Food and Agriculture Organization of the United Nations

VOLUME 6

RECARBONIZING GLOBAL SOILS

CASE STUDIES

A technical manual of recommended management practices

FORESTRY, WETLANDS, URBAN SOILS

ITPS INTERGOVERNMENTAL TECHNICAL PANEL ON SOILS

GLOBAL SOIL PARTNERSHIP

VOLUME 6

CASE STUDIES

RECARBONIZING GLOBAL SOILS

A technical manual of recommended management practices

FORESTRY, WETLANDS, URBAN SOILS

Food and Agriculture Organization of the United Nations
Rome, 2021

Required citation:

FAO and ITPS. 2021. *Recarbonizing global soils: A technical manual of recommended management practices. Volume 6: Forestry, wetlands, urban soils – Case studies.* Rome. https://doi.org/10.4060/cb6605en

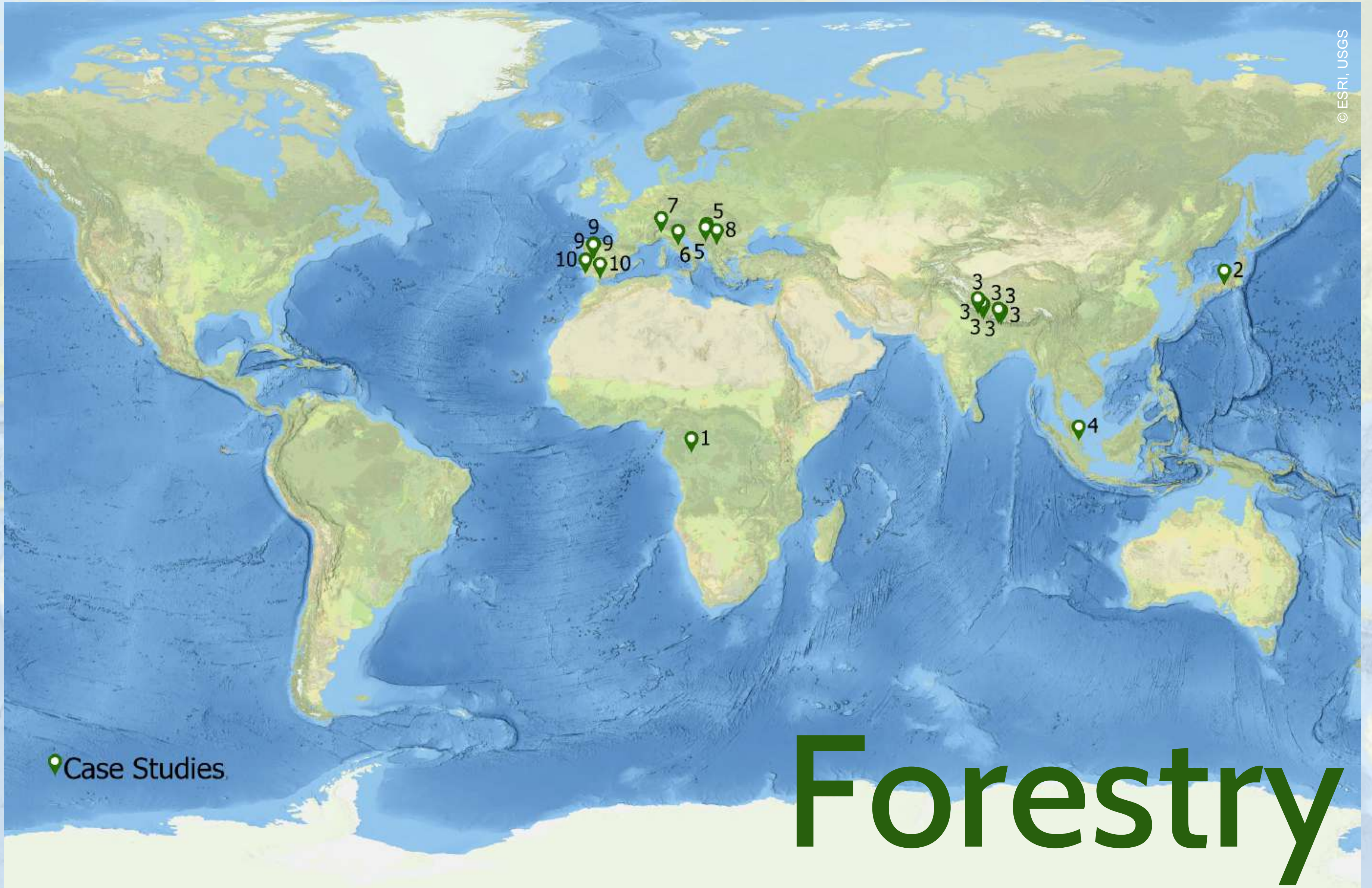
ISBN 978-92-5-134899-4

Contents

Tables

Figures

Photos



Forestry

Case Study ID	Region	Title	Practice 1	Practice 2	Practice 3	Duration
1	Africa	Soil fertility improvement of nutrient-poor and sandy soils in the Congolese coastal plains	Afforestation	Introduction of nitrogen-fixing trees (NFT) in forest plantations		7
2	Asia	Afforestation of a former farmland in Japan	Afforestation			5
3	Asia	Forest conservation through community management in the hill and plain areas of Nepal	Forest conservation	Agroforestry	NA	NA
4	Asia	Soil organic carbon stocks in forests of Singapore	Forest consservation	Natural regeneration		Various
5	Europe	Reforestation of highlands in Javor Mountain, Republic of Srpska, Bosnia and Herzegovina	Afforestation	Reforestation		15
6	Europe	Natural afforestation of abandoned mountain grasslands along the Italian peninsula	Natural regeneration			23 to 80
7	Europe	Afforestation of vineyards in Italy	Afforestation			30

Case Study ID	Region	Title	Practice 1	Practice 2	Practice 3	Duration
8	Europe	Afforestation by planting in bench terraces: Kalimanska watershed, Grdelica gorge, Southeastern Serbia	Forest restoration	Afforestation	Reforestation	60
9	Europe	Conservation of degraded forests of central and western Spain	Forest conservation	Community forest		22 to 80
10	Europe	Straw mulch and biochar application in recently burned areas of Algarve (Portugal) and Andalusia (Spain)	Fire management	Biochar	Mulch	1

1. Soil fertility improvement of nutrient-poor and sandy soils in the Congolese coastal plains

Lydie-Stella Koutika

CRDPI, Pointe-Noire, Republic of the Congo

1. Related practices

Forest afforestation, Introduction of nitrogen-fixing trees in forest plantations

2. Description of practice

Afforestation or reforestation represents a significant silviculture and forest management practice worldwide (Paquette and Messier, 2010). It potentially improves soil fertility, provides wood and sequesters C in both soil and biomass with further impact in mitigating climate change (Bauters *et al.*, 2015; Sang *et al.*, 2013; Lee *et al.*, 2015; Rumpel *et al.*, 2020). Afforestation of tropical native savannas on the inherently poor and sandy soils in the Congolese coastal plains using eucalypt started in the 1950's (Makany, 1964). This practice was implemented to preserve natural forests as forest plantations substitute for harvesting natural forests and provide both wood for the pulp industry and fuel energy for the rural population (Delwaulle, Garbaye and Laplace, 1978, 1981) since around 94 percent of Congolese homes use forest wood fuel (Shure *et al.,* 2010), and also use unsuitable soils for agriculture. However, nutrient availability often declines along with the productivity of these fast-growing plantations after successive rotations and harvests and nutrients exported at harvest are not replenished in the Congolese coastal plains as well as in other ecosystems such as south-western Australia (Corbeels *et al.,* 2005; Laclau *et al.*, 2005). Nitrogen-fixing trees (NFT) e.g. *Acacia mangium* and *Acacia auriculiformis* have been therefore introduced since the 1990s to restore soil fertility, intensify and sustain forest productivity (Bernhard-Reversat, 1993; Bouillet *et al.*, 2013).

3. Context of the case study

This is a regional case. According to Koppen-Geiger climate classification, the coastal plains in the southwest Democratic Republic of Congo are savanna subtropical (type Aw). The primary drainage is the Kouilou-Niari River.

4. Possibility of scaling up

Introducing NFT in forest plantations may scale up to soils, such as those inherently poor and coarse textured with low fertility status and low potential for agriculture beneath tropical native savannas in the Congolese coastal plains. Those soils extend to around 6 million hectares in Central African countries, including the Democratic Republic of Congo (DRC), Gabon and the Republic of the Congo (RoC) (Schwartz and Namri, 2002). For instance, the benefits of this practice were enhanced in more nutrient-poor soil (<1 percent of SOM and > 90 percent of sand) in Congo than in Brazil (higher SOM content and 84 percent of sand). Ferralic Arenosols at Tchissoko in the Congolese coastal plains responded better to NFT introduction into eucalypt plantations through increase C and N stocks, stand wood biomass (Epron *et al.*, 2013; Koutika *et al.*, 2014; Tchichelle *et al.*, 2017) than the Ferralsol counterpart at Itatinga, Sao Paulo in Brazil (Voigtlander *et al.*, 2012; Epron *et al.*, 2013).

5. Impact on soil organic carbon stocks

Introducing NFT in forest plantation leads to C sequestration in both soil and biomass (Epron *et al.*, 2013; Koutika *et al.*, 2014). Soil organic C stocks down to 25 cm increased in the mixed-species (50 percent *A. mangium* and 50 percent eucalypt) stands (17.8 ± 0.7 t/ha) relative to pure acacia (16.7 t/ha) and eucalypt stands (15.9 t/ha), at the end of the first 7-year rotation in Congo (Koutika *et al.*, 2014; **Table 1**).

Table 1. Additional SOC storage on the 0-25 cm layer depth after 7 years of afforestation with NFT in the Republic of the Congo

Koutika et al. (2014)

Climate is subtropical and soils are Ferralic Arenosols

Baseline C stock (tC/ha)	Additional C storage potential (tC/ha/yr)	Treatment
15.9 (Plantation of Eucalypt)	0.8	Plantation of *Acacia*.
	1.9	Plantation of *Acacia mangium* and Eucalypt.

6. Other benefits of the practice

6.1. Improvement in soil properties and stand wood biomass

Introducing *A. mangium* in the eucalypt plantations significantly improved N status of the nutrient- poor soils of the Congolese coastal plains (Koutika *et al.*, 2014; Koutika *et al.*, 2017; Tchichelle *et al.*, 2017) and stand wood biomass (Epron *et al.*, 2013). Changes in soil properties are involved by the shift in soil bacterial community composition, specifically a prevalence of *Firmicutes* phylum in stands containing *A. mangium* against Proteobacteria in those of pure eucalypt (Koutika *et al.*, 2020) as already found in other part in the tropics (Bini *et al.*, 2013; Pereira *et al.,* 2018).

6.2 Minimization of threats to soil functions

Table 2. Soil threats

Soil threats	
Soil erosion	Reduction in wind speed following afforestation (No reference available).
Nutrient imbalance and cycles	Improve N soil status (Koutika *et al.* 2014, 2017); Cumulative net N stocks of 343 ± 21 kg/ha in acacia, 287 ± 17 kg/ha in acacia-eucalypt, and 189 ± 12 kg/ha in eucalypt stands in the first two years of the second 7-year (Tchichelle *et al.*, 2017).
Soil acidification	Reductions in pH value (Koutika, Mareschal and Epron, 2016; Koutika *et al.*, 2014 and 2019). The effects increase with the length of rotation duration, i.e. in a longer term.
Soil biodiversity loss	Soil biodiversity enrichment occurs (Koutika *et al.*, 2020).
Soil water management	No specific studies have been conducted. However, soil water content was lower beneath acacia relative to eucalypt in the 15 first centimeters (Koutika, Mareschal and Rudowski, 2018).

