures and to increase moisture over a distance of up to 60 meters away (Grilo *et al.,* 2020).

4.5 Socio-economic benefits

Natural, semi-natural, and artificial networks of multifunctional ecological systems, gardens, parks, and lawns are extremely important for sustainable cities because they contribute to ecosystem health and human health (Tzoulas *et al.*, 2007). Pleasure, recreation, exercise, socialization, social inclusion, education, aesthetics, environmental improvement, and therapy are benefits of managed systems, such as gardens, parks, and lawns. The act of maintaining and recreating in these areas often involves groups of people that share common interests. Maintenance and management of gardens, parks, and lawns involve an ongoing learning and training process. Maintenance can be aesthetically pleasing and beneficial to the birds, insects, bees, earthworms, microbes, mammals, and other biology within an urban area. The process of designing, managing, and maintaining gardens, parks, and lawns can be a calming and peaceful way to experience outdoor areas in an otherwise stressful urban environment.

4.6 Additional benefits to the practice

The inclusion of communities is essential in decision making for the design and management of public land, ecological landscape types, and soil health that promotes carbon sequestration. By involving the public in the formation of adaptable and ecologically functional landscapes, greater education, local maintenance, and even use of these areas are enhanced by a sense of collective involvement, ownership, and responsibility.

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Runoff, erosion, and deep percolation following the application of excess chemical fertilizers can cause both surface and groundwater pollution by nitrogen, phosphorous, and potassium. Certain herbicides, pesticides, and fungicides can build up in the soil, harm humans, or harm beneficial insects, birds, bees, and earthworms. Reading and following label directions are advised to minimize threats. Turfgrass management practices, such as frequent mowing, can cause soil compaction and may require significant amounts of supplemental irrigation in warm, semi-arid and arid regions.

Table 79. Soil threats

Soil threats	
Soil erosion	Uncovered soil in gardens, parks, and lawns can be subject to erosion.
Soil salinization and alkalinization	Soil salinization is increased on golf courses and lawns that are watered with saline groundwater, especially in soils that have surface compaction, restricted subsoil permeability, and higher clay contents (Miyamoto and Chacon, 2006).
Soil contamination/pollution	Private gardening activities may result in misuse of fertilizers and other agrochemicals. Heavy metal contamination may accumulate in close proximity to vehicle traffic or where the green infrastructure is located at an old building site.
Soil acidification	Fertilizing with sulfur or sulfate products can lead to acidification of the soil. Large additions of acidic organic matter or compost can also.

Soil threats	
Soil biodiversity loss	Excess additions of herbicides, pesticides, and fungicides can lead to loss of beneficial insects and biodiversity in soils.
Soil water management	Supplemental water may be needed to maintain vegetative cover in semi-arid and arid regions

5.2 Increases in greenhouse gas emissions

Management of gardens, parks, and lawns may include trade-offs of greenhouse gas emission following their maintenance. A recent study assessed SOC sequestration and greenhouse gas emissions from turfgrass on athletic fields and ornamental lawns of urban parks down to 50 cm (Townsend-Small and Czimzik, 2010). The results showed that greenhouse gas emission from fuel use, fertilization, and irrigation outweighed SOC sequestration and that turfgrass establishment is not a greenhouse gas mitigation option (Townsend-Small and Czimzik, 2010). Use of petrol-powered machinery for garden maintenance, e.g., mowing, may have an important carbon footprint.

5.3 Conflict with other practice(s)

In most gardens, parks, and lawns, the tree density is reduced as compared to woodlands. This practice may thus compete with urban forestry.

5.4 Other conflicts

Establishment of public gardens, parks, and lawns has been prevented in some places because of the need for land to build housing for city dwellers in highly populated urban areas. Space is needed for commercial buildings, transportation, and sealed surfaces, such as pavements.

6. Recommendations before implementation of the practice

Because of the wide range of potential carbon sequestration as well as many other performance factors, an integrative design approach should be used for the creation or renovation of any park, garden, or other public green space. Consideration should be given to all of the benefits and values these areas can provide if they are designed and managed as a system. It could be good to maintain areas of park landscapes as naturalized landscapes, either through preservation of existing natural conditions or through ecological restoration practices. Such practices increase biodiversity and lower the carbon and energy footprint, or the green infrastructure, because they can be maintained with minimal or no inputs of supplemental water, chemical fertilizers, herbicides, or pesticides. Semi-natural gardens and lawns, which have a reduced management intensity, should be preferred. Reduced management intensity has a positive effect on biodiversity and leads to reduction of external inputs, thereby increasing sustainable SOC sequestration. In most cities, plant litter is removed for cleanliness and to prevent inconvenience for users. Removal of grass clippings and plant and tree leaves in these areas was found to reduce the soil organic content (Yoon *et al.*, 2016). In such cases, organic waste treatment that includes transformation into organic amendments should be performed. For example, in the Seoul Forest Park, increased SOC storage could be achieved by returning composted litter layers to soil (Bae and Ryu, 2015).

Salinization can be remedied by decreasing soil compaction, choosing soil with a lower clay content and good subsoil permeability (Miyamoto and Chacon, 2006), decreasing surface evaporation, watering infrequently with salt-free water, and ensuring free drainage after watering. Alkalization is a common problem in urban areas because of dust derived from concrete and calcareous gravel road traffic. Garden management, therefore, must aim to achieve adequate soil pH. Soil reaction (pH) can be lowered by adding organic matter or sulfate compounds and can be raised by adding lime. Gypsum can be added to improve soil structure, and dense soil can be loosened by infrequent tillage or spading and by the addition of organic matter.

Because soil in humid and subhumid climates is prone to erosion, gardens, parks, and lawns should have minimal soil exposed to wind or water erosion, except briefly during establishment, construction, or reclamation.

Risks of nitrate and phosphate losses should be avoided through adequate fertilizer use. Heavy metal contamination should be avoided by testing amendments (such as composts, biosolids, and other waste products) for contaminants before use. Atmospheric dust and other potential current contamination sources (old buildings, construction activities, etc.) should be considered when establishing a proper management plan for a garden, park, or lawn.

Foot and vehicle traffic should be concentrated in a few areas rather than across the area to be managed for plant growth. When sidewalks and pavements must be added, pervious materials should be used if possible. Soils that have been compacted by foot and vehicle traffic should be ripped or loosened to lessen the density, and organic matter should be added to promote soil structure development. Walkways between garden beds and through parks can be covered with steppingstones or layers of wood chips to minimize compaction.

7. Potential barriers to adoption

Competition for land use in urban areas is high and is regulated through zoning, ordinances, and homeowners' associations. Ownership of gardens and lawns is private and public, while parks are most likely publicly owned. Biophysical, cultural, social, economic, institutional, legal, and knowledge barriers may exist for using land for gardens, parks, and lawns in urban areas.

Table 80. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Urban soils may be impoverished and polluted, thus preventing establishment of vegetation.
Cultural	No	In general, people like to live close to green, nature-like environments. Likewise, studies have shown that creating neat edges on the perimeter of ecological landscapes can help to increase receptivity to these sites (Nassauer, 2007).
Economic	Yes	Land values are extremely high in urban areas, and conversion of privately-owned gardens and lawns into other land uses is economically advantageous. Parks require maintenance, which may be costly.
Institutional	Yes	Poor leadership, governance, and city planning may have reduced the number of gardens, parks, and lawn infrastructures. Miss-management may lead to their destruction.
Legal (Right to soil)	No	Zoning and ordinances protect gardens, parks, and lawns from alternative land uses.
Knowledge	Yes	There may be a misunderstanding or underappreciation of the importance of gardens, parks, and lawns.