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

The practice provides fuel energy for the local population since around 94 percent of Congolese homes use forest wood fuel (Shure *et al.*, 2010).

6.4 Mitigation of and adaptation to climate change

The practice has great potential as a measure in the mitigating climate changing through C sequestration in both soils (Koutika *et al.*, 2014; Tchichelle *et al.*, 2017) and biomass (Epron *et al.*, 2013).

6.5 Socio-economic benefits

No studies have been conducted up to date. However, for instance the benefit of 27 000 frs cfa (local money) i.e. 41 euros is made for eucalypt considering the stand wood biomass of 7.3 t/ha/yr in eucalypt relative to 10 t/ha/yr in acacia and eucalypt stands at the end of the 7 year first rotation. (Epron *et al.* 2013)

6.6 Other benefits of the practice

The benefits of the current practice are several since knowledge on soil process can be easily be used in Agro forestry and agriculture.

7. Conflict with other practice(s), possible drawbacks

7.1 Tradeoffs with other threats to soil functions

Table 3. Soil threats

Soil threats	
Nutrient imbalance and cycles	Decrease in available P due to NFT requirement for P during N fixation (Koutika *et al.*, 2014).
Soil acidification	Reductions of pH (Koutika, Mareschal and Epron, 2016; Koutika *et al.*, 2019). The effects increase with the experiment duration i.e. in the longer term.
Soil biodiversity loss	Increase in biodiversity after introducing acacia in eucalypt plantations i.e. birds, plants species and macrofauna (Koutika *et al.*, personal communication).

7.2 Increases in greenhouse gas emissions

No direct measurements have been made. However, GHG emissions may have potentially decreased due to the good practice. Measurements of GHS emissions will be performed in the future.

7.3 Conflict with other practice(s)

Afforestation of tropical savannas of the Congolese coastal plains leads to a strong conflict with other practices such as traditional slash-and-burn practice common in the region. The traditional practice is annual, less expensive and farmers may have incomes in the shorter term.

7.4 Negative impact on production

No negative impact on production has estimated. However, the practice harbour several advantages such as fuelwood and non-timber forest products supply, increase in soil biodiversity, mitigation of desertification by both sustaining forest plantations and preserving natural forests as 94 percent of the population used fuel wood energy. Another advantage of intercropping nitrogen fixing trees with eucalyptus in a commercial plantation is due to the high costs and low availability of inorganic fertilizers in the Republic of the Congo (Koutika *et al.*, personal communication), while the practice guarantees an improvement in soil N status (Tchichelle *et al.*, 2017; Paula *et al.* 2018) in addition the mentioned above advantages.

8. Recommendations before implementing the practice

Studies on impact assessment and risk of invasiveness of the introduced species must be conducted before the implementation.

9. Potential barriers for adoption

Table 4. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	A longer dry season may threaten acacia growth at the younger age (up to 2 years), especially in the second rotation (Koutika *et al.*, 2018). There is a potential risk of invasiveness of *A. mangium* (Koutika and Richardson, 2019). Preference should be given to native NFTs.
Social	Yes	Land tenure conflicts (government / landowners).
Economic	No	In addition to soil fertility improvement the practice in the contrary provides fuel wood energy for rural population.
Institutional	Yes	No support for the government and no official reforestation plan including the practice.
Legal (Right to soil)	Yes	No legal producer's property titles.
Knowledge	Yes/No	Producers do not know how to enhance the benefits of acacia. Yes, there are interested students or more research is underway, but there are no funds (proposals with other four African countries in progress).
Other	Yes	Lack of motivation.

Photos

Photo 1. Tropical native savanna beside forest plantation in the Congolese coastal plains, Republic of the Congo

Photo 2. Mixed-species plantation of Acacia mangium and Eucalyptus urophilla grandis in the Congolese coastal plains (4° 44' 41''S & 12° 01' 51'' E), Republic of the Congo

References

Bauters, M., Ampoorter, E., Huygens, D., Kearsley, E., De Haulleville, T., Sellan, G., Verbeeck, H., Boeckx, P. & Verheyen, K. 2015. Functional identity explains carbon sequestration in a 77-year-old experimental tropical plantation. *Ecosphere*, 6(10): 198. http://dx.doi.org/10.1890/ES15-00342.1

Bernhard-Reversat, F. 1993. Dynamics of litter and organic matter at the soil-litter interface of fast-growing tree plantations on sandy ferrallitic soils (Congo). *Acta Ecologica*, 14 (2):179-195.

Bini, D., dos Santos, C.A., Bouillet, J.-P.P., Gonçalves, J.L.M. & Cardoso, E.J.B.N. 2013. Eucalyptus grandis and Acacia mangium in monoculture and intercropped plantations: evolution of soil and litter microbial and chemical attributes during early stages of plant development. *Applied Soil Ecology*, 63: 57-66. https://doi.org/10.1016/j.apsoil.2012.09.012

Bouillet, J-P., Laclau, J.P., Gonçalves, J.L.M., Voigtlaender, M., Gava, J.L., Leite, FP., Hakamada, R., Mareschal, L., Mabiala, A., Tardy, F., Levillain, J., Deleporte, P., Epron, D. & Nouvellon, Y. 2013. Eucalyptus and Acacia tree growth over entire rotation in single- and mixed-species plantations across five sites in Brazil and Congo. *Forest Ecology and Management*, 301: 89-101. https://doi.org/10.1016/j.foreco.2012.09.019

Corbeels, M., McMurtrie, R.E., Pepper, D.A., Mendham, D.S., Grove, T.S. & O'Connell, A.M. 2005. Long-term changes in productivity of eucalypt plantations under different harvest residue and nitrogen management practices: a modelling analysis. *Forest Ecology and Management,* 217, 1–18. https://doi.org/10.1016/j.foreco.2005.05.057

Delwaulle, J.C., Garbaye, J. & Laplace, Y. 1978. *Un record de production forestière: Les reboisements en eucalyptus hybrides de la savane côtière Congolaise.* Rapport CTFT.

Delwaulle, J.C., Garbaye, J. & Laplace, Y. 1981. Ligniculture en milieu tropical: Les reboisements en eucalyptus hybrides de la savane côtière Congolaise. *Revue Forestière Française*, 3: 248-255. https://doi.org/10.4267/2042/21511

Epron, D., Nouvellon, Y., Mareschal, L., Moreira, R.M., Koutika, L.S., Geneste, B., Delgado-Rojas, JS., Laclau, J-P., Sola, G., Gonçalves, JLM. & Bouillet, J-P. 2013. Partitioning of net primary production in *Eucalyptus* and *Acacia* stands and in mixed-species plantations: Two case-studies in contrasting tropical environments. *Forest Ecology and Management*, 301: 102-111. https://doi.org/10.1016/j.foreco.2012.10.034

Koutika, L.-S. 2019. Afforesting tropical savannas with *Acacia mangium* and eucalyptus improves soil P availability in Arenosols of the Congolese coastal plains. *Geoderma Regional*, e00207. https://doi.org/10.1016/j.geodrs.2019.e00207

Koutika, L-S. & Richardson, D.M. 2019. *Acacia mangium* Willd: Benefits and threats associated with its increasing use around the world (Review). *Forest Ecosystems*, 6(2) : 1-13. https://doi.org/10.1186/s40663-019-0159-1

Koutika, L.-S., Mareschal, L. & Epron, D. 2016. Soil P availability under eucalypt and acacia on Ferralic Arenosols, republic of the Congo. *Geoderma Regional*, 7(2): 153–158. https://doi.org/10.1016/j.geodrs.2016.03.001

Koutika, L.-S., Mareschal, L. & Rudowski, S. 2018. Fate of *Acacia mangium* in eucalypt mixed-species plantation during drought conditions in the Congolese coastal plains. *Bosque*, 39(1): 131-136. https://doi.org/10.4067/S0717-92002018000100012.

Koutika, L.-S., Epron, D., Bouillet, J.-P. & Mareschal, L., 2014. Changes in N and C concentrations, soil acidity and P availability in tropical mixed acacia and eucalypt plantations on a nutrient-poor sandy soil. *Plant Soil*, 379: 205-216. http://dx.doi.org/10.1007/s11104-014-2047-3

Koutika, L,-S., Tchichelle, S.V., Mareschal, L., & Epron, D. 2017. Nitrogen dynamics in a nutrient-poor soil under mixed-species plantations of eucalypts and acacias. *Soil Biology and Biochemistry*, 108: 84-90. https://doi.org/10.1016/j.soilbio.2017.01.023

Koutika, L.-S., Fiore, A., Tabacchioni, S., Aprea, G., Pereira, A.P. de A. & Bevivino, A. 2020. Influence of Acacia mangium on Soil Fertility and Bacterial Community in Eucalyptus Plantations in the Congolese Coastal Plains. *Sustainability*, 12(21): 8763. https://doi.org/10.3390/su12218763

Laclau, J.P., Ranger, J., Deleporte, P., Nouvellon, Y., Saint André, L., Marlet, S. & Bouillet, J.P. 2005. Nutrient cycling in a clonal stand of eucalyptus and an adjacent savanna ecosystem in Congo. 3. Input-output budget and consequences for the sustainability of the plantations. *Forest Ecology and Management*, 210: 375-391. https://doi.org/10.1016/j.foreco.2005.02.028

Lee, K.L., Ong, K.H., King, P.J.H., Chubo, J.K. & Su, D.S.A. 2015. Stand productivity, carbon content, and soil nutrients in different stand ages of Acacia mangium in Sarawak, Malaysia. *Turk J Agric For* 39: 154-161. https://doi.org/10.3906/tar-1404-20

Makany, L. 1964. La côte atlantique du Congo cadres géographiques et géologiques, leur influence sur la répartition de la végétation et sur les possibilités agricoles du territoire. Symposium Science, Pekin, 891-907.

Paula, R.R., Bouillet, J.P., de M. Gonçalves, J.L., Trivelin, P.C.O. Balieiro , F.C., Nouvellon, Y., Oliveira, J.C., de Deus Junior, J.C., Bordron, B. & Laclau, J.P. 2018. Nitrogen fixation rate of *Acacia mangium* Wild at mid rotation in Brazil is higher in mixed plantations with *Eucalyptus grandis* Hill ex Maiden than in monocultures. *Annals of Forest Science,* 75: 14. https://doi.org/10.1007/s13595-018-0695-9

Paquette, A. & Messier, C. 2010. The role of plantations in managing the world's forests in the Anthropocene. *Frontiers in Ecology and Environment*, 827-34. https://doi.org/10.1890/080116

Pereira, A.P.A., Zagatto, M.R.G., Brandani, C.B., Mescolotti, D. de L., Cotta, S.R., Gonçalves, J.L.M. & Cardoso, E.J.B.N. 2018. Acacia Changes Microbial Indicators and Increases C and N in Soil Organic Fractions in Intercropped Eucalyptus Plantations. *Frontiers in Microbiology,* 9. https://doi.org/10.3389/fmicb.2018.00655

Rumpel, C., Amiraslani, F., Chenu, C., Garcia-Cardenas, M., Kaonga, M., Koutika, LS., Ladha, J., Madari, B., Shirato, Y., Smith, P., Soudi B., Soussana, J-F., Whitehead, D. & Wollenberg, L. 2020. The 4p1000 initiative: Opportunities, limitations and challenges for implementing soil organic carbon sequestration as a sustainable development strategy. *Ambio*, 49: 350-360. https://doi.org/10.1007/s13280-019-01165-2.