Photos of the practice

Photo 25. A flower garden in Chicago, Illinois, United States of America

Photo 26. Echo Urban Park in Los Angeles, California, United States of America

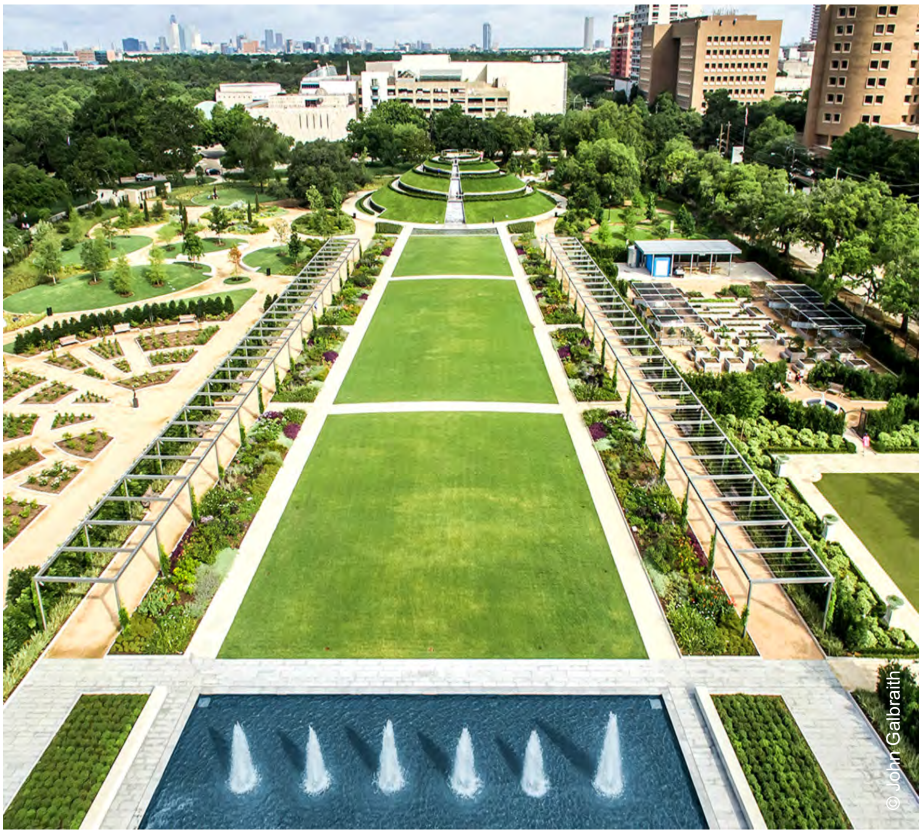

Photo 27. An urban park in Houston, Texas, United States of America

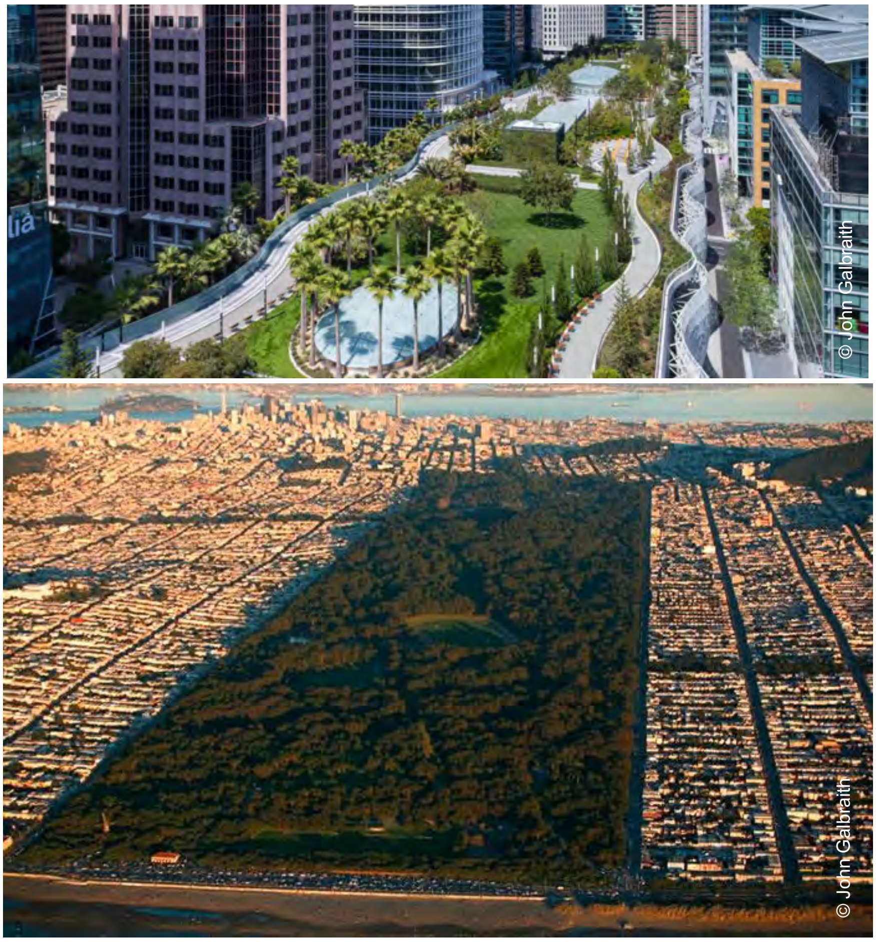

Photo 28. Two urban parks in San Francisco, California, United States of America

Table 81. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Compost application to restore post-disturbance soil health in Montgomery county, Virginia, United States	North America	4	6	28
Management of ornamental lawns and athletic fields in California, United States	North America	2, 10, 20 and 33	6	29
Water and residues management on a golf course, Nebraska, United States	North America	4	6	30

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21. Bioretention systems

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1. Description of the practice

Bioretention or biofiltration systems (e.g. bioswales, constructed stormwater facilities, and raingardens) have multiple functions in cities. Their main role is in hydrology, as the systems are intended to channel, retain, and purify rainwater. They are an alternative to conventional drainage systems composed of networks of artificial pipes and pits. They consist of vegetated channels and are designed specifically to attenuate and treat stormwater runoff from large impermeable surfaces, such as roads and parking lots (Figure 14). Their size is variable but should be at least one percent of the area being drained (EPA, 2000). They are designed to have rapidly permeable, sand-based soil media that reduces stormwater volume. Commonly, they receive organic matter amendments. Addition of labile organic matter and a submerged zone at the bottom of the facility significantly affect the removal of nutrients and heavy metals from the stormwater. Vegetation in these infrastructures includes selected species and ranges from herbal species to woody plants. Trees planted in bioretention facilities can provide many co-benefits, including shade, carbon sequestration and storage, air pollution reduction, and aesthetic improvement. However, tree planting is a challenge because the trees generally cannot tolerate a soil substrate that has high hydraulic conductivity, low nutrient availability, and toxic concentrations of pollutants. The success of tree planning depends on the type and depth of the growing substrate (Tirpak *et al.*, 2018).

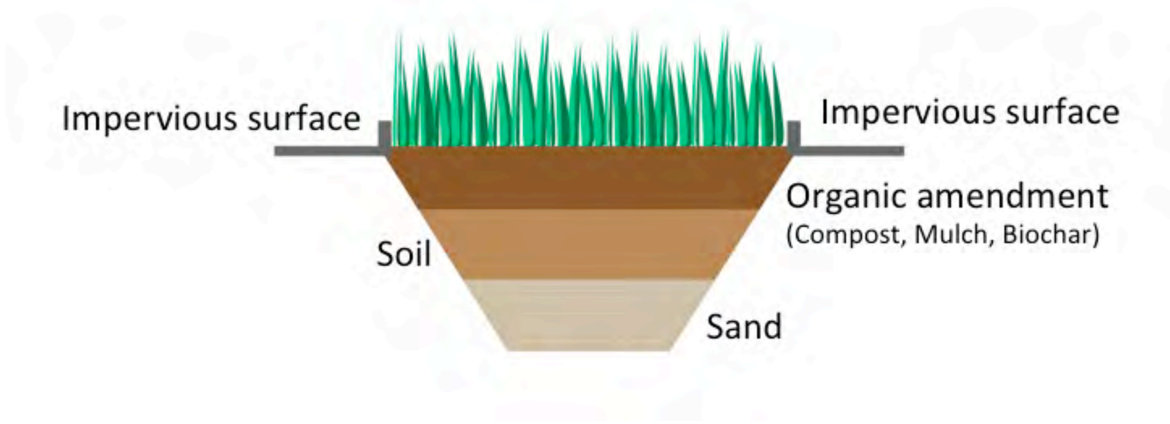

Figure 14. Schematic view of a bioretention system (adapted from https://www.idsnews.com/article/2017/10/bioswale-installation-will-help-slow-flooding)

2. Range of applicability

Bioretention systems were introduced in the 1990s as a best management practice for urban stormwater management (County, 1993). Traditionally, they were installed to reduce water volume, control erosion, and improve water quality. In recent years, their crucial role for human well-being and various other ecosystem services, such as air quality improvement, biodiversity, and carbon storage, has been documented (Prudencio and Null, 2018). These systems may be a sustainable practice to provide hydrologic ecological restoration of urban areas and may help to meet goals for downstream water quality (Liu *et al.*, 2014). Therefore, in the context of sustainability, the importance of bioretention systems is increasing in temperate regions as well as in tropical regions. The City of New York is an example for massive investment in bioretention systems, such as bioswales and raingardens (City of New York, 2010). Singapore's Active Beautiful Clean (ABC) Waters Program was implemented in 2006 and includes a wide variety of urban bioretention systems (Lim and Lu, 2016).

3. Impact on soil organic carbon stocks

Bioswales may sequester 1.5 to 9 tons of soil organic carbon (SOC) per hectare per year in soil (Bouchard *et al.*, 2013). Because they are built from various materials, including soil, their baseline SOC stock can vary (

Table 82). Soil organic carbon accumulation rates are influenced by the type of bioswale. Such engineered bioretention systems have been found to support distinct microbial communities as compared to non-engineered urban soils, suggesting that management of these urban infrastructures may influence biogeochemical cycling as well as physical properties (Gill, Lee and McGuire, 2017)

Table 82. Changes in soil organic carbon stocks reported for bioretention systems

Location	Climate zone	Soil type	Baseline SOC stock (tC/ha)	Additional SOC storage (tC/ha/yr)	Duration (Years)	Depth (cm)	More information	Reference
Southeast Queensland, Australia	Subtropical	Constructed: 4 layers (drainage, transition, filter, mulch). Filter media of sandy loam or loamy sand with pH 5.5–7.5	1	3.1	13	20	Bioretention basin with *Carex*, *Ficinia*, and *Lomandra* vegetation	Kavehei *et al.* (2019)
United States of America	Subhumid, subtropical	Felsic and crystalline soils and lower coastal plain	0.05	1	21	20	Vegetated swale with grass vegetation	Bouchard *et al.* (2013)
		Topsoil material, acidic, 58% clay	2–5	0.8	15	10	Constructed stormwater wetland with grasses, sedges, and macrophytes	Moore and Hunt (2012)
Sweden	Subarctic	Podzol	NA	0.8	17	10	Constructed wet retention pond	Merriman *et al.* (2017)
	Humid continental	Entisol	NA	0.8	26	10		
Singapore	Humid tropical	Acid soil, 20% clay	NA	1.4	15	10		

4. Other benefits of the practice

4.1. Improvement of soil properties

Biofiltration systems are a great improvement compared to conventional drainage systems, which consist of a network of pits and pipes and are mainly based on impervious materials. Biofiltration systems preserve soil from sealing and may counter impacts of urbanization (Prudencio and Null, 2018).