Sang, P.M., Lamb, D., Bonner, M. & Schmidt, S., 2013. Carbon sequestration and soil fertility of tropical tree plantations and secondary forest established on degraded land. *Plant Soil*, 362: 187-200. https://doi.org/10.1007/s11104-012-1281-9

Shure, J., Marien, J.N., de Wasseige, C., Drigo, R., Salbitano, F., Dirou, S. & Nkoua, M. 2010. Contribution du bois énergie à la satisfaction des besoins énergétiques des populations d'Afrique Centrale. *Perspectives pour une gestion durable des ressources disponibles*, 109-122.

Schwartz, D. & Namri, M. 2002. Mapping the total organic carbon in the soils of the Congo. *Global Planet Change*, 33:77-93.

Tchichelle, S.V., Epron, D., Mialoundama, F., Koutika, L.S., Harmand, J.M., Bouillet, J.P. & Mareschal, L. 2017. Differences in nitrogen cycling and soil mineralization between a eucalypt plantation and a mixed eucalypt and *Acacia mangium* plantation on a sandy tropical soil. *South Forests*, https://doi.org/10.2989/20702620.2016.1221702.

Voigtlaender. M., Laclau, J-P., Gonçalves, JLM., Piccolo, MC., Moreira, MZ., Nouvellon, Y., Ranger, J. & Bouillet, J-P. 2012. Introducing *Acacia mangium* trees in *Eucalyptus grandis* plantations: consequences for soil organic matter stocks and nitrogen mineralization. *Plant Soil*, 352: 99-111. https://doi.org/10.1007/s11104-011-0982-9

2. Afforestation of a former farmland in Japan

Hisao Sakai

Forestry and Forest Products Research Institute (FFPRI), Japan

1. Related practices

Afforestation

2. Description of the case study

This case-study introduces measurements of changes in soil carbon when trees are planted in the experimental site of the Forestry and Forest Products Research Institute in Japan. The examined tree species are Japanese cedar (*Cryptomeria japonica*) and Hinoki cypress (*Chamaecyparis obtusa*), which are the main tree species planted for timber production in Japanese mountains.

In 2001, the first soil sampling was conducted at the depths of 0-5, 5-15 and 15-30 cm for cedar sites (4-year-old, 14-year-old and 23-year-old) and cypress sites (4-year-old, 12-year-old and 25-year-old). The second sampling was done in 2006. Repeated sampling showed that an increase in soil carbon was mainly seen at the depth of 0-5 cm and soil carbon did not show statistically significant changes at the depth of 5-15 and 15-30 cm. Soil carbon was estimated to accumulate at an average rate of 0.229 tC/ha/yr in Japanese cedar sites and 0.211 tC/ha/yr in Hinoki cypress sites. The rate of increase in the surface soil carbon was relatively larger in older stands than in young stands, indicating that the soil carbon increased slowly but steadily about 30 years after the planting.

Japan does not encourage land-use change from farmland to forest, because farmland is the basis for food production (Ministry of Agriculture, Forestry and Fisheries, 2016). However, the area of idled farmland is increasing due to the aging population. If there is no way to maintain these areas as active farmland, afforestation may be applied as an option in some cases (Ministry of Agriculture, Forestry and Fisheries, 2007).

3. Context of the case study

The location is a place used as a nursery flat field for trees at the Chiyoda Experimental Station of the Forestry and Forest Products Research Institute (36°11´N, 140°13´E, 35 m a.s.l.) on the Kanto Plain in central Japan. The mean annual precipitation and air temperature are 1 186 mm and 13.5 °C, respectively, and climatic conditions belong to the moist temperate zone. The soil type is Andisol derived from volcanic ash and is widely distributed throughout Japan. At the Japanese cedar sites, more than 3 000 trees were planted per hectare, while 2 400 trees per hectare were planted at the Hinoki cypress sites. At these research sites, no thinning was done and fertilization has not been applied. The stem volume reached 469 m^3/ha at the 23-year-old Japanese cedar site and 291 m^3/ha at 25-year-old Hinoki cypress site; both species showed above-average growth in the region. Fertilization has not been applied at the research sites. Weeding was carried out every year for 5 years after planting, and subsequent weeding was done as needed. At the grass-covered (Lawn site) as a reference site, the grasses were mowed several times a year and were kept at a low height.

4. Possibility of scaling up

Afforestation practices may be applicable, for example, to abandoned cultivated land. Japanese cedar and Hinoki cypress can be planted all over Japan except Hokkaido of the most northern part of Japan. If the needs of local community are not for suitable for timber production, but are for soil conservation and/or biodiversity restoration, it would be beneficial to select some deciduous hardwood species with bright forest floors and easy understory development. In this case, although we cannot provide data on soil carbon sequestration, similar levels of soil carbon sequestration would be expected, as litterfall amounts do not vary greatly among tree species.

5. Impact on soil organic carbon stocks

Soil surveys were conducted in 2001 and 2006 (Sakai *et al.*, 2010). Soil carbon in most soil depths at afforested sites showed the tendency of increase (slightly decreased in control Lawn site), but only the 0-5 cm increment showed a statistically significant increase in 5 years (Table 5). It is also predicted that soil carbon will increase even 30 years after planting.

Table 5. Evolution of SOC stocks at the study site in the 5-year study

Species	forest age at 1st – 2nd survey	Soil Depth (cm)	C stock at 1st survey (tC/ha)	Additional C storage in 5 years (tC/ha/yr)	Bulk density at 1st survey (kg/m^3)	Bulk density at 2nd survey (kg/m^3)
Japanese cedar	4-9	0-5	14.0	0.052	698	613
		5-15	21.1	0.321	616	628
		15-30	32.2	0.383	657	643
	14-19	0-5	16.0	0.286	657	619
		5-15	21.4	0.222	611	630
		15-30	29.5	0.358	605	608
	23-28	0-5	17.5	0.454	470	476
		5-15	16.0	0.125	540	563
		15-30	21.7	0.031	577	561
Hinoki cypress	4-9	0-5	13.1	0.219	706	741
		5-15	20.8	0.055	625	645
		15-30	29.3	0.189	633	629
	12-17	0-5	9.6	0.539	669	624
		5-15	6.8	0.012	548	596
		15-30	11.8	-0.497	645	647
	25-30	0-5	17.5	0.325	545	477
		5-15	19.2	0.090	624	607
		15-30	26.9	0.074	712	705
Managed as Lawn		0-5	14.3	-0.032	719	704
		5-15	21.3	-0.264	642	645
		15-30	30.4	-0.164	636	641

6. Other benefits of the practice

6.1. Benefits for soil properties

In general, after afforestation, dead leaves and branches begin to accumulate on the forest floor as the forest grows, and fine root biomass in surface soil also increases. These result in increased soil organic matter (soil carbon). The increase in organic matter in soil also leads to a decrease in soil bulk density. Our study resulted that this decrease in bulk density was noticeable at the surface 0-5 cm (Table 5). Also, the organic matter (carbon) in this forest floor reached a certain amount about 23-25 years after planting (Sakai *et al.*, 2010), but this increased organic matter can be counted as carbon sequestration amount. The presence of this forest floor organic matter mitigates raindrop impact and protects the topsoil, and the decrease in bulk density also helps rainwater infiltrate the soil surface and contributes to the stability of the topsoil. In addition, organic matter in forest floor contains organic nitrogen, which is a source of nutrients required for plant growth. Our study showed a clear carbon increase in surface soils with forest growth, but the increase in soil nitrogen was very small, indicating that some of the supplied nitrogen from litter was presumed to be used for tree growth (Sakai *et al.*, 2010).

In conclusion, for at least 30 years after planting, soil carbon sequestration occurs primarily in surface soils that are supplied with large amounts of dead organic matter. This was observed not only for Japanese cedar, which is said to have deep roots, but also for Hinoki cypress roots, which are said to have shallow roots, grow to a depth far exceeding 30 cm. Therefore, in Japanese cedar and Hinoki cypress forests that have a timber harvesting period of more than 40 to 50 years, root growth and mortality will increase with forest growth, and the depth of soil carbon sequestration will gradually increase.

6.2 Minimization of threats to soil functions

Table 6. Soil threats

Soil threats	
Soil erosion	The stability of the topsoil is considered to increase due to an increase in organic matter in the forest floor following afforestation (Miura, Yoshinaga and Yamada, 2003).
Nutrient imbalance and cycles	Nitrogen cycle is improved by litter supply and development of organic matter in forest floor (Sakai *et al.*, 2010).
Soil acidification	Soil pH was not measured in our study site. In Japan, Hinoki cypress plantations are likely to acidify soil, but it is known that Japanese cedar plantations do not acidify soil if soil lime saturation is high (Tanikawa *et al.*, 2014).

Soil threats	
Soil compaction	Bulk density in surface soil decreases with litter coverage and increased soil organic matter following afforestation (Table 5)

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

After the land use change administrative procedure, the goal of afforestation of idle farmland is timber production and soil conservation (Forest Agency, 2017). However, it should be noted that this site is only managed (thinning and updating) for research purposes.

6.4 Mitigation of and adaptation to climate change

Since no information is available on $CO_2/CH_4/N_2O$ emissions from soil, this study cannot accurately assess the benefits of afforestation of former farmlands for climate change mitigation. However, measurements of tree size, the amount of organic matter in forest floor, and soil carbon indicate that the net amount of carbon dioxide captured by the afforested sites has definitely increased (Sakai *et al.*, 2010).

6.5 Socio-economic benefits

In Japan, what is expected of afforested forests is that they will have multiple functions such as soil conservation at the growth stage, and that they will be used as timber resources after they reach the usable diameter (Ministry of Agriculture, Forestry and Fisheries 2016).

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 7. Soil threats

Soil threats	
Soil acidification	Soil pH was not measured in our study site. Afforestation of Hinoki cypress may promote acidification of soil (Tanikawa, Sobue and Hirano, 2014).

7.2 Increases in greenhouse gas emissions

No measurement data. The amount of N_2O produced in forests counts less than in farmlands where fertilizer is used (GIO, 2020). Most forests in Japan are also CH_4 sinks (Ishizuka, Sakata and Ishizuka, 2000, Morishita *et al.*, 2007).

7.3 Conflict with other practice(s)

Grazing and community farms are more commonly considered as the ways to use idle farmland (Ministry of Agriculture, Forestry and Fisheries, 2007).

7.4 Negative impact on production

In order to grow valuable planted forest, it is necessary to thin them and leave trees with good growth and characteristics. If proper forest management were not done, it may result in poor quality plantations (Forest Agency, 2017).

8. Recommendations before implementing the practice

Once afforested, it should be considered very difficult to return it to the farmland in terms of cost and labor. Especially in places where farmland conditions are good, with high productivity and easy access, it should be considered first to continue based on the "Act on Promotion of Improvement of Agricultural Management Foundation" (Ministry of Agriculture, Forestry and Fisheries, 2007).

Regarding the decision as to whether or not it is possible to plant trees in the target area, it is considered that the soil in the place that has been used as a field for many years does not become extremely dry or humid and maintains a certain productivity. Therefore, if the soil depth suitable for growing roots of trees is 70 cm or more, you can plant cedar, cypress, deciduous broad-leaved trees, etc. without problems. If your goal is timber production, it is good to select tree species in consideration of suitability for wet or dry of the field. For example, if the field you plan to plant is a little dry, Hinoki cypress would be recommended. If the field to be planted is from moderately to slightly over moist, Japanese cedar would be recommended.

It is also necessary to attention the management plan after planting trees. Especially for the purpose of harvesting timber, it is necessary to carry out weeding, thinning, and main cutting after planting trees. If you perform these tasks yourself, you will need to acquire the necessary skills. However, in Japan it is possible to consult or outsource such specialized work to forest cooperatives in the region.

9. Potential barriers for adoption

Table 8. Potential barriers to adoption

Barrier	YES/NO	
Social	Yes	Land owners may be required by local policy to make continuous efforts to use the land as agricultural land before planting it (Ministry of Agriculture, Forestry and Fisheries 2007).
Legal (Right to soil)	No	In principle, the conversion of farmland is prohibited in the agricultural promotion areas established by the prefectural governor (Ministry of Agriculture, Forestry and Fisheries 2015).

Photos

Photo 3. Hinoki Cypress, 25 years old

Photo 4. Japanese Cedar, 25 years old

Photo 5. Japanese Cedar, 4 years old

Photo 6. Hinoki cypress, 4 years old

Photo 7. Hinoki cypress, 12 years old

Photo 8. Japanese Cedar, 14 years old

Photo 9. Lawn site

References

Forest Agency, Ministry of Agriculture, Forestry and Fisheries. 2017. Annual Report on Forest and Forestry in Japan [online]. [Cited 26 June 2020]. (also available at: https://www.rinya.maff.go.jp/j/kikaku/hakusyo/29hakusyo/attach/pdf/index-1.pdf

)**Greenhouse Gas Inventory Office of Japan (GIO).** 2020. National Greenhouse Gas Inventory Report of Japan [online]. [Cited 24 June 2020]. http://www-gio.nies.go.jp/

Ishizuka, S., Sakata, T. & Ishizuka, K. 2000. Methane oxidation in Japanese forest soils. *Soil Biology and Biochemistry*, 32(6): 769-777. https://doi.org/10.1016/S0038-0717(99)00200-X

Ministry of Agriculture, Forestry and Fisheries. 2016. FY2015 Annual Report on Food, Agriculture and Rural Areas in Japan – Summary – [online]. [Cited 24 June 2020]. (also available at: https://www.maff.go.jp/e/data/publish/attach/pdf/index-38.pdf)

Ministry of Agriculture, Forestry and Fisheries. 2015. Article 4, Restrictions on Cropland Conversion in "Clopland Act" (Japanese Law Translation) [online]. [Cited 26 June 2020]. (also available at: http://www.japaneselawtranslation.go.jp/law/detail/?id=3174&vm=&re=).

Ministry of Agriculture, Forestry and Fisheries. 2007. Measures for idle farmland based on the Act on Promotion of Improvement of Agricultural Management Foundation in "Present state and problems of abandoned cultivated land" (in Japanese). [online]. [Cited 26 June 2020] (also available at: https://www.maff.go.jp/j/study/nouti_seisaku/senmon_04/pdf/data6.pdf)

Miura, S., Yoshinaga, S. & Yamada, T. 2003. Protective effect of floor cover against soil erosion on steep slopes forested with *Chamaecyparis obtusa* (hinoki) and other species. *Journal of forest research*, 8: 27-35. https://doi.org/10.1007/s103100300003

Morishita, T., Sakata, T., Takahashi, M., Ishizuka, S., Mizoguchi, T., Inagaki, Y., Terazawa, K., Sawata, S., Igarashi, M., Yasuda, H., Koyama, Y., Suzuki, Y., Toyota, N., Muro, M., Kinjo, M., Yamamoto, H., Ashiya, D., Kanazawa, Y., Hashimoto, T. & Umata, H. 2007. Methane uptake and nitrous oxide emission in Japanese forest soils and their relationship to soil and vegetation types. *Soil Science and Plant Nutrition*, 53: 678-691. https://doi.org/10.1111/j.1747-0765.2007.00181.x

Sakai, H., Inagaki, M., Noguchi, K., Sakata, T., Yatskov, M.A., Tanouchi, H. & Takahashi, M. 2010. Changes in soil organic carbon and nitrogen in an area of Andisol following afforestation with Japanese cedar and Hinoki cypress. *Soil Science and Plant Nutrition*, 56: 332-343. https://doi.org/10.1111/j.1747-0765.2010.00446.x

Tanikawa, T., Sobue, A. & Hirano, Y. 2014. Acidification processes in soils with different acid buffering capacity in *Cryptomeria japonica* and *Chamaecyparis obtusa* forests over two decades. *Forest Ecology and Management*, 334: 284-292. https://doi.org/10.1016/j.foreco.2014.08.036

3. Forest conservation through community management in the hill and plain areas of Nepal

Ram Asheshwar Mandal

School of Environment Science and Management, Baneshwor Kathmandu Nepal

1. Practice(s) used

Forest conservation, Agroforestry

2. Description of the case study

Nepal has over 40 years of experience with Community Forest (CF) management[1] (Maharajan *et al.*, 1990) (Table 9). This case-study introduces the first quantitative assessment of the effects of CF management in forests located in hilly and plain areas of Nepal on soil carbon (C), nitrogen (N), phosphorous (P) and potassium (K).

In Nepal, local users have been managing forests since 1978, applying different management practices, such as thinning, cleaning, pruning, planting, weeding and harvesting operations. Altogether, there are today over 23 000 CF in Nepal spread over more than 2 million hectares (DoFS, 2019), and the establishment of CF in the hilly areas has been one of the most successful forestry programmes in the country (Maskey *et al.*, 2006).

[1] A community forest is used, managed, and conserved by communities. Communities can have full, partial, or no ownership of such forests, and their management is often practiced in various degrees of collaboration with state forest agencies, donor organizations, knowledge institutions and/or companies (Arts and de Koning, 2017).

Table 9. History of community forest (CF) development in Nepal

Year	Policy related to CF	Purpose	Remarks
1976	National Forest Plan	Need of people's participation in forest management	
1978	Panchayat forest rules and Panchayat protected forest rules	Involvement of local people in forest management in village Panchayat	Local people participate to manage the forest and also to plant trees in open and degraded forest
1989	Master Plan for the Forestry Sector (MPFS)	CF management as first national (primary) programme	Streamlining the CF management in national priority
1993 & 1995	Forest Act (1993) & Forest Rules (1995)	To manage the forest by local people (District forest officer hand over the CF to local users)	Speed up the CF handover process Protect and develop the forest for product utilization
1995	CF Directives	Formation of user groups and handover process	Users group registration to Operational plan preparation
2004	CF inventory guideline	Manage the forest according to the condition of the forest	Assessment of growing stock and regeneration to determine the forest condition
2019	Forest Act	Sustainable management of CFs	

Although several studies were conducted to show the effect of management practices on biophysical conditions of the forest and socio-economic condition of the local people in Nepal (Pokharel and Larsen, 2007; Pandit and Bevilacqua, 2011), studies assessing the effect of CFM practices on forest soil carbon and fertility are limited so far.

For this study fourteen areas were selected to represent different models of community managed forests and pedo-climatic conditions:

1. In the plain area (Terai[2]):

- one buffer zone community forest (BZCF)[3] in the Terai plain (Bhudkaya BZCF, Bardia district);
- six community plantation forests of the Terai plain (Mahottari district): Shreepur, Banauta, Bisbitty, Sita, Ramnagar and Jogikuti.

2. In the hilly area:

- five CF of the mid hills: Pokhara Bhanjyang and Neware CF (Surkhet district), Nawadurga CF (Dang district), Banpala Danda CF (Khotang district) and Godawari Kunda CF (Lalitpur district);
- two catchments: Badekhola (Doti district) and Brindaban (Baitadi district)

In the hilly areas, the dominant species of selected CF are *Castanopsis indica, Schima wallichii, Alnus nepalensis* and *Fraxinus floribunda*. While some forest management practices have been applied since 2010 to manage Badekhola, no specific management has yet been applied in the Brindaban catchment. These two watersheds were also selected to evaluate the effects of management on soil texture.

In Terai (plain area), tree species are mostly *Shorea robusta*, *Terminalia alata* and *Buchanania latifolia*. In community plantation forests, trees (*Eucalyptus camaldulensis*) were planted in 2005. Agroforestry has been implemented in Shreepur community plantation forest and management practices were adopted in Sita community plantation forest.

3. Context of the case study

Climate varies from warm in Terai to mild in the hilly area (Table 10). In the frame of the study, selected areas were divided into a managed block where management practices were implemented by the users, and an unmanaged (control) block where no management practice was implemented. In each study site, soil samples were collected from both managed and unmanaged forest blocks at 0-10, 10-20 and 20-30 cm depths using soil corer (Table 10). C, N, P and K were analyzed using the Walkley and Black method (1958), the Kjeldahl Method (1883), Bray and Kurtz (1945) as well as Toth and Prince (1949) respectively. Soil texture was only assessed in Badekhola and Brindaban (Doti and Baitadi catchments) (Table 11).

[2] Terai is a lowland region in northern India and southern Nepal that lies south of the outer foothills of the Himalayas, the Sivalik Hills, and north of the Indo-Gangetic (Maharajan *et al.*, 1990).

[3] Buffer zone in Nepal means the peripheral area of national park/wildlife reserve where people have usufruct right on the resources (New ERA/UNDP, 2004). Buffer zone concept implies that the establishment of protected areas has measurable impact on adjoining areas and the people living there and vice-versa (New ERA/UNDP. 2004).