4.2 Minimization of threats to soil functions

Table 83. Soil threats

Soil threats	
Soil erosion	Due to the presence of vegetation, water infiltration is enhanced and soil loss is minimized.
Nutrient imbalance and cycles	The soils used for bioswale construction should be capable of retaining high amounts of phosphorus or nitrogen, which may be present in drainage water. For example, a high content of iron and aluminium in the soil may help retain phosphorus, and mulch may help retain nitrogen.
Soil contamination / pollution	The systems are designed to clean stormwater. They may, therefore, prevent downstream soil pollution.
Soil sealing	Bioretention systems prevent soil sealing and are, therefore, preferable to other stormwater management practices.
Soil water management	Bioretention systems have a positive effect on stormwater infiltration. Moreover, they improve water quality through retention of pollutants. They have higher removal efficiency for particulate pollutants than for dissolved pollutants (Boger *et al.*, 2018). Particulate pollutants, including Pb, Zn, and PAH, were very efficiently removed (around 90 percent) by biofiltration swales (Flanagan *et al.*, 2018). Removal efficiency was low in winter.

4.3 Increases in production (e.g. food/fuel/feed/timber)

Due to their limited size, biofiltration systems do not affect production.

4.4 Mitigation of and adaptation to climate change

Biofiltration systems may improve climate change adaptation of cities because the systems attenuate stormwater flow, which is an effect of extreme events (Prudencio and Null, 2018).

4.5 Socio-economic benefits

Biofiltration system installation is a low-cost multifunctional technology with relatively simple and quick implementation. Building a system using organic and mineral-waste material from within a city encourages recycling and a circular economy.

4.6 Additional benefits to the practice

A biofiltration system may provide cultural services, such as landscape improvement and education opportunities.

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Table 84. Soil threats

Soil threats	
Nutrient imbalance and cycles	Bioswale establishment may lead to infiltration of high loads of nitrogen and phosphorus during stormwater events, especially if organic amendments were used.
Soil contamination / pollution	Pollutants may be released from artificial biofilter components if the components were not chosen adequately (Flanagan *et al.*, 2018).
Soil biodiversity loss	Constructed stormwater ponds may have reduced biodiversity as compared to natural ponds (Moore and Hunt, 2012).

5.2 Possible greenhouse gas (GHG) emissions

N_2O and CH_4 emissions from bioretention basins were found to be low, while CO_2 emissions from bioretention basins may increase with increasing soil organic matter content (McPhillips, Goodale and Walter, 2018). Greenhouse gas emissions are driven by hydrologic conditions. Wet basins have higher CH_4 and N_2O emissions than dry basins (McPhillips and Walter, 2015). Nitrification and denitrification processes can be promoted by introducing a low-permeability layer below a higher-permeability layer in the bioretention soil media (Hsieh, Davis and Needelman, 2007).

5.3 Conflict with other practice(s)

Bioretention facilities may conflict with conventional stormwater management strategies that are based on networks of impervious pits and pipes.

5.4 Decreases in production (e.g. food/fuel/feed/timber)

The material used for building bioretention facilities may have adverse effects on plant growth.

5.5 Other conflicts

In tropical and subtropical climates, bioretention facilities can favor mosquito proliferation and thus transmission of epidemic diseases, such as dengue-fever, zika virus disease, chikungunya, and malaria (Batalini de Macedo *et al.*, 2017).

6. Recommendations before implementation of the practice

The presence of vegetation was found to be more controlling than clay content for SOC accumulation in retention systems (Merriman *et al.*, 2017). The establishment of vegetation, therefore, is necessary in these systems. Littoral-shelf surface coverage around the perimeter of constructed wetland ponds should exceed the recommended 1- to 3-meter-wide swaths (Merriman *et al.*, 2017). Suitable shallow water levels can be achieved by an adjustable outlet structure and proper maintenance (Hunt *et al.*, 2011). Choice of vegetation is important. For example, plants that can use N and P effectively should be used to enhance water purification. In dry areas, local, drought-resistant species should be chosen. The tolerance of tree species to environmental conditions can differ widely; therefore, choice of trees should be adapted to soil type, sun exposure, soil moisture regime, soil pH, and climate (Tirpak *et al.*, 2018). Use of native plants should be encouraged. They were found to effectively remove nutrients from urban stormwater (Lim and Liu, 2016).

Organic matter additions in the form of compost should be limited to the amount necessary for plant growth (recommended <15 percent of volume) because a high content of organic matter leads to loss of excess P and N and thereby increases risks for water pollution and greenhouse gas emissions. If organic materials are added, materials that have C/N >20 and a low P content should be chosen. Such materials minimize excess nutrient availability and greenhouse gas production (McPhillips, Goodale and Walter, 2018). Using biochar in bioretention systems may greatly improve water purification through effective retention of organic carbon, total nitrogen, nitrates, and total dissolved phosphorus (>60 percent) and trace organic contaminants (>99 percent) (Ulrich, Loehnert and Higgins, 2017). Biodiversity in bioretention systems should be encouraged. It prevents soil erosion and is beneficial for water infiltration (Batalini de Macedo *et al.*, 2017). Biodiverse systems also increase below-ground root carbon input and SOC sequestration (Lange *et al.*, 2015). Inoculation of bioretention systems with earthworms may benefit water infiltration and therefore enhance the function of the system in terms of stormwater attenuation.

Bioretention systems require regular maintenance to prevent surface clogging and to increase the amount and quality of stormwater runoff. The maintenance is mainly necessary to retain functioning and diverse vegetation cover (Batalini de Macedo *et al.*, 2017). To prevent the spread of disease in tropical and subtropical environments during the rainy season, weekly maintenance is required to deplete water in possible accumulation places (Erickson, Weiss and Gulliver, 2013).

7. Potential barriers to adoption

Table 85. Potential barriers to adoption

Barrier	Yes/No	
Biophysical	No	As these are artificial structures, biophysical barriers other than lack of space can be overcome.
Cultural	No	People generally favor vegetated spaces in cities (Kim and An, 2017).
Social	Yes	Public acceptance can be a barrier due to dirty appearance caused by plant litter and other organic residues (Dobbie, 2016).
Institutional	Yes	Some cities have strong stormwater management performance standards that may not be met by bioretention systems.
Legal (Right to soil)	Yes	Little regulatory control exists for design, construction, operation, and maintenance (Ashley *et al.*, 2015).
Knowledge	Yes	For optimal efficiency, technical knowledge is needed (Batalini de Macedo *et al.*, 2017).
Other	Yes	Perception that bioretention systems have higher maintenance costs than conventional stormwater retention systems can be a barrier; although this was found to be wrong (Houle *et al.*, 2013).

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22. Green roofs

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1. Description of the practice

A green roof is a roof that is partly or completely covered by plants and has an impermeable layer underneath. A green roof has an anti-root membrane and waterproofing, drainage, and water retention layers covered by growth substrate (Lata *et al.*, 2018). Mineral as well as organic materials are used as growth substrates for green roof construction. Green roofs may be managed intensively or extensively. Extensive green roofs are lightweight and support herbal vegetation on a shallow growing substrate that requires minimal maintenance. Intensive green roofs support heavier weight and thus a deeper layer of growing substrate and a wider variety of plant types, including herbaceous species and shrubs (Figure 15, Besir and Cuce, 2018).

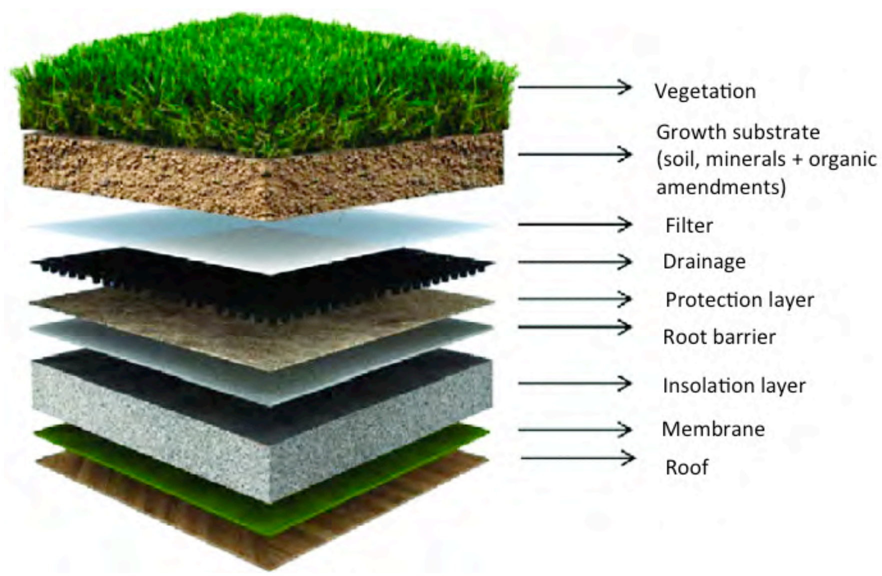

Figure 15. Schematic view of green roof composition. Modified from Besir and Cuce (2018)

2. Range of applicability

Green roofs are used for limiting rainfall run-off, for aesthetics, and for thermal protection (Lata *et al.*, 2018). Green roofs may also be installed as a sustainable strategy to improve urban water availability through harvesting and purification of rainwater (Semeraro, Aretano and Pomes, 2019). In northern countries that have high rainfall, grass species may be used for green roof establishment. In drier areas (e.g. deserts), roofs are covered with biological soil crusts (biocrusts), which are composed of a complex mosaic of cyanobacteria, green algae, lichens, mosses, microfungi, and other bacteria/archaea. These biocrusts can photosynthesize when water is available, but their entire metabolism ceases in drought conditions (Paço *et al.*, 2014). Biocrusts can remain under these conditions for long periods and return to their normal functions after rain or dew events. They could be a solution for green roofs in these harsh conditions. Although governments all over the world have been promoting the establishment of extensive green roofs, the practice is poorly adopted in many cities (Paço *et al.*, 2014). In addition to improving water management, green roofs may enhance wildlife in urban areas. They were, for instance, found to support diverse wild bees (Tonietto *et al.,* 2011). Establishment of green roofs with vegetable crops has been suggested recently as a possibility to improve agricultural sustainability and food security in urban areas (Walters and Midden, 2018).