Table 10. Main characteristics of the studied areas

CF = Community forest

Location				Climate		Forest type	Major Species	Nb. samples	Reference
Zone	District	Type	GPS coordinates	Average min-max temperature (°C)	Annual rainfall (mm)				
Hills	Surkhet	CF	28°20' to 28°58' N 80°50' to 82°2' E	5–38	1 618	Natural	*Shorea robusta, Terminalia alata, Buchanania latifolia*	30	Kharel (2013)
	Dang		27º37' N to 28º29' N 82º02' E to 82º54' E	2.5 –39.9	1 706		*Shorea robusta, Terminalia alata, Adina cardifolia, Buchanania*	34	Adhikari (2015)
	Khotang		26º50' to 27º28' N 86º26" to 86º58" E	6.8–26.4	1 402		*Castanopsis indica, Schima wallichii, Alnus, fraxinus floribunda, Medhuka indica, Rhododendron arborium*	16	Rai (2014)
	Lalitpur		27°22' to 28°50' N 85°14' to 85°26' E	10.70–23.6	1 232		*Schima wallichii, Castonopsis indica, Myrica esculenta, Dendrocalamus spp. Alnus nepalensis*	20	KC (2012)
Plain (Terai)	Bardia	Buffer zone CF	28°07' to 28°39' N and 81°03' to 81°41' E 152-1457 m a.s.l	9.3–33.9	2 075		*Shorea robusta, Terminalia alata, Adina cardifolia, Buchanania latifolia, Syzigium cumini*	35	KC (2013)
Hills	Doti and Baitadi	Catchments	Badekhola: 29º17'23.20" N; 80º46'16.86" E Brindaban: 29º33'09.32" N; 80º43'13.60" E	4.1–27.1	1093		*Shorea robusta* (Sal), *Pinus roxburghii* (Salla), *Quercus spp* (Banjh)	20 (10 from each catchment)	Nepal (2015)
Plain (Terai)	Mahottari	Community plantations	26°36' to 28°10' N and 85°41' to 85°57' E	10.2–33.4	1309	Planta-tion	*Eucalyptus camaldulensis*	52	Mandal *et al.* (2010)

Table 11. Soil texture in the catchments area: Badekhola (Doti district) and Brindaban (Baitadi district)

Locations	Soil depth (cm)	Soil texture (%)			Textural class	Reference
		Sand	Silt	Clay		
Badekhola	0-10	26.4 ± 2.06	66.35 ± 1.47	8.1 ± 0.95	Silt	Nepal (2015)
Brindaban		28.2 ± 5.46	62.6 ± 4.4	9.2 ± 1.6	Silt	
Badekhola	10-30	31.64 ± 4.55	61.47 ± 3.99	6.9 ± 1.29	Silt	
Brindaban		37.5 ± 3.24	58.25 ± 2.74	3.55 ± 0.79	Silt	

4. Possibility of scaling up

In this case study, soil C, N, P and K were assessed in natural and plantations CF of Terai and mid hills of Nepal. These assessments are context, site and objective specific and limited to small areas. There are still research gaps related to the effects of management practices on forest soil carbon and fertility and hence, such research needs to be scaled up and include assessments of forest soil carbon and fertility according to forest types, geographical slope gradient and aspect, altitude and others. For instance, assessing soil carbon and fertility as well as their dynamics in Riverain forest, Mixed Sal Forest and degraded forest in Terai, pine forest, broad-leaved mixed forest of the mid hills and High Mountain, rangelands, wetlands and Ramsar sites, sustainably managed forest, private forests, natural and plantation forests are lacking and would be highly needed. Soil carbon and fertility dynamic analysis are unstudied research issues in Nepal.

5. Impact on soil organic carbon stocks

5.1 Effect of management practices on soil carbon in natural community forests (CF)

The management practices implemented by local users of CF are thinning, pruning, cleaning, planting, weeding, enrichment planting and harvesting operations. Research was conducted to assess the effect of management practices on soil carbon especially in CF in Surkhet, Dang, Khotang, Bardia and Lalitpur districts (Table 12 and Table 13). In all the natural CF, the recorded carbon stocks were higher in the managed than in the unmanaged blocks. In plantation forests, the highest carbon stocks were recorded in the alluvial soils of Shreepur and Sita plantation forests. It is likely that management practices lead to a better exposition of the forest cover to the sunlight, creating a suitable environment for microbial activities, resulting in higher soil carbon stocks.

Table 12. Soil organic carbon variation in managed and unmanaged (control) block in CF in Surkhet Dang Khotang and Bardia disctricts

District	CFs	Blocks	Soil type	Soil carbon according to soil depth (t/ha)			Reference
				0-10 cm	10-30 cm	30-60 cm	
Surkhet	Pokhara Bhanjyang & Neware CF	Managed	Entisol	11.8	10.2	8.0	Kharel (2013)
		Unmanaged		8.9	6.4	5.0	
Dang	Nawadurga CF	Managed	Alluvial	25.7	13.3	10.1	Adhikari (2015)
		Unmanaged		20.1	13.6	10.0	
Khotang	Banpala Danda CF	Managed	Entisol	18.6	15.8	9.1	Rai (2014)
		Unmanaged		15.2	11.0	6.2	
Bardia	Bhudkaya BZCF	Managed	Alluvial	23.4	12.8	10.8	KC (2013)
		Unmanaged		18.5	9.5	8.1	

Table 13. Soil organic carbon variation in managed and unmanaged (control) block of CF of Lalitpur district

District	CFs	Block	Soil types	Soil carbon according to soil depth (t/ha)		Reference
				0-20 cm	20-40 cm	
Lalitpur	Godawari Kunda CF	Managed	Alluvial	37.1	15.2	KC (2012)
		Unmanaged	Alluvial	35.6	15.2	

5.2 Soil carbon in the community plantation forests:

Carbon stocks varied according to the sites and soil depths (Table 14). The highest soil carbon stocks were recorded in Shreepur and Sita. Although the plantations were made in 2005 at all sites, their spatial heterogeneity may explain variations in SOC stocks. Specifically, Shreepur and Sita are both situated on alluvial soils, known to be fertile, while the other sites are located at riverbank where soils are sandy, and therefore less rich. Agroforestry is practiced in Shreepur with the culture of gram, beans and lentil, while in Sita, the management consists mainly in collecting grass, cleaning, weeding and climber cutting. Such practices are not applied in Bisbitty and Jogikuti, which may explain the lower SOC stocks.

Table 14. Soil carbon in the 6 community plantation forests (CPF)

CPF	Soil type	0-10 cm	10-30 cm	30-60 cm	Reference
		Soil carbon (t/ha)			
Shreepur	Alluvial	31.5	29.4	18.3	Mandal (2010)
Banauta	Sandy loam	15.2	15.2	6.9	
Bisbitty	Sandy	8.0	5.0	3.7	
Sita	Alluvial	28.4	26.5	16.5	
Ramnagar	Sandy loam	13.5	13.3	6.1	
Jogikuti	Sandy loam	7.3	4.7	3.5	

5.3 Soil carbon in the two catchments:

Soil carbon contents were found to be higher in Badekhola catchment where forest management practices like plantation, selection felling and thinning are applied (Table 15). The 2 sites are very close to each other, and their soils were initially degraded. Management practices were applied in order to improve soil conditions.

Table 15. Soil carbon in the two catchments at 0-10 and 10-30 cm depth (Doti and Baitadi districts)

District	Catchment	Soil types	Soil carbon according to soil depth (t/ha)		Reference
			0-10 cm	10-30 cm	
Doti	Badekhola	Alluvial	51.3 ± 6.9	37.2 ± 6.2	Nepal (2015)
Baitadi	Brindaban	Alluvial	41.6 ± 9.9	25.8 ± 5.7	

6. Other benefits of the practice

6.1. Benefits for soil properties

Soil properties were evaluated in Badekhola and Brindaban catchments only (Table 16). Higher contents of N, P and K in Badekhola catchment are most likely because of the application of management practices (i.e. site preparation, cleaning, plantation, weeding, selection felling, canopy opening for regeneration promotion) in comparison to Brindaban catchment where no management occurs (Table 16). These activities create an aeration condition that improves the soil health and fertility.

Table 16. Soil properties variation in catchment

Source: Nepal, 2015, data from Dadeldhura Nepal

Catchment	Soil depth (cm)	pH value	N (kg/ha)	P (kg/ha)	K (kg/ha)
Badekhola	0-10	5.7 ± 0.2	92.6 ± 12.3	43.3 ± 3.0	435.7 ± 98.9
Brindaban		6.1 ± 0.2	81.3 ± 9.1	10.7 ± 2.5	370.2 ± 81.4
Badekhola	10-30	6.0 ± 0.2	37.8 ± 6.2	16.4 ± 2.3	269.0 ± 79.2
Brindaban		6.0 ± 0.1	31 ± 4.4	8.4 ± 2.1	310.9 ± 82.1

6.2 Minimization of threats to soil functions

Table 17. Soil threats

Soil threats	
Soil erosion	The hilly areas of Nepal are highly vulnerable to soil erosion and soil loss that affect soil fertility, including soil carbon (Sidle *et al.*, 2006; Tarolli, Preti and Romano, 2015). The resultant effect of soil erosion, landslide, mass slide and bank cutting are not only onsite but also nearby and distance site as well (Farhan and Nawaiseh, 2015). Rivers in Nepal generally flow to Terai from North to South and originate from the Himalaya and Chure[4].

[4] Chure: The Sub-himalaya also known as the Muree, Chure Hills or Siwaliks, which are the southernmost foothills of the Himalayan Range and mainly composed of folded and overlapping sheets of sediment from the erosion of the Himalaya (Singh and Tandon, 2010).

Soil threats	
Nutrient imbalance and cycles	Community managed natural and plantation forests serve to add nutrients in the soil (Chen *et al.*, 2019). Litter and humus layers are a source of soil formation, but it depends on microbial activity. More microbial activity will lead to fostered nutrient cycles (Andersson, Kjoller and Struwe, 2004; Jandl *et al.*, 2007). This process supports the maintenance of balanced nutrients cycles (Prescott, Maynard and Laiho, 2000). The study conducted in different parts of Nepal validates this principle. The main finding was that there was addition in soil nutrients in community plantation forests and managed block of CF after application of management practices. The applied CF management practices support to increase of soil nutrients including soil carbon.

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

CF users have been using forest products (e.g. timber, firewood, fodder, litter and medicinal plants) and adopting the management practices to meet their daily needs (DoFSC, 2009).

6.4 Mitigation of and adaptation to climate change

Forest management practices play an important role in reducing GHG emissions. Healthy soils and forests can sequester carbon from the atmosphere. The Nepalese study showed that soil carbon stocks were higher in managed forest blocks than in the unmanaged (control) blocks.

6.5 Socio-economic benefits

The socio-economic benefit to the local people were not specifically assessed in the study. It is known however, that forest users sell forest products in order to generate their income. Besides, forest and soil carbon trading create good opportunities to generate incomes for the CF users. This study will be useful to promote carbon trading under the REDD+ mechanism[5] and payment for ecosystem services (Line, *et al.*, 2014; Muttaqin *et al.*, 2019). Moreover, the agricultural lands located close to the forest area use and benefit from the water coming from the forest. Farmers also prepare their mulches and manure collecting litter and grasses from the forest, which enhance their crop yields (Singh *et al.*, 2015). These are therefore direct and indirect socio-economic benefits to local farmers. Eventually, CF users benefit from an important social respect from the rest of the society (Poudel *et al.*, 2015).

[5] REDD+ is an international framework whose name stands for "reducing emissions from deforestation and forest degradation, conservation of existing forest carbon stocks, sustainable forest management and enhancement of forest carbon stocks". The REDD+ framework incentivises developing countries either to reduce GHG emissions or to increase the removal of carbon dioxide from the atmosphere by forest land.

6.6 Other benefits of the practice

This kind of study is important to increase the knowledge needed to meet the objectives set in the Paris agreement (2015), on atmospheric GHG reduction (Tanaka and O'Neill, 2018).

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

No tradeoffs were recorded as part of this study.

7.2 Increases in greenhouse gas emissions

CO_2 equivalent was highest in managed blocks in comparison to unmanaged blocks of forest (Table 18). The lower value of CO_2 equivalent in the unmanaged forests is explained by their low soil nutrients content (Soe and Buchmann, 2005; Blankinship, Niklaus and Hungate, 2011).

Table 18. CO_2 equivalent variation in community forests (CF) and community plantation forest

District	Forest name and typen (CF, BZCF, catchment, CPF)	CO_2 eq in block	
		Managed	Unmanaged
Surkhet	Pokhara Bhanjyang & Neware CFs	109.96	74.32
Dang	Nawadurga CF	179.81	160.16
Khotang	Banpala Danda CF	159.13	118.91
Lalitpur	Godawari Kunda CF	191.88	186.23
Bardia	Bhudkaya BZCF	172.29	132.18
Doti	Badekhola catchment	324.46	NA
Baitadi	Brindaban catchment	NA	247.28
Mahottari	Shreepur CPF	290.14	NA
	Sita CPF	261.51	NA
	Banauta CPF	NA	135.67

District	Forest name and typen (CF, BZCF, catchment, CPF)	CO_2 eq in block	
		Managed	Unmanaged
	Bisbitty CPF	NA	61.38
	Ramnagar CPF	NA	121.00
	Jogikuti CPF	NA	56.61

7.3 Conflict with other practice(s)

There is no conflict with other practices though there are several management practices applied to improve the growing stock of the forest. Sustainable forest management has been adopted by few CF users particularly in Terai area.