3. Impact on soil organic carbon stocks

Substrates used to establish green roofs may already contain organic carbon, especially if organic materials are included. Additional soil organic carbon (SOC) storage may depend on the vegetation, climate, and management practices. Intensively managed green roofs may store more SOC than extensively managed green roofs but may also require more external inputs in terms of irrigation, fertilisation, and other maintenance.

Results from studies of extensive green roofs in the United States of America indicated varying carbon storage in biomass depending on species. The average was 1.7 tC/ha. Substrate carbon storage averaged 9.1 tC/ha without species effect, representing a SOC sequestration 1.0 tC/ha/yr (Table 86, Getter *et al.*, 2009). Higher carbon storage can be achieved with the use of organic waste materials. A one-year study from China using sewage sludge for green roof construction indicated carbon storage of 18 tC/ha with an average carbon sequestration of 64 tC/ha/yr in the combined biomass and substrate organic matter (Luo *et al.*, 2015).

Table 86. Changes in soil organic carbon stocks reported for green roofs

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration	Depth (cm)	More information	Reference
East Lansing MI, United States of America	Temperate	XeroFolr XF-105 drainage layer and greenroof substrate (79% sand, pH 8.2)	8	1	2 years	6	*Sedum* vegetation (*S. acre, S. album, S. kamtschaticum, S.spurium*)	Getter *et al.* (2009)
			31	15–208	3 years	10.2	3 types of vegetation with contrasting additional C storage: Sedum (15 t/ha) Native prairie mix (18 t/ha) Herbaceous perennials and grasses (208 t/ha)	Wittinghill and Rowe (2014)
Berlin, Germany	Temperate	-	-	0.8	1 year	9	Sedum	Heusinger and Weber (2017)
DuJiangyan, China	Subtropical, humid	Local natural soil (pH 6,3, 1,2%C)	38	3.89	1 year	25	3 plant species (*Ligustrum vicaryi, Liriope spicata, Nephrolepis auriculata*)	Luo *et al.* (2015)
		Local natural soil amended with sewage sludge (1:1; v:v)	159	3.81				
Murcia, Spain	Semi-arid	Mixture compost/silica sand/crushed bricks (1:1:1.8; v:v:v)	3.9	0.8	10 months	10	SOC increase only under *Lotus creticus* L.; *Asteriscus maritimus* L. showed SOC loss	Ondoño, Martínez-Sánchez and Moreno (2016)
		Compost/crushed bricks (1:9, v:v)	10.9	-5.2			2 plant species (*Lotus creticus* L. and *Asteriscus maritimus* L.)	
		Compost/Haplic calcisol/crushed bricks (1:1:1.8; v:v:v)	7.6	1.1				
		Compost/silica sand/Haplic calcisol (1:1:1.8; v:v:v)	9.7	2.3				

4. Other benefits of the practice

4.1. Improvement of soil properties

Green roofs are constructed entities, and their properties are strongly dependent on the organic and mineral materials used (Dusza *et al.*, 2016). Soil properties are therefore not "improved" by this practice; rather, the establishment of green roofs adds soil functions to areas that previously supported impervious surfaces.

4.2 Minimization of threats to soil functions

Table 87. Soil threats

Soil threats	
Soil water management	Green roofs have positive effects on stormwater management and water purification (Shafique *et al.*, 2018) as well as on rainwater recovery, which is of particular importance in urban areas with impermeable pavements (Semeraro, Aretano and Pomes, 2019)

4.3 Increases in production (e.g. food/fuel/feed/timber)

Green roofs may have the potential to support urban agriculture. However, before the practice can be incorporated at scale, installation costs must be reduced, roof-weight limitations must be addressed, and appropriate management practices must be developed (Whittinghill and Rowe, 2012). Extensive green roof systems can support high productivity of lettuce, kale, and radish if sufficient moisture and nutrient inputs are available (Walters and Midden, 2018).

4.4 Mitigation of and adaptation to climate change

Green roofs alleviate the huge ecological footprint of cities (Getter *et al.*, 2009). This function is due to their isolating impact—reducing energy needed for cooling and heating—and to the creation of sustainable buildings. In tropical China, Hong Kong SAR, green roofs prevented 43.9 TJ of solar energy from penetrating buildings in summer (Tsang and Kim, 2011). In Italy, green roofs are used to isolate wine cellars (Conti, Barbari, and Monti, 2016). Moreover, green roofs may contribute to carbon removal from the atmosphere due to their great organic carbon sequestration potential in growth substrates. Nitrogen fixation from the atmosphere (both through free-living organisms in the substrate and symbiotic N-fixing organisms) must be ensured to avoid the need for fertilisation of the plants used.

4.5 Socio-economic benefits

Rainwater retention by green roofs reduces the urban heat-island effect in buildings (which results from an increase in temperature in urban environments with reduced vegetation cover); helps mitigate heat loss during the winter; increases urban biodiversity and carbon sequestration; improves air quality, soundproofing, and durability of buildings; and potentially slows the spread of fires. Green roofs therefore greatly reduce the footprint of cities through reduction of heating and cooling consumption while also contributing to rainwater retention (Karteris *et al.*, 2016).

4.6 Additional benefits to the practice

Green roofs provide cultural services by supplying or improving aesthetics and may have a positive effect on the psychology of urban dwellers (Lata *et al.*, 2018).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Table 88. Soil threats

Soil threats	
Soil contamination / pollution	Organic building materials in high concentrations (e.g. composts) could lead to water pollution due to P and N leaching (Wang, Tian and Zhao, 2017).

5.2 Increases in greenhouse gas emissions

The carbon footprint due to construction and maintenance has been estimated to be between 6.4 and 155.8 kg CO_2 eq/m^2, depending on material used and management intensity. The lowest carbon footprint (6.4 kg CO_2 eq/m^2) was found for a lightweight green roof (Chenani, Lehvävirta and Häkkinen, 2015). Green roofs may be effective CH_4 sinks and do not significantly affect N_2O emissions (Teemusk *et al.*, 2019), but the number of studies concerning this issue is still very limited.

5.3 Conflict with other practice(s)

Green roofs replace traditional roofs and may thus create conflicts with other actors in the building industry.

5.4 Other conflicts

Green roofs have a carbon footprint due to the manufacturing process. The carbon "cost" of installing a root barrier, drainage layer, substrate, and plant material was assumed to be similar to the gravel ballast of a traditional roof (Kosareo and Ries, 2007). The use of permanent green roofs in the Mediterranean regions, which have hot and dry summers, requires the use of irrigation. The vegetation cannot survive summer without water because the installed vegetation is not native to the climatic region. Even species that are suited to the climate require irrigation to maintain growth and aesthetic appearance during the dry period. Green roofs in this context thus also have a water footprint.

6. Recommendations before implementation of the practice

Provision of ecosystem services of green roofs is strongly dependent on the growth substrates used (Lata *et al.*, 2018). Adequate substrate for green roof installation needs to find a compromise between an average pore size that allows for aeration of plant roots and a water holding capacity that minimizes the need for irrigation (Ondoño, Martínez-Sánchez and Moreno, 2016). Typical inorganic building materials are light, porous materials, such as volcanic rock (pumic or pozzolan), expanded clay, expanded shale, or a mixture of these elements (Lata *et al.*, 2018). However, artificial substrates are more vulnerable in terms of water loss than natural soil because they dry faster (Dusza *et al.,* 2016). An organic component of the growth substrate is needed to provide nutrients and improve physical conditions for plant growth. Although content of organic matter in green roofs may vary due to national regulations, an initial organic matter content between 10 and 20 percent seems to be optimal for plant growth (Ondoño, Martínez-Sánchez and Moreno, 2016). In the spirit of circular economy, local waste materials, such as compost and municipal sewage sludge, may be used for green roof construction. Biochar could be included in green roof substrate due to its beneficial effects on amending water quality and quantity of runoff (Qianqian *et al.*, 2019). Choice of plant species may be important as it can greatly influence the carbon sequestration potential of the roofs (Ondoño, Martínez-Sánchez and Moreno, 2016). While Sedum species are often used, SOC sequestration potential was found to be higher with thicker substrate layers when herbaceous species were included (Wittinghill and Rowe, 2012). However, this may also decrease water quality due to higher nitrogen and carbon leaching (Seidl *et al.*, 2013) and increase the maintenance effort and cost.

There are three different management systems for green roofs (Lata *et al.*, 2018):

- Extensive, without harvesting and other management activities (succulent plants, such as Sedum; light substrate; 4–15 cm thick; no irrigation)
- Semi-intensive (grasses or shrubs; light substrate; 12–30 cm thick; irrigation)
- Intensive (all plant types; natural soil; >30 cm thick; irrigation)

Moss- dominated biocrusts appear to be an innovative solution to urban landscape. These taxa do not have a root system, thereby reducing the thickness of the substrate, decreasing the installation costs, and decreasing the weight load on the building structure. The resulting product of this biological technology applied to urban landscape was designated by Paço *et al.* (2014), as MedMossRoofs (https://www.facebook.com/medmossroofs).

7. Potential barriers to adoption

Table 89. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Climatic conditions in arid areas or boreal areas may impede green roof installation.
Social	Yes	Lack of awareness among public and private sectors on the benefits of green roofs (Hossain *et al.*, 2019); difficulty of acceptance from construction industry sector; negative visual appreciation of roofs when plants are too diverse or during winter.
Economic	Yes	Increased maintenance costs.
Institutional	Yes	Lack of promotion and incentives from the government; lack of guidelines (Hossain *et al.*, 2019).
Legal (Right to soil)	Yes	A planning permit may be necessary. Rooftop structures could be prohibited.
Knowledge	Yes	Many knowledge gaps exist concerning the effect of green roofs on ecosystem services related to biodiversity (e.g., pollination) and air quality (Lata *et al.*, 2018).

Table 90. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Urban agriculture on rooftops in Paris, France - the T4P research project (Pilot Project of Parisian Productive Rooftops)	Europe	5	6	23

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23. Urban agriculture

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1. Description of the practice

Urban agriculture comprises all forms of agricultural production within or around (peri-urban) cities (Wagstaff and Wortman, 2013). It consists of growing, harvesting, processing, and distributing fresh vegetables and fruits in urban areas for personal consumption or for sale. Urban agriculture may involve urban dwellers who cultivate food in private yards, allotment gardens, common-area (community) gardens, containers, and raised beds or inside hoop houses and greenhouses. It also includes rooftop farming and urban farms run by private companies. According to FAO, urban agriculture refers to plant and animal production in an urban context, including not only small vegetable gardens but also farming activity on community land (FAO, 2010). Urban agriculture is substantial: 15 to 20 percent of the world's food is grown in urban areas (Gerster-Bentaya, 2013). In the United States, according to the 1990 Census, urban areas produced 40 percent of the dollar value of U.S. agricultural products (Deelstra and Girardet, 2000).

2. Range of applicability

The production of food in urban areas addresses food-insecurity issues of a growing population in global megacities. Urban agriculture is practiced by 800 million people and produces 15 to 20 percent of the world's food. The dependence on urban agriculture for food security is growing, especially in developing countries (Gallaher *et al.*, 2013; Karanja and Njenga, 2011). Supplementing food production by city farming is not new. It has been especially common during wars and depressions, when food scarcity was an issue (Burkhardt and Schneider, 2018). Nowadays, more city farming is practiced as a leisure time activity, but it is also practiced to improve personal health and economic situation. It improves personal and community access to food, creates jobs and higher incomes, improves the appearance and cohesiveness of the community, educates about gardening and farming, and provides ecosystem services (Benth, *et al.* 2013). In some parts of Europe, the concepts of urban and peri-urban agriculture (UPA) are still a recent novelty in academic and political agendas. Regulated urban gardens, such as the regulated Municipal Urban Parks, started to emerge in 2008, largely due to a deep economic crisis. In a study conducted by Cordovil, Rodrigo and Gonçalves (2015), however, urban farming proved not to be related to unemployment. It was related to pleasure or passion for agriculture and working with the land. It was also related to a need to alter consumption habits towards the use of organic products and zero use of fertilizers. Most urban farmers have a previous link to rural areas, or have a history of farmers in their family, or both.

In many urban areas, soils and organic-growth media may be unfavourable for plant growth because of low physical quality and chemical fertility (Jim, 1998). Urban soils that were less than 20 years old in Pullman, Washington, United States of America, and Moscow, Idaho, United States of America, had lower soil quality and soil health than non-urban soils. These soils tended to be more compacted than natural soils, have a higher content of clay, and have less nitrogen mineralization, biological activity, and organic matter. However, urban soils that were 50 years old had reduced soil bulk densities, increased biological activity, and increased soil organic matter (Scharenbroch, Lloyd and Johnson-Maynard, 2005). Urban soils generally have a higher content of heavy metals (Burt *et al.*, 2014), pollutants, and artifacts than surrounding non-urban soils; however, their properties are quite variable (Pouyat *et al.*, 2002). Urban soils have higher pH and higher temperatures than comparable land uses in rural areas (Brown, Miltner and Cogger, 2012). Food production in urban agriculture often utilizes and recycles compost, bio-solids, and bio-liquids produced by human settlements (Lal, 2017). It also imports large quantities of topsoil materials from agricultural lands or forestlands. Urban farming often relies on specific engineered systems and techniques, such as “lasagne” gardening (Lanza, 1999) that consist of different layers of organic materials. Rooftop farming is practiced in many urban areas because of limited yard and garden space, better exposure to sunlight, and better ability to protect food from theft and predation.

3. Impact on soil organic carbon stocks

Agricultural soils have high plant production but are bare for large parts of the year, which could have contrasting effects on SOC sequestration. Many urban agriculture managers, such as gardeners, regularly add organic matter to improve their soil. Some of these soils could have elevated carbon storage compared to rural agricultural soils found in the same region (Edmondson *et al.*, 2014). Urban areas are a source of organic matter that can be added to the soil or containers to increase carbon storage over other land uses (Craul, 1999). The organic matter can be in the form of fresh and composted food waste, mulch from yard waste, tree trimmings, park maintenance waste, and hay and manure from zoos and horse stables. Brown, Miltner and Cogger (2012) reported that over 50 percent of yard wastes are land applied. Moreover, in urban areas, soil amendments may be produced by recycling of human waste products (Brown, Miltner and Cogger, 2012). Urban agriculture areas are hotspots for biodiversity of plant and animals, both below and above ground (Tresch *et al.*, 2019). This biodiversity may increase SOC sequestration in urban agricultural soils.

Table **91** reports on SOC stocks and their evolution in two trials in Europe and North America.

Table 91. Changes in soil organic carbon stocks reported for two urban agriculture trials

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Years)	Depth (cm)	More information	Reference
Bottrop, Germany	Temperate, oceanic	Hortic Gleysol	164	-	-	30	14 vegetable patches in allotments	Burghardt and Schneider (2018)
Youngtown, Ohio, United States of America	Temperate	Degraded, compacted urban soil from demolition site	22	14	2	10	Compost and biochar amendment on ground; tomato and sweet potato	Beniston, Lal and Mercer (2014)
			19	21			Raised bed with compost and biochar; tomato and sweet potato	

4. Other benefits of urban agriculture

4.1. Improvement of soil properties

In urban areas, soil is subjected to several deleterious effects that deplete its quality, such as sealing, compaction, erosion, and contamination by organic and inorganic pollutants (Ferreira, Walsh and Ferreira, 2018). Urban agricultural activity that includes high plant diversity may improve many physical, chemical, and biological soil properties, especially in urban gardens (Tresch *et al.*, 2019). By implementing urban agriculture, soil cover promotes restoration of soil organic matter and sequestration of soil organic carbon, reduces erosion, and improves water management. Adding organic matter to garden soils on a regular basis is a common practice to improve the physical and chemical properties of the soil. This practice results in better fertility, lower bulk density, improved soil structure, increased water-holding capacity, and decreased surface evaporation losses and crusting (Bretzel *et al.*, 2016).

4.2 Minimization of threats to soil functions

Table 92. Soil threats

Soil threats	
Soil erosion	Agricultural activities may limit soil erosion if they include maintenance of soil cover and adequate water management that prevents runoff from entering or leaving agricultural areas.
Nutrient imbalance and cycles	Appropriate additions of organic matter to mineral soils builds soil health by improving soil structure and promotes establishment of healthy and diverse microbial populations, allowing for better water and nutrient holding capacities (Salomon *et al.*, 2020). Cover crops and crop rotations keep nutrients and greenhouse gases from escaping. If fertilizer is applied, the use of recommended rates minimizes the threat to groundwater pollution from nitrate leaching and from nitrogen and phosphorus pollution in runoff.
Soil salinization and alkalinization	Some agricultural practices, such as mulching, can help to prevent surface evaporation and thus minimize salinization in arid regions.
Soil contamination / pollution	Increasing the content of soil organic matter attenuates contaminants through adsorption and microbial use, reducing plant uptake, leaching of pollutants, and human exposure from skin contact and ingestion (Lal, 2017; Lehmann and Kleber, 2015; Chaney, Sterrett and Mielke, 1984). Biosolids have been applied with great success to sequester carbon, increase fertility, and attenuate heavy metal contamination.
Soil acidification	Adding calcitic or dolomitic lime to raise pH in acidic soils offsets acidification.
Soil biodiversity loss	Urban agriculture minimizes soil biodiversity loss if it is based on sustainable practices (Scialabba *et al.*, 2003). Urban gardens are hotspots of biodiversity because they include many plant species (Tresch *et al.*, 2019).
Soil sealing	Pervious pavements and paving materials minimize soil functional losses, reduce runoff, and increase water infiltration.
Soil compaction	Mitigating concentrated vehicle and foot traffic when the soil is wet minimizes soil compaction. Compacted soil has high bulk density and loss of soil structure, gas exchange, water holding and movement, and root tip expansion.
Soil water management	Urban agricultural soils that have a high content of organic matter may improve soil structure, thus enhancing water infiltration.

4.3 Increases in production (e.g. food/fuel/feed/timber)

Urban agriculture brings many benefits and helps to build more resilient urban communities by ensuring stronger urban food systems (Dubelling, van Veenhuizen and Halliday, 2019). It can help to fulfil the food requirements of urban populations, especially in developing countries. Regardless of the objective of the urban agriculture (for personal use or for profit), it always contributes to a more cohesive community.