8. Recommendations before implementing the practice

The findings of this study will be a valuable reference and can be used as standard to evaluate carbon stock changes. The methodology applied to analyze the soil carbon and nutrient level will be useful for the scientific community and policy makers. However, the presented study was unable to cover soil carbon and nutrient status and dynamics according to forest types, management practices, geographical slope gradient and aspect, altitude and others in Nepal. Specifically, intensive research works are required related to soil carbon and fertility dynamics in Riverain forest, Mixed Sal Forest and degraded forest in Terai, pine forest and broad leaved mixed forest of mid hills and High Mountain, range land, wet land and Ramsar sites, sustainably managed forest, community managed forest, private forest, natural and plantation forest is crucial scope of scaling up of the study.

9. Potential barriers for adoption

Table 19. Potential barriers to adoption

Barrier	YES/NO	
Biophysical/ Natural resource	Yes	Limiting factors like disease, fire, overgrazing and over exploitation affect forest management practice which lead to affect the soil carbon and nutrients (Coulson and Stephen, 2008; Mayer *et al.*, 2020).
Cultural / Social	No	The products from forests are connected with the local people's income and daily need. They use the timber, firewood, fodder, litter produced from forest management practice. CF distribute the timber to the users who need to construct or renovate houses. The timber is distributed at reduced price or free of cost to the poor disadvantaged users or disaster affected family. Local users use firewood from the forest in the festival, ritual functions and wedding etc., or even to burn the bodies of the dead. Users collect the leaf to prepare the leaf plate, fodder for the domestic animals and also the medicinal herbs to treat the disease. So, CFs have a cultural and social importance.
Economic	No	The forest management practices have been creating the employment opportunity and hence the income generation. CF fund is used to support the poor people in order to uplift their livelihood.
Institutional	No	CF and community plantation forest users have their own constitution and operation management plan to manage the forests.
Legal (Right to soil)	Yes	According to government policy, users of CF and community plantation forests have the right to use forest products but they have no legal right to use the land or soil (Pokharel *et al.*, 2007).
Knowledge	Yes	CF and community plantation forest users have knowledge to run the institution, but knowledge of forest management is still lacking. Capacity building like training, workshop and seminar to manage the forest is basic requirement of forest users.

Photos

Photo 10. Establishment of plot in Godawari Kund community forest

Photo 11. Soil sample collection from Banpala community forests

Photo 12. Condition of the forest at Brindaban Catchment area

Photo 13. Status of Bhudkaya (BBZCF)

References

Adhikari, D. 2015. *Effects of silvicultural practices on Carbon Stock in Shorea robusta Forests* (A Case Study from Nawadurga CF, Dang district, Nepal). A Project paper submitted for the partial fulfillment of Bachelor of Science in Forestry degree, Tribhuvan University, Kathmandu Forestry College, Kathmandu Nepal.

Andersson, M., Kjoller, A. & Struwe, S. 2004. Microbial enzyme activities in leaf litter, humus and mineral soil layers of European forests. *Soil Biology and Biochemistry*, 36(10): 1527-1537. https://doi.org/10.1016/j.soilbio.2004.07.018

Arts, B. & de Koning, J. 2017. Community Forest Management: An Assessment and Explanation of its Performance Through QCA. *World Development*, 96: 315–325. https://doi.org/10.1016/j.worlddev.2017.03.014

Bhandari, K., Parida, P. & Singh, P. 2013. Estimation of carbon footprint of fuel loss due to idling of vehicles at signalised intersection in Delhi. *Procedia-Social and Behavioral Sciences*, 104: 1168-1177. https://doi.org/10.1016/j.sbspro.2013.11.213

Blankinship, J.C., Niklaus, P.A. & Hungate, B.A. 2011. A meta-analysis of responses of soil biota to global change. *Oecologia*, 165(3): 553-565. https://doi.org/10.1007/s00442-011-1909-0

Bray Roger, H. & Kurtz, L.T. 1945. Determination of Total Organic and Available Forms of Phosphorus in Soils. *Soil Science*, 59(1): 39–46. https://doi.org/10.1097/00010694-194501000-00006

Chen, C., Chen, H.Y., Chen, X. & Huang, Z. 2019. Meta-analysis shows positive effects of plant diversity on microbial biomass and respiration. *Nature communications*, 10(1): 1-10. https://doi.org/10.1038/s41467-019-09258-y

Coulson, R.N. & Stephen, F.M. 2008. Impacts of insects in forest landscapes: implications for forest health management. *In Invasive forest insects, introduced forest trees, and altered ecosystems*. pp. 101-125. Springer, Dordrecht.

DoFS. 2019. *Hamro Ban*. Annual report produced by Department of Forest and Soil Conservation, Babarmahal Kathmandu Nepal. Ministry of Forest and Environment. pp. 45-55.

DoFSC. 2009. *Community forest directives*. Department of Forest and Soil Conservation. Kathmandu Nepal

Farhan, Y. & Nawaiseh, S. 2015. Spatial assessment of soil erosion risk using RUSLE and GIS techniques. *Environmental Earth Sciences*, 74(6): 4649-4669. https://doi.org/10.1007/s12665-015-4430-7

Jandl, R., Lindner, M., Vesterdal, L., Bauwens, B., Baritz, R., Hagedorn, F. & Byrne, K.A. 2007. How strongly can forest management influence soil carbon sequestration? *Geoderma*, 137(3-4): 253-268. https://doi.org/10.1016/j.geoderma.2006.09.003

KC, A. 2013. *Assessment of carbon stocks in managed and unmanaged block of Bhudkaya Community Forest (Buffer Zone Community Forest of Bardia National Park)*. Golden Gate International College Tribhuvan University Affiliate, Kathmandu, Nepal

KC, S. 2012 *Assessment of Carbon Stocks in Broad Leaved Forest, A Case Study from Godawari Kunda Community Forest, Lalitpur District, Nepal).* A Project paper submitted for the partial fulfillment of Bachelor of Science in Forestry degree, *Tribhuvan University, Kathmandu Forestry College, Kathmandu, Nepal*

Kharel, B. 2013. *Comparative Study of Forest Carbon Stocks in Community Forests Managed by Other than Women Members*. (A case study from two community forests of Surkhet district). A Project paper submitted for the partial fulfillment of Bachelor of Science in Forestry degree, *Tribhuvan University, Kathmandu Forestry College, Kathmandu, Nepal*

Maharjan, M., Sanaullah, M., Razavi, B.S. & Kuzyakov, Y. 2017. Effect of land use and management practices on microbial biomass and enzyme activities in subtropical top-and sub-soils. *Applied Soil Ecology*, 113: 22–28. https://doi.org/10.1016/j.apsoil.2017.01.008

Maskey, R.B., Sharma, B.P. & Joshi, M. 2003. Human Dimensions in Sustainable Land Use Management in Degraded Land Areas of Nepal. A Paper prepared for presentation at the Open Meeting of the Global Environmental Change Research Community, Montreal, Canada, 16-18 October 2003

Maskey, V., Gebremedhin, T.G. & Dalton, T.J. 2006. Social and cultural determinants of collective management of community forest in Nepal. *Journal of forest economics*, 11(4): 261-274. https://doi.org/10.1016/j.jfe.2005.10.004

New ERA/UNDP. 2004. *Impact assessment of buffer zone programme in Nepal.* Kathmandu: New ERA. pp. 34-40.

Kjeldahl, J. 1883. Neue Methode zur Bestimmung des Stickstoffs in organischen Körpern (New method for the determination of nitrogen in organic substances). *Zeitschrift für analytische Chemie*, 22(1): 366-383.

Lal, R. 2008. Sequestration of atmospheric CO_2 in global carbon pools. *Energy & Environmental Science*, 1(1): 86-100. https://doi.org/10.1039/B809492F

Lin, L., Sills, E. & Cheshire, H. 2014. Targeting areas for reducing emissions from deforestation and forest degradation (REDD+) projects in Tanzania. *Global Environmental Change*, 24: 277-286. https://doi.org/10.1016/j.gloenvcha.2013.12.003

Maharajan, P.L., Bhadra, B., Roy, P., Yadav, R.P. & Rongsu, Z. 1990. Environmental diversity and its influence on farming systems in the Hindu Kush-Himalayas. *In Mountain Agriculture and Crop Genetic Resource.* New Delhi, India: Oxford & IBH Publishing CO. PVT. LTD, 9-42.

Mayer, M., Prescott, C.E., Abaker, W.E., Augusto, L., Cecillon, L., Ferreira, G.W. & Laganiere, J. 2020. Influence of forest management activities on soil organic carbon stocks: A knowledge synthesis. *Forest Ecology and Management*, 466: 118127. https://doi.org/10.1016/j.foreco.2020.118127

Muttaqin, M.Z., Alviya, I., Lugina, M. & Hamdani, F.A.U. 2019. Developing community-based forest ecosystem service management to reduce emissions from deforestation and forest degradation. *Forest policy and economics*, 108: 101938. https://doi.org/10.1016/j.forpol.2019.05.024

Nepal, S. 2015. *Soil Quality Index and Nutrient in Badekhola and Brindaban Catchments, Nepal, (A case study from two community forests of Surkhet district).* A Project paper submitted for the partial fulfillment of Bachelor of Science in Forestry degree, Tribhuvan University, Kathmandu Forestry College, Kathmandu, Nepal.

Pandit, R. & Bevilacqua, E. 2011. Forest users and environmental impacts of community forestry in the hills of Nepal. *Forest Policy and Economics*, 13(5): 345-352. https://doi.org/10.1016/j.forpol.2011.03.009

Pokharel, R.K. & Larsen, H.O. 2007. Local vs official criteria and indicators for evaluating community forest management. *Forestry*, 80(2): 183-192. https://doi.org/10.1093/forestry/cpm005

Pokharel, B.K., Branney, P., Nurse, M. & Malla, Y.B. 2007. Community forestry: Conserving forests, sustaining livelihoods and strengthening democracy. *Journal of Forest and Livelihood*, 6(2): 8-19.

Poudel, M., Thwaites, R., Race, D. & Dahal, G.R. 2015. Social equity and livelihood implications of REDD+ in rural communities–a case study from Nepal. *International Journal of the Commons*, *9*(1). http://doi.org/10.18352/ijc.444

Prescott, C.E., Maynard, D.G. & Laiho, R. 2000. Humus in northern forests: friend or foe? *Forest ecology and management*, 133(1-2): 23-36. https://doi.org/10.1016/S0378-1127(99)00295-9

Rai, S. 2014. *Effect of Silvicultural Treatments on Carbon Stock (A Case Study from Shima-castonopsis Forest of Banpala Community Forest, Khotang District, Nepal).* A Project paper submitted for the partial fulfillment of Bachelor of Science in Forestry degree, Tribhuvan University, Kathmandu Forestry College, Kathmandu Nepal.

Sidle, R.C., Ziegler, A.D., Negishi, J.N., Nik, A.R., Siew, R. & Turkelboom, F. 2006. Erosion processes in steep terrain—truths, myths, and uncertainties related to forest management in Southeast Asia. *Forest Ecology and Management*, 224(1-2): 199-225. https://doi.org/10.1016/j.foreco.2005.12.019

Singh, V. & Tandon, S.K. 2010. Integrated analysis of structures and landforms of an intermontane longitudinal valley (Pinjaur dun) and its associated mountain fronts in the NW Himalaya. *Geomorphology*, 114(4): 573-589. https://doi.org/10.1016/j.geomorph.2009.09.019

Soe, A.R. & Buchmann, N. 2005. Spatial and temporal variations in soil respiration in relation to stand structure and soil parameters in an unmanaged beech forest. *Tree physiology*, 25(11): 1427-1436. https://doi.org/10.1093/treephys/25.11.1427

Tanaka, K. & O'Neill, B.C. 2018. The Paris Agreement zero-emissions goal is not always consistent with the 1.5 °C and 2 °C temperature targets. *Nature Climate Change*, 8(4): 319. https://doi.org/10.1038/s41558-018-0097-x

Tarolli, P., Preti, F. & Romano, N. 2014. Terraced landscapes: From an old best practice to a potential hazard for soil degradation due to land abandonment. *Anthropocene*, 6: 10-25. https://doi.org/10.1016/j.ancene.2014.03.002

Toth S.J. & Prince A.L. 1949. Estimation of cation- exchange capacity and exchangeable Ca, K and Na contents of soils by flame photometer techniques. *Soil Science*, 67: 435-439.

Tyson, R.V., Simonne, E.H., White, J.M. & Lamb, E.M. 2004. Reconciling water quality parameters impacting nitrification in aquaponics: the pH levels. *In Proceedings of the Florida State Horticultural Society*, 117: 79-83.

Walkley, A.E. & Black, J.A. 1958. An Examination of the Method for Determining Soil Organic Method, and Proposed Modification of the Chromic Acid Titration Method. *Soil Science,* 37: 29-38.

4. Soil organic carbon stocks in forests of Singapore

Ernst Leitgeb[1], Michael Kleine[2], Hassan bin Ibrahim[3], Mohamad Fairoz bin Mohamed[3], Denise Chng Pei Lin[4], Mohamed Lokman bin Mohd Yusof[5], Kerstin Michel[1], Subhadip Ghosh[5]

[1]Department of Forest Ecology and Soil, Austrian Research Centre for Forests, Vienna, Austria

[2]Austrian Natural Resources Management and International Cooperation Agency (ANRICA), Vienna, Austria

[3]International Biodiversity Conservation, National Parks Board, Singapore

[4]Parks Central, National Parks Board, Singapore

[5]Centre for Urban Greenery and Ecology, National Parks Board, Singapore

1. Related practices and hot-spots

Forest conservation; Forests

2. Description of the case study

In 2013, Singapore began to establish a Monitoring/Measuring, Reporting and Verification System (MRV) for the Land Use, Land Use Change and Forestry (LULUCF) sector as part of its reporting obligations of greenhouse gases (GHG) to the United Nations Framework Convention on Climate Change (UNFCCC). This interdisciplinary approach follows the 2006 IPCC Guidelines for National GHG Inventories (IPCC, 2006) and utilizes within its GHG inventory expertise in remote sensing, assessment of aboveground biomass and of SOC, and quality assurance towards reporting of the GHG inventory. Remote sensing techniques are used to collect activity data on respective land-use categories as prescribed in the IPCC guidelines, such as Forest Land, Cropland, Wetlands and Settlements, as well as sub-categories that are nationally defined. Ground-truthing exercises of the national forest inventory in selected sampling plots of key sub-categories of Forest Land and Settlements are conducted at a five-year interval as part of the MRV process. In these plots, data on above-ground biomass (AGB) and on soil organic carbon (SOC) were collected. However, while AGB data could be collected during each ground-truthing exercise, changes in SOC stocks cannot be assessed by a short-time repetition of the soil inventory due to the large variation of soil carbon in the field. Hence, soil carbon stocks are assessed only once to serve as a baseline and for estimating soil carbon changes, soil carbon models are used. The methods of soil sampling and of soil analyses are well documented (Leitgeb *et al.*, 2018).

The soil inventory in Singapore forests provides valuable insights as there is little information about soils and their characteristics available. Although few studies have been conducted to quantify the SOC distribution and assess the soil quality of roadside soils and their influence on street tree performance (Ghosh, Scharenbroch and Ow, 2016; Ghosh *et al.,* 2016), but the majority of the soil studies in Singapore focused on civil engineering applications such as foundations, slope stability and other development projects. A comprehensive study, dealing with concentrations of soil nutrients and trace elements in forests of Singapore was carried out by Leitgeb *et al.* (2019) utilizing the large soil data set of the soil inventory.

3. Context of the case study

Singapore is a city-state with a land area of approximately 725.7 km^2 located at the southern tip of the Malay Peninsula (Department of Statistics, 2019). The climate is tropical with abundant rainfall and high humidity. The climate is characterized by two monsoon seasons - the Northeast Monsoon (December to early March) and the Southwest Monsoon (June to September) separated by inter-monsoonal periods. Singapore's 1981-2010 long-term average daily temperature was around 27.5 °C, with an average annual rainfall of 2 166 mm (Department of Statistics Singapore, 2019). In the humid tropics, soils are generally poor in nutrients due to intensive weathering which is responsible for higher decomposition of organic matter and removal of elements (Ghosh *et al.*, 2019). The prevailing soil types in Singapore are Ultisols and Oxisols and soils developed on igneous and on sedimentary rocks are characterized as acidic with low cation exchange capacity (Leitgeb *et al.*, 2019). In some of the sampling plots, high pH values and high nitrogen concentrations gave evidence of anthropogenic influence (e.g. fertilization).

The current landscape in Singapore is highly fragmented and dominated by urban environment. According to Corlett (1991), the vegetation in Singapore can be classified into primary forests, secondary forests consisting of low and tall trees and herbaceous vegetation, as well as managed vegetation such as parks, gardens and plantations and in intertidal vegetation like mangroves. Primary lowland forests, however, cover only 0.28 percent of Singapore's total area (Yee *et al.*, 2011), which are located mainly in the nature reserves such as the Bukit Timah Nature Reserve and the Central Catchment Nature Reserve. Today, the most of the forest areas in Singapore are covered by secondary forests of different successional stages, including in areas that are abandoned from past agricultural cultivation and resettlement of villages. In such areas, the vegetation is in the process of recovery both in terms of AGB and sequestration of soil organic carbon.

In this study, soil organic carbon stocks in forests were analyzed. Forests in Singapore were classified into 4 classes: primary forests, secondary forests representing natural forest succession, secondary forests after tree plantation/fruit orchard and secondary forests after agriculture crop cultivation.

Class 1: Primary forests

The forest stands are dominated by late succession native species (> 80 percent) with a total AGB ranging from 261 to 414 t/ha, which is typical for these lowland primary forests. The vegetation here has not been subjected to timber extraction nor been negatively affected by excessive weather events (e.g. heavy rainstorm and strong winds), thus could largely keep its primary forest structure and composition.

Class 2: Secondary forests representing natural forest succession

These forests are the result of heavy disturbances in the past. Plots within these forests contain scattered large native remnant trees of late succession stages as well as long-lived pioneers. This indicates that the forest has been affected by selective land clearance activities at different intensities, but most likely the areas have not been converted for purpose of agriculture with non-perennial crops or exotic tree plantations.

Class 3: Secondary forests after tree plantation/fruit orchard

These forests are regenerated vegetation from abandoned tree plantations and fruit orchards dominated by exotic species such as rubber, coffee or fruit trees. Such stands show a completely different species composition compared to secondary forests following natural forest succession as described above.

Class 4: Secondary forests after agriculture crop cultivation

These plots represent forest stands that are located in rather flat terrain and have been subjected to intensive agriculture of mainly annual crops like vegetables in the early days of settlements. Today these forests are composed of common pioneer native trees and many non-native tree species.

The objectives of this study are to compare the SOC stocks of forest soils (litter layer and mineral soil up to 50 cm soil depth) of the different forest classes and to evaluate the impact of historic land use on the carbon sequestration potential in forest soils.

4. Possibility of scaling up

The statistical design of the IPCC reporting system, a random grid system, was focused on scaling up for the entire area of Singapore. However, there are inherent limitations when it comes to estimating the SOC stocks. Due to the highly fragmented forested vegetation in Singapore, it is difficult to get a sufficient number of replicates for each forest class, in particular for primary forests, where data from only 2 plots could be collected and examined. To address carbon sequestration in forest soil, denser sample grids would be appropriate. Furthermore, for the IPCC compliant reporting a limited soil depth up to 50 cm was adequate. Studies by Ngo *et al.* (2013) on the primary forests of Singapore showed that the SOC pool comprised a soil depth of 3 m.

5. Impact on soil organic carbon stocks

We assume that the soil carbon stocks in primary forests represent the potential of carbon sequestration and that this potential was gradually lowered by anthropogenic impact, like land clearance and development, or conversion to agricultural practices. In Table 20, the soil carbon stocks from all plots of the inventory are presented allowing an overview over the variability of the stocks, while Table 21 shows the carbon stocks according to the forest classes mentioned above. In nine of the cases, it was not possible to assign the inventory plots to forest classes, but consolidated data has been included in the Table 21.

Table 20. SOC stocks (in t/ha) in the forest inventory plots (n=30)

Sample location	Mean	Std dev	Min	Max
Litter layer	6.8	6.6	0.3	32.8
Mineral soil 0-50 cm	74.6	39.2	25.0	218.0

The carbon stocks in Table 20 demonstrate the high heterogeneity in both the litter layer and mineral soil up to 50 cm soil depth. A high variety in the litter layer is common in forest ecosystems, but also the broad range of carbon stocks in the mineral soil is considerable. As the soil conditions within this soil inventory (mostly infertile Ultisols and Oxisols) are to some extent comparable, these differences are most likely caused by some outliers. For instance, the maximum value of 218 t/ha was found in a water-logged soil (Histosol) in an offshore island in the north of Singapore. Such cases could not be classified and were omitted subsequently. Table 21 shows the carbon stocks according to the four forest classes. From the 30 inventory plots, 21 plots could be assigned to forest classes, but data on SOC are only available for 20 plots. Unfortunately, the number of plots in the respective forest classes is rather unbalanced, especially in Class 1. The small number of replicates prevents detailed statistics.

Table 21. SOC stocks (in t/ha) in the different forest classes

n = number of plots.