Many people plant fruit trees in their yards and share the excess. In humid areas, vegetables can be grown at a rate of over 20 000 pounds per acre (22 400 kg per ha). Community gardens and school gardens promote growing and cooking of fresh foods and sharing among community members (Cordovil, Rodrigo and Gonçalves, 2015). Education and training are provided through school programs, teachers, and trainers, such as extension agents and specialists. By using containers, rooftop gardens, and raised beds, impervious surfaces can be converted into areas for food production. Distribution of the food raised in urban areas is simple and inexpensive compared with shipping from remote sources (Lee, Lee and Lee, 2015). Non-profit organizations and charity groups help grow, glean, and distribute food in their communities.

4.4 Mitigation of and adaptation to climate change

Urban agriculture reduces greenhouse gas emissions by significantly reducing the amount fuel burned to transport foods from outside sources (Lee, G-G., Lee, H-W. and Lee, J-H., 2015; Kulak, Graves and Chatterton, 2013). Further reductions in greenhouse gas emissions are associated with rooftop gardens, which reduce heating and cooling costs (Kulak, Graves and Chatterton, 2013).

4.5 Socio-economic benefits

Urban agriculture contributes to creation of jobs, both voluntary and paid. It serves to educate young people, gives new meaning to life, and promotes respect towards the elderly. It also serves as a recycling pool for kitchen and other domestic wastes, thereby serving as a tool to clean urban waste sites (Turner, 2010). Pleasure, recreation, exercise, socialization, social inclusion, education, aesthetics, environmental improvement, and therapy are benefits of gardening and food production on larger scales (Leake, Adam-Bradford and Rigby, 2009). Growing food often involves groups of people that share common interests. Food production, preservation, storage, and use involve an ongoing learning and training process.

4.7 Additional benefits to the practice

In addition to being an economical way to produce food, urban agriculture can also be aesthetically pleasing and beneficial to the birds, insects, bees, earthworms, microbes, mammals, and other biology of an urban area (Benth *et al.*, 2013). The art of growing and harvesting food is a calming and peaceful way to experience outdoor areas in an otherwise stressful urban environment (Leake, Adam-Bradford and Rigby, 2009).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Runoff, erosion, and deep percolation following the application of excess chemical fertilizers or manure can cause both surface and groundwater pollution by nitrogen, phosphorous, and potassium. Certain herbicides, pesticides, and fungicides can build up in the soil, harm humans, or harm beneficial insects, birds, bees, and earthworms. Reading labels and following directions are advised to minimize threats. Successful food production, however, often requires a balance between controlling weeds, insects, and diseases and accepting damaged or minimal vegetables and fruits.

Table 93. Soil threats

Soil threats	
Soil erosion	Some erosion is to be expected following tillage and between growing seasons (unless cover crops are used) and in sloping land.
Nutrient imbalance and cycles	Excess fertilizer can drive fertilizer dependence and drive higher fluxes of CO_2 and other greenhouse gases (Lorenz and Lal, 2009; Oertel *et al.*, 2016). Nutrient loss results from exposed soils, tillage, overworking the soils, poor irrigation management, insufficient soil depth, and an insufficient mineral soil component (Oertel *et al.*, 2016)
Soil salinization and alkalinization	In areas where the only water source is saline, irrigation may lead to soil salinization–careful water management is required. Treated wastewater may also be a source of salinity.
Soil contamination / pollution	Excessive fertilizer use can result in nitrate and phosphate losses to waterways. Heavy metals may accumulate near vehicle traffic. In constructed growth substrates, organic matter may be lost due to leaching (Grard *et al.*, 2018). Amendments (such as composts, biosolids, and other waste products) and imported soils may be sources of contaminants (Kumar and Hundal, 2016; Haynes, Murtaza and Naidu, 2009).
Soil acidification	Fertilizing with sulfur or sulfate products can lead to acidification of the soil. Large additions of acidic organic matter or compost can also.
Soil biodiversity loss	Abandoning a farm or garden can lead to weeds taking over or to a change in land use that causes soil biodiversity loss. Dependence on inorganic fertilizers, pesticides, and herbicides; overwatering; bare soils; monoculture planting; or tilling, reworking, or replacing the soil every season leads to soil biodiversity loss.
Soil sealing	Soil sealing is a risk due to establishment of impervious foot- and vehicle-traffic paths or roads.
Soil compaction	Soils may be compacted by foot- or vehicle-traffic that are concentrated in specific areas.

5.2 Possible greenhouse gas (GHG) emissions

If rooftop agricultural space is installed, GHG emission may occur due to transportation of building materials. Use of high amounts of labile organic matter, such as compost, or high amounts of mineral fertilizer can lead to CO_2 and N_2O emissions (Barthod *et al.*, 2018).

5.3 Conflict with other practice(s)

Urban agriculture may conflict with other urban planning priorities. Moreover, the correction of inequalities in the food system is neither inevitable nor guaranteed by urban agriculture. On the contrary, urban farms may even lead to displacement through eco-gentrification (Siegner, Sowerwine and Acey, 2018).

5.4 Other conflicts

Land tenure and the installation of security and anti-vandalism measures are possible sources of conflict arising from the implementation of urban farms (Turner, 2010). Conflicts between farming and other land uses, such as urbanization and municipal green spaces, are also possible.

6. Recommendations before implementation of the practice

Management of organic matter in soils or of engineered growth substrates used for food production is of primary importance to urban agriculture. Implementation of the practice must first consider the underlying (substrate) materials. The choice of materials depends on the site where production will take place (i.e. rooftop, garden, or urban farm). Organic materials used as growth substrates in urban agriculture include bark and composted materials, such as green (yard) wastes, municipal solid wastes, and even sewage sludge (Carlile, Cattivello and Zaccheo, 2015). "Lasagne" gardens with compost were shown to be a valuable technique for vegetable production on rooftops (Grard *et al.*, 2018).

Some urban soils have low fertility. Gardeners should avoid overusing chemical fertilizers and should instead use mature compost, which may greatly improve soil quality and allow for organic waste recycling (Bretzel *et al.*, 2016). Before compost application, soil properties should be evaluated to avoid detrimental effects, such as enhanced metal mobility (Murray, Pinchin and Macfie, 2011) and anti-germinative effects (Vidal *et al.,* 2020). As organic amendments and existing soils may be a source of contaminants, they should be tested and, if contaminated, amended to ensure safe gardening and edibles. Atmospheric dust and other potential current contamination sources (old buildings, construction activities, etc.) should be considered when establishing a proper garden or farm management plan.

The content of soil organic carbon (SOC) in soils that are used for urban agriculture in allotment gardens is strongly driven by the presence of trees (Edmondson *et al.*, 2014). The presence of trees and shrubs and the addition of carbon in the form of manure and compost are therefore recommended to maintain or increase SOC storage (Edmondson *et al.* 2014). Urban agriculture should increase soil biodiversity by using cover crops and companion plants, incorporating native crops and plants, integrating animals when possible, creating habitats, and minimizing monoculture farming, pesticides, herbicides, and commercial fertilizers (Scialabba *et al.*, 2003).

Erosion is a common threat in urban areas because of the high runoff from sealed and compacted surfaces. Poor soil structure and poor vegetative cover lead to exposure of soil particles and thereby erosion from wind and water. Managed food-plant growth and mulches can increase infiltration and minimize some of the excess exposure to wind, runoff, and erosion (Broz, Pfost and Thompson, 2017). Pavements can be made of pervious materials that still create a firm surface for walking or driving. Walking and traffic paths should be ripped or loosened to lessen their density; and organic matter should be added to promote development of soil structure. Walkways between garden beds can be covered with stepping stones or with layers of wood chips. These actions can minimize soil sealing, increase infiltration, and decrease runoff (Broz, Pfost and Thompson, 2017). Gardens on sloping ground should be terraced on the contour to decrease runoff and erosion. Water can be captured in cisterns or stormwater basins and used to water gardens. Most vegetable plants and fruits grow best in well drained soils that get regular, deep watering but do not stay saturated. Food plants, therefore, should not be grown in closed-bottom containers. Adequate water management is necessary in arid regions and in areas where irrigation water is saline. Cuevas *et al.* (2019) reviewed soil-improving cropping systems that minimize salinization and increase crop yields. Excess additions of pesticides, herbicides, and fungicides can lead to pollution and to loss of beneficial insects and biodiversity in soils (Scialabba *et al.*, 2003). Excess additions should thus be avoided; integrated pest management and use of organic gardening methods are advised whenever possible.

7. Potential barriers to adoption

Competition for land use in urban areas is high; and land use is regulated by zoning, ordinances, and homeowners' associations. Agriculture may be practiced on privately or publicly owned land. Biophysical, cultural, social, economic, institutional, legal, and knowledge barriers may exist for agricultural land use in urban areas.

Table 94. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Land for agriculture may not be available to urban dwellers. Fertility of urban soils may be low. Soils may contain a high concentration of toxic elements. Especially in the developing world, urban farmers may have limited access to such resources as organic matter, fertilizers, and water.

Barrier	YES/NO	
Cultural	No	Urban agriculture unites cultures and communities. It allows various cultures to share methods and to grow a variety of different crops, diversifying the palette and experiences.
Social	Yes/No	Gardens and farms nurture community and local pride and can become a social gathering place and outlet for neighborhoods. However, agricultural activities may not be accepted in some urban contexts due to the nuisance caused to neighbouring estates.
Economic	Yes	In cities, acquiring space for farming can be difficult. Significant funding may be required to secure a property and retain the right to farm year after year.
Institutional	Yes	Policy support and strategies for urban agricultural development are commonly missing in developing countries (Crush, Frayne and Pendleton, 2012).
Legal (Right to soil)	Yes	Inexperienced municipalities may enforce restrictions in soil testing or installation of structures such as high tunnels, resulting in a risk of liabilities.
Knowledge	Yes	Although gardening and farming are always a learning process best learned by experience, basic knowledge of potential crops and composting techniques requires capacity building.

Photos of the practice

Photo 29. Urban-agriculture community gardens, Commerce, Texas, United States of America

Photo 30. Rooftop garden, Chicago Botanical Gardens, Chicago, Illinois, United States of America

Photo 31. Urban community garden, Los Angeles, California, United States of America

Photo 32. Community garden, West Hollywood, California, United States of America

Photo 33. Community garden, Governors Island, New York City, New York, United States of America

Table 95. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Urban agriculture on rooftops in Paris, France - the T4P research project (Pilot Project of Parisian Productive Rooftops)	Europe	5	6	23
Urban Agriculture in Tacoma, Washington, United States of America	North America	1	6	26

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24. Urban forestry

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1. Description of the practice

Urban forestry is the management of single trees and forest resources, such as woodlands, in and around urban settings for the benefits they provide society. Urban trees are part of the urban green infrastructure, and they provide physiological, sociological, economic, and aesthetic benefits (Konijnendijk *et al.*, 2006). Proper forest management that promotes stand structure and production and minimizes soil disturbance also promotes retention of soil organic carbon (SOC) (Jandl *et al.*, 2007). Urban areas of all sizes with human populations of 50 000 and higher have included trees in urban gardens, parks, and lawns, and many have preserved forests and woodlands. While trees cover 10–67 percent of urban areas in the United States of America (Edmondson *et al.*, 2014), most cities in Europe have a tree density of 50–80 street trees per 1000 inhabitants (Pauleit *et al.,* 2002). In a major African city like Addis Ababa, urban forest covers <10 percent of the surface and consists mainly of introduced species intended to satisfy local wood demand (Woldegerima, Yeshitela and Lindley, 2017).

2. Range of applicability

Trees in urban areas range in extent from natural forests, planted woodlots, and tree-lined avenues down to individual trees planted in soil containers. Containerized trees line city streets and are grown on buildings and rooftops. Trees in urban areas often are exposed to stressful growing conditions, such as excess heat, air pollution, inadequate water and air supply (Fite *et al.*, 2011), inadequate soil pH, poor soil quality, heavy metals and other pollutants, and scarce availability of nutrients. Trees frequently are damaged by humans, animals, and cars. Root systems and trunks are constricted by pavements and sidewalks and are exposed to invasive diseases, insects, and other pests. This situation may be worsened by climate change because urban trees worldwide are composed of few species with low genetic variability of the species and varieties (Lohr, Kendal and Dobbs, 2016). Despite the stressful environment in which they grow, trees add substantial environmental functions and services to urban ecosystems.

3. Impact on soil organic carbon stocks

In the United States of America, the national average carbon storage density is 25.1 tC/ha for urban forests, compared with 53.5 tC/ha in forest stands (Nowak and Crane, 2002). Carbon storage in urban forests was found to vary between 15 and 160 tC/ha down to 1 m depth in cities in the United States of Americaand in China (Nowak *et al.*, 2013; Pouyat, Yesilonis and Novak, 2006). It was found to increase with increasing forest age (

Table **96**). Urban forest soils may contain greater proportions of recalcitrant carbon due to poorer quality leaf litter, enhanced mineralization of readily available carbon, and higher soil temperatures (Groffman *et al.,* 1995). Increases in urban soil carbon storage were observed by Pouyat *et al.* (2002) in New York City, where soil carbon density was about 30 percent higher in the urban forest than in suburban and rural forest soils. They also cited increased heavy metal and salt concentrations in urban forest soils. Higher soil carbon may be due to decreased respiration (Koerner and Klopatek, 2010) or reduced degradability of litter material (Pouyat *et al.,* 2002). However, the soil organic carbon storage under urban forests was much lower than under natural forests in the same region of Northeast China and increased with increasing age of the cities (Lv *et al.*, 2016). Studies have found that SOC is dependent on tree density (Mexia *et al.*, 2018) and on tree species or genus (Scharenbroch, 2012; Edmondson *et al.*, 2014). Urban trees that live longer and reach larger sizes have greater potential to sequester more carbon relative to shorter-lived, smaller trees (Nowak *et al.*, 2002).

Species mixture has been investigated as a strategy for conserving SOC in urban forests. This management technique is also associated with ecosystem resilience. However, the rate of carbon accumulation and its distribution within the soil profile differs between tree species. SOC enhancement was found to be highest under *Fraxinus* and *Acer spp*. and lower under *Quercus* and mixed woodland (Edmondson *et al.*, 2014).

For managed urban forests, Yoon *et al.* (2016) outline a strategy to enhance SOC by leaving litter and grass clippings in place to offset the loss of organic matter supplies. Application of compost, mulching, or both could increase urban soil carbon storage (Brown, Miltner and Cogger, 2012; Beesley *et al.*, 2012), as could application of biochar; however, it is not known how management practices might control inorganic carbon from carbonate reactions (Lorenz and Lal, 2015). Application of residual organic matter, such as yard and food waste and biosolids, to pervious surfaces in Tacoma, Washington, United States of America, would result in an annual carbon sequestration rate of 0.22 tC/ha/yr. This rate is similar to rates observed for no-till agriculture (Brown, Miltner and Cogger, 2012). Similarly, in arid settings, leaving plant material behind during maintenance has been shown to increase soil carbon storage (Rockhill, 2017; Swartz, 2019).

In arid regions, soils beneath the urban forest often have significantly higher soil carbon than the surrounding natural landscapes (Rockhill, 2017). This pattern of urban soil carbon content being higher in dry climates but lower in more humid settings has been documented as part of a broader convergence hypothesis for soil biogeochemistry in urban landscapes of North America (Trammell *et al.*, *2020*).

Table 96. Changes in soil organic carbon stocks reported for urban forests from diverse regions

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Years)	Depth (cm)	More information	Reference
Cities in United States of America and Canada	Temperate-subtropical	-	-	0.20-1.23	-	100	17 urban forests	Nowak *et al.* (2013) Pouyat, Yesilonis and Novak, (2006)
Shanghai, China	Subtropical	Alluvial	77.3	0.6	22	100	Young-near mature *Metasequoia. glyptostroboides* stands	ChunBo *et al.* (2010)
Kumasi, Ghana	Tropical	Haplic Alisol and Lithic Leptosol	45.5	-	-	60	Plantations increase SOC storage (83.5 tC/ha)	Nero *et al.* (2017)
Harbin city, NW China	Temperate, cold	Luvic Phaeozem	50	0.15	100	20	219 plots in urban forests	Lv *et al.* (2016)

4. Other benefits of the practice

4.1. Improvement of soil properties

Edmondson *et al.* (2014) studied soils under three genera (*Fraxinus*, *Acer*, and *Quercus*) in Leicester, England, and found that tree cover type did not influence soil bulk density or C/N ratio. However, Livesley *et al.* (2016) studied trees, grass areas, and turfgrass in sandy urban soils in Melbourne, Australia, and found that the soils under tree canopy were less compacted (1.07 g/cm^3) than in grassy areas (1.32 g/cm^3) and had higher mean C/N ratio.

4.2 Minimization of threats to soil functions

Erosion is a common threat in urban areas because of the high runoff from sealed, compacted, or bare surfaces. Erosion can result from the loss of leaf litter in shady areas because of excessive earthworm activity (Pouyat, Yesilonis and Novak, 2006) and from tree or shrub density shading out understory perennial ground plane vegetation. Poor soil structure and poor vegetative cover expose soil particles to the hazard of erosion by wind and water. Properly designed and managed planting areas can increase infiltration and minimize soil exposure to wind, runoff, and erosion. Pavements can be made of pervious materials that create a firm surface for walking or parking under trees but still allow infiltration and decrease runoff. Supplemental water can maintain tree growth in semi-arid and arid regions and in humid areas where pavements limit infiltration of rainwater. Excess additions of herbicides, pesticides, and fungicides can lead to loss of beneficial insects and biodiversity in soils. Hydrophobicity of soil and litter under trees increase runoff from rainfall at the beginning of storm events. Removal of leaf litter soon after fall is recommended but must be compensated for by other runoff control measures.

Table 97. Soil threats

Soil threats	
Soil erosion	Depending on tree species, soil erosion can be reduced by tree planting in urban areas, especially in sloping areas (Woldegerima, Yeshitela and Lindley, 2017).
Nutrient imbalance and cycles	Fertilizing at recommended rates minimizes the threat of groundwater pollution from nitrate leaching and from nitrogen and phosphorus pollution in runoff. Net nitrogen mineralization rates in samples from the forest floor and A horizon vary by tree type (White and McDonnell, 1988). Trees can reduce oxidized nitrogen, reactive phosphorous, and stormwater nitrogen pollution (Denman, May and Moore, 2016).
Soil contamination / pollution	Planting trees may reduce the risk of soil contamination and in particular enhance the breakdown of organic contaminants (Dickson *et al.,* 2000).
Soil biodiversity loss	The importance of urban forest as a reserve of biodiversity will increase in the future (Alvey, 2006).
Soil sealing	Establishing urban woodland and trees prevents soil from sealing.
Soil compaction	Soil compaction may be reduced following tree planting.
Soil water management	Trees in flood plains slow down the velocity and increase turbulence of floodwaters, mitigating the harmful effects of flooding.

4.3 Increases in production (e.g. food/fuel/feed/timber)

Very little timber is harvested from individual urban trees for purposes other than firewood or making furniture because of the high cost to cut the tree safely and transport the pieces away using heavy machinery. Also, in places, production is low due to trace metal pollution. Most of the poor-quality material is used for firewood or made into mulch, especially in developing countries (Woldegerima, Yeshitela and Lindley, 2017). Woodlots or forests that have become assimilated into expanding urban areas may have timber values, especially in the southeastern United States where commercial stands of pine are common.

4.4 Mitigation of and adaptation to climate change

Urban forests also provide climate change mitigation services, including carbon sequestration and the reduction of the urban heat-island effect (Kleerekoper, van Esch and Salcedo, 2012). Temperature reductions to the heat island can be as much as 6 °C due to vegetation canopy (Shiflett *et al.*, 2017). Deciduous trees on southern and westerly aspects of urban buildings in the Northern hemisphere may potentially reduce energy emissions by shading in summer (McPherson, 1994).

4.5 Socioeconomic benefits

Urban forests provide a variety of essential ecosystem services, including decreasing air, water, and noise pollution, mitigating flood risk, and providing recreational areas (Escobedo, Kroeger and Wagner, 2011; Roy, Byrne and Pickering, 2012).

The economic benefits of urban forests have been quantified in a recent study. According to the authors, the economic benefit increases with tree cover and amounts to 0.93 million USD savings in air pollution health care costs, USD 20000 by capturing water runoff, and USD 478000 in building energy heating and cooling savings (Endreny *et al.*, 2017).

4.6 Additional benefits to the practice

Urban trees and woodlands can act as biological filters and remove airborne pollutants and thus improve air quality in urban areas (Beckett, Freer-Smith and Taylor, 1998).

5. Potential drawbacks to the practice

5.1 Tradeoffs with other threats to soil functions

Repeated timber harvesting and similar management practices can cause soil compaction and erosion. Following best management practices prescribed by conservation agencies and cooperative extension specialists is recommended.

Table 98. Soil threats

Soil threats	
Soil erosion	Commercial timber production or removal of stumps may cause soil erosion. Exposed and non-vegetated soil under urban forest and individual tree soils in semi-arid and non-arid climates are vulnerable to erosion. Excess recreation under tree canopies can result in exposure of the surface soil to erosion.
Soil salinization and alkalinization	Soil salinization of trees in golf courses is increased when saline groundwater is used to water soils that have surface compaction, restricted subsoil permeability, and higher clay contents (Miyomoto and Chacon, 2006).
Soil contamination / pollution	Tree canopies may capture atmospheric dust, which is a potential source of contamination.
Soil acidification	Urban trees that have highly acid needles (juniper, pines, spruce, larch, fir, other evergreens) may acidify the soil. Fertilizing with sulfur or sulfate products can lead to acidification of the soil, as can large additions of acid-producing leaf litter or compost from such matter.
Soil compaction	Excess recreational activities under tree canopies can result in compaction.
Soil water management	Trees are not suitable for growth within stormwater retention basins or on onsite waste disposal drain fields and must be removed if they grow there. All forest-floor and A-horizon samples from an urban forest study in New York City were extremely hydrophobic (White and McDonnell, 1988). Tree planting may lead to enhanced evaporation and thus less water in soil.

5.2 Increases in greenhouse gas emissions

Greenhouse gas emissions are associated with timber harvesting that uses mechanized equipment, such as chainsaws, skidders, and haul trucks. Cutting trees reduces the amount of carbon dioxide removal until the biomass regains its former level. Waste products and firewood that are burned release carbon dioxide into the atmosphere.

5.3 Conflict with other practice(s)

Forestry in urban areas can conflict with other land use choices that are higher producers of income or are deemed to be more valuable, such as housing and business developments, landfills, and transportation projects.

5.4 Decreases in production (e.g. food/fuel/feed/timber)

Soil compaction from recreational or sports activities may be harmful to nearby tree root systems. Construction of infrastructure and damage by impact from vehicles is harmful to tree growth. Air and water pollution reduce growth and make trees more vulnerable to diseases and pests. Invasive diseases and insects threaten native species that are not adapted to the threat.

6. Recommendations before implementation of the practice

Relatively few studies specifically address soil carbon management in urban forests. However, long-term studies of forest management strategies of rural forests recommend increasing productivity, e.g., through afforestation and planting of fast-growing tree species. Increased productivity has immediate effects on SOC by incorporating CO_2 in plant biomass and increasing carbon input to the soil (Jandl *et al.*, 2007). Urban sites are often characterized by limited rooting space and compacted or polluted soil, poor drainage, and high variability (Day and Bassuk, 1994), which need to be amended before trees can be planted. Species choice is important. Site-specific properties need to be matched to species performance. Eucalyptus species were reported to have high erosion incidence in arid sloping land, while mixed forests performed better and also increase biodiversity (Woldegerima, Yeshitela and Lindley, 2017). Foot and vehicle traffic should be concentrated in a few areas rather than underneath trees. When sidewalks and pavements must be added, pervious materials should be used if possible. Soils that have been compacted by foot and vehicle traffic should be ripped or loosened to lessen the density, and organic matter should be added to promote soil structure development. Walkways beneath trees can be covered with layers of wood chips to minimize compaction.

7. Potential barriers to adoption

Table 99. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	In more arid regions, availability of water can be an impediment to the benefits of urban forests.
Economic	Yes	Excessive tree growth in humid areas may conflict with views from buildings and need costly thinning or removal. Cost of tree establishment and maintenance may be a barrier to planting, especially in arid and semi-arid areas where trees are not native.
Institutional	Yes	Inadequate funding and lack of political and public support despite community interest were identified as main barriers to urban forestry.

Barrier	YES/NO	
Legal (Right to soil)	Yes	Damage to infrastructure and personal property from falling trees may cause litigation.
Other	Yes	Inadequate care and maintenance of trees after planting can lead to high mortality rates.

Photos of the practice

Photo 34. Tree growing in a containerized bunker in New York City. Beneath the pavement is a subway line

Photo 35. Trees surrounded by pervious brick pavement

Photo 36. Trees lining an urban park in New York City (Central Park)

Photo 37. Urban woodlots

Table 100. Related case studies available in volumes 4 and 6

Title	Region	Duration of study (Years)	Volume	Case-study n°
Carbon storage in soils built from waste for tree plantation in Angers, France	Europe	3	6	22
Urban Forestry effects on soil carbon in Leicester, United Kingdom of Great Britain and Northern Ireland	Europe	20 to 100	6	25
Soil Organic Carbon in forested and non-forested urban plots in the Chicagoland Region, United States of America	North America	Various	6	27
Compost application to restore post-disturbance soil health in Montgomery county, Virginia, United States	North America	4	6	28

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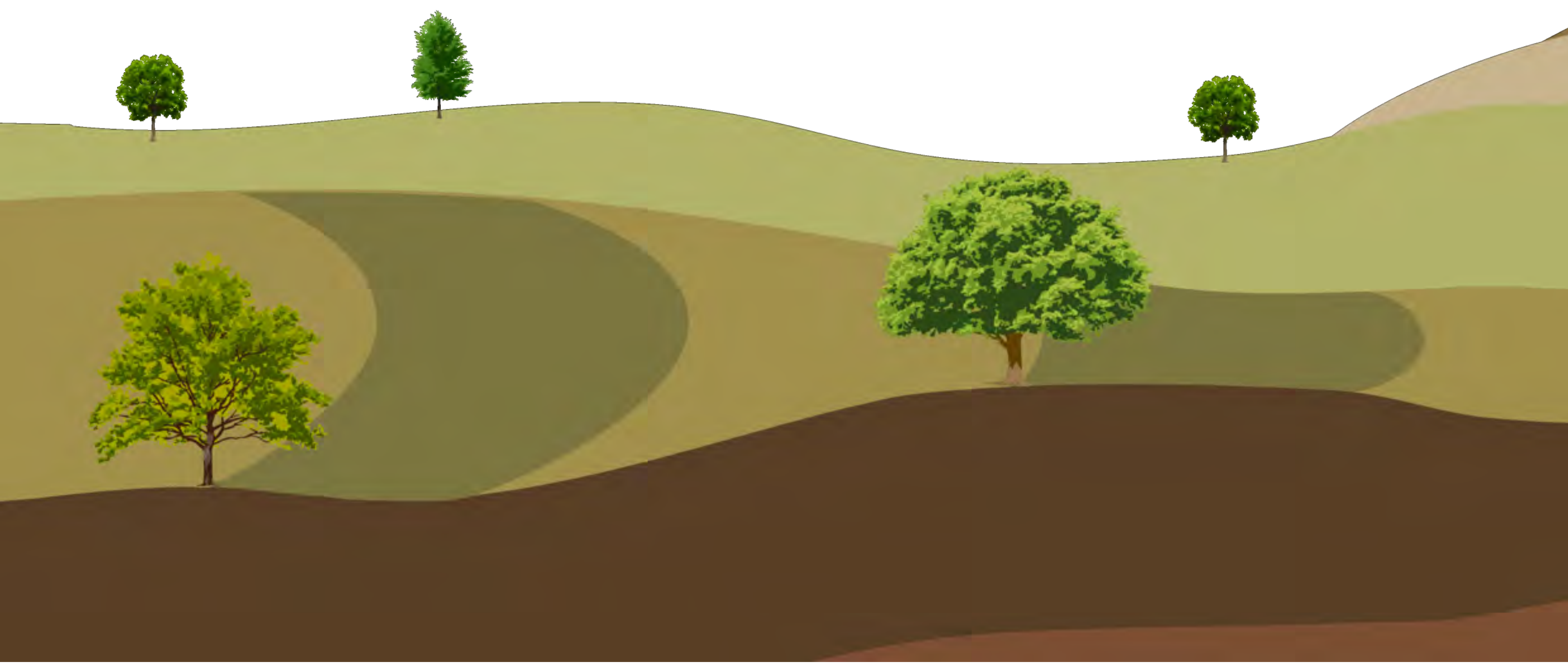

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