In classes with only 2 plots (Class 1 and 4) standard deviations were not applicable (NA)

	Mean	Std dev	Min	Max
Class 1 (n=2)				
Litter layer	8.2	NA	7.2	9.3
Mineral soil 0-50 cm	105.7	NA	83.7	127.7
Class 2 (n=8)				
Litter layer	11.7	10.3	2.6	32.8
Mineral soil 0-50 cm	85.2	18.8	58.5	112.8
Class 3 (n=8)				
Litter layer	3.2	1.3	1.8	5.3
Mineral soil 0-50 cm	57.7	17.1	26.0	80.1
Class 4 (n=2)				
Litter layer	5.1	NA	2.4	10.1
Mineral soil 0-50 cm	30.1	NA	25.0	35.2

The carbon stocks in the litter layer of all forest classes showed no specific pattern and were highly variable due to the high heterogeneity of soil organic accumulation in the forest litter. In contrast, the range of carbon stocks in the mineral soil up to 50 cm depth showed a clear trend for Class 3 and 4. Although the maximum SOC stock (127.7 t/ha) was found in Class 1, the low number of plots in this Class did not allow a clear differentiation to SOC stocks in secondary forests with natural succession (Class 2), showing a range from 58.5 t/ha to 112.8 t/ha. Ngo *et al.* (2013) compared SOC stocks in primary and secondary forests in Bukit Timah, Singapore, and found 110.8 t SOC/ha in primary forests and 143.2 t SOC/ha in secondary forests. Secondary forests after tree plantation/fruit orchard (Class 3) and secondary forests after agriculture crop cultivation (Class 4) showed depleted SOC stocks, ranging from 26.0 to 80.1 t/ha and 25.0 to 35.2 t/ha, respectively. This indicates the effect of land disturbances like agricultural practices on soil organic carbon.

Despite the small number of sample plots selected, the impact on soil carbon stock from past land management practices could be demonstrated. However, it is recommended that further research could be undertaken to arrive at detailed perspective of the soil carbon sequestration in tropical forests.

6. Other benefits of the practice

The assessment approach presented here contributes to improved estimation of SOC baselines and to an approximation of the long-term potential for SOC recovery through forest conservation.

6.1. Benefits for soil properties

This study was carried out in forested areas only. Forest conservation and natural forest regeneration respectively foster soil properties for tree growth, especially in maintaining the nutrient cycling in tropical forest soils, which are mostly low in nutrients (Leitgeb *et al.*, 2019). Biological soil properties like biological activity play an important role in this context.

6.2 Minimization of threats to soil functions

Soils in the remnant forests in Singapore play an important role for forest conservation and for providing slope stability as well as protection against soil erosion. This is of particular importance of forests in the water catchment areas of the nature reserves. There are strict conservation regulations in place in order to maximize the socio-economic benefits of these forests in a highly urbanized country.

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

There is no forestry industry in Singapore. The forests are managed for conservation, recreation, research, and educational purposes.

6.4 Mitigation of and adaptation to climate change

According to Singapore's IPCC-compatible GHG inventory the forests in Singapore serve as a carbon sink. Though the contribution from this carbon sink is small as compared to the total GHG emissions from other sectors, we continue to strengthen our climate resilience by safeguarding and enhancing our natural ecosystems that provide carbon storage and sequestration. Aside from that, enhancing extensive greenery through a network of nature reserves, nature parks, and streetscapes will strengthen the ecological resilience and help to reduce impacts of climate change.

6.5 Socio-economic benefits

The main benefits of forests to the society of Singapore include:

- Recreation for people providing a natural environment for nature walks, exercises, etc.;
- Conservation of tropical biodiversity, in particular of endangered flora and fauna;
- Nature and environmental education to the public including to schools;
- Improvement of knowledge on biodiversity and the natural environment through field surveys and scientific research including from forest inventory and monitoring.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

No tradeoffs recorded.

7.2 Increases in greenhouse gas emissions

For forest land, net GHG balance = -7.91 Gg CO_2eq (as reported in Singapore's Fourth National Communication and Third Biennial Update Report under the UNFCCC, NEA 2018).

7.3 Conflict with other practice(s)

With increasing urbanization, cities face problems of urban sprawling resulting in loss of natural vegetation and face challenges to maintain adequate green spaces. However, despite the high population density, Singapore is dedicated to improve the liveability for its citizens as well as environmental sustainability by adopting initiatives that also promote biodiversity conservation and protection of green spaces.

8. Recommendations before implementing the practice

Forest soils are characterized by an efficient nutrient cycling and high organic matter content. The vegetation helps to stabilize the soil and protect it from degradation. Therefore, no soil disturbance and removal are recommended when habitat enhancement or forest rehabilitation is undertaken.

The SOC recovery through forest conservation is a long-term process; the recovery could be faster through active reforestation. However, there are no areas available for large-scale reforestation. Instead diversity of trees is enhanced through small-scale enrichment planting of native species – and in this way promoting natural recovery and successional processes with the aim to create, enhance and maintain diverse and resilient forest ecosystems.

9. Potential barriers

Barriers to forest conservation are mainly related to urban sprawling resulting in a reduction of forest area and multiple use of the forest areas. However, strict nature conservation policies are put in place to ensure that core forests areas will be conserved. Efforts are made to ensure that the forests are sustainably managed for continued multiple use and to minimize environmental damage. These include regulating the number of visitors and research permits as well as ensuring robust environmental impact assessments for developments affecting the forests.

As a city state, Singapore is one of the most densely populated countries in the world. With no hinterland, the key challenge is to integrate nature into economic development. This requires a pragmatic approach in balancing development and biodiversity conservation, finding unique solutions to create a nature conservation model that champions environmental sustainability in a small urban setting through implementation of a Nature Conservation Master Plan. Aside from forest restoration and safeguarding key habitats, an important factor towards the long-term success in conserving this natural heritage is community involvement to ensure that both people and nature can co-exist in harmony. Ultimately, this is part of Singapore's vision to transform the country into a City in Nature and to ensure greening of the city increases even as it further develops.

Photo

Photo 14. Soil pit for sample collection in a forested area in Singapore

Acknowledgements

This study on Singapore soils was part of a national project to report on carbon stocks and fluxes from the land use and vegetation sector of Singapore. This project was funded by the Government of Singapore and administered by the National Parks Board (NParks). The work was coordinated and carried out by the Austrian Natural Resources Management and International Cooperation Agency (ANRICA).

References

Corlett, R.T. 1991. Vegetation. In L.S. Chia, A. Rahman & D.B.H. Tay (Eds.) *The Biophysical Environment of Singapore*. pp. 134-154. Singapore, Singapore University Press.

Department of Statistics Singapore, 2019. Yearbook of Statistics Singapore. [online]. [cited 20 February 2020]. https://www.singstat.gov.sg/-/media/files/publications/reference/yearbook_2019

Ghosh, S., Deb, S., Ow, L.F., Deb, D. & Yusof, M.L. 2019. Soil characteristics in an exhumed cemetery land in central Singapore. *Environmental Monitoring and Assessment,* 191: 174. https://doi.org/10.1007/s10661-019-7291-9

Ghosh, S., Scharenbroch, B. & Ow, L.F. 2016a. Soil organic carbon distribution in roadside soils of Singapore. *Chemosphere,* 165: 163-172. https://doi.org/10.1016/j.chemosphere.2016.09.028

Ghosh, S., Scharenbroch, B., Burcham, D., Ow, L.F., Shenbagavalli, S. & Mahimairaja, S. 2016b. Influence of soil properties on street tree attributes in Singapore. *Urban Ecosystems* 19: 949-967. https://doi.org/10.1007/s11252-016-0530-8

IPCC. 2006. *Guidelines for nation greenhouse gas inventories*. Eggelston, S., Buendia, L., Miwa, K., Ngara, T. & Tanabe, K. (Eds.) Institute for Global Environmental Strategies (IGES), Japan. [online]. [cited 10 June 2020]. https://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html

Leitgeb, E., Jandl, R., Englisch, M. & Weiss, P. 2018. Soil Carbon Inventory - Data Management, Processing and Modelling Manual. In L. Chan & M. Sommerauer (Eds.) *Singapore LULUCF Greenhouse Gas Reporting. Technical Handbook*, Vol. 1, chapter 3. National Parks Board, Singapore (unpublished).

Leitgeb, E., Ghosh, S., Dobbs, M., Englisch, M. & K. Michel. 2019. Distribution of nutrients and trace elements in forest soils of Singapore. *Chemosphere*, 222: 62-70. https://doi.org/10.1016/j.chemosphere.2019.01.106

National Environmental Agency. 2018. Singapore's Fourth National Communication and Third Biennial Update Report, Under The United Nations Framework Convention on Climate Change, December 2018. (also available at: https://www.nea.gov.sg/our-services/climate-change-energy-efficiency/climate-change/national-communications-and-biennial-update-reports)

Ngo, K.M., Turner, B.L., Muller-Landau, H.C., Davies, S.J., Larjavaara, M., Nik Hassan, N.F. bin & Lum, S. 2013. Carbon stocks in primary and secondary tropical forests in Singapore. *Forest Ecology and Management*, 296: 81–89. https://doi.org/10.1016/j.foreco.2013.02.004

Yee, A.T.K., Corlett, R.T., Liwe, S.C. & Tan, H.T.W. 2011. The vegetation of Singapore – an updated map. *Gardens' Bulletin Singapore*, 63: 205-212.

5. Reforestation of highlands in Javor Mountain, Republic of Srpska, Bosnia and Herzegovina

Marijana Kapović Solomun[1], Carlos Cruz-Gaistardo[2]

[1]University of Banja Luka, Faculty of Forestry, Department of Forest ecology, Banja Luka, Republika Srpska, Bosnia and Herzegovina

[2]National Institute of Geography and Statistical. Soils Department, Mexico (retired)

1. Related practices and hot-spots

Afforestation and reforestation; Mountain soils, Forests

2. Description of the case study

The annual estimated erosion rate in Bosnia Herzegovina is 9.88 t/ha (Tošić, Dragićević and Lovrić, 2012). This value represents a removed sheet of 1.3 cm of superficial soil in a period of 15 years. Javor Mountain in the northwest of the country is a relatively preserved area, with an average erosion rate of 0.46 t/ha/yr, and a maximum erosion rate of 3.73 t/ha/yr (Figure 2). However, the variability of soil organic carbon stocks is significant (11.2 to 283.3 tC/ha in first 30 cm) due to geomorphological, climatic and anthropogenic changes almost always occurring in combination (Figure 1). In this case study, we investigated the effect of reforestation of deforested areas subjected to erosion with species such as *Picea abies*, *Abies alba*, *Pinus nigra*, *Pinus sylvestris* and *Fagus sylvatica*, because these may have a triple effect: improving the production potential of the soil, providing better protection against erosion (especially on recently deforested steep slopes) and converting scrubland into high-quality forest.

3. Context of the case study

The research was realized in southeast Europe, Balkan Peninsula covering area of 24 100 ha, known as the region of Javor Mountain, Republika Srpska (Bosnia and Herzegovina), at 1 000 to 1 500 meters of altitude.

The average annual temperature of this region is 5.6 °C, with an annual precipitation of 1 010 mm and mountain climate that becomes perhumid on the highest altitude (Kapovic, 2013).

The main soil groups in this region are the Podzols, Cambisols, Luvisols, Acrisols, Phaeozems and Leptosols (IUSS working group WRB, 2015). The soils of this region are generally loamy (21.1 ± 8.4 percent of clay), with a moderately developed structure, slightly stony (less than 20 percent of the total weight) and moderately acid (pH 5.4 ± 0.8), except in the sedimentary region in the south part of Javor (pH 7.3 ± 0.2).

4. Possibility of scaling up

Diverse geomorphological and edaphic conditions evaluated during the study allow the results to be scaled to a regional level, under similar pedo-climatic and edaphic conditions, i.e. loamy soils with similar erosion rates, as Dinari, Balkan or Panonian cool temperate moist forest mountains environments mainly in Bosnia but also in some bordering areas with Serbia.

5. Impact on soil organic carbon stocks

Table 22. SOC changes in the Javor Mountain (Republic of Srpska, Bosnia and Herzegovina) after 15 years of reforestation, at 0-30 cm depth

Adapted from Kapovic Solomun (2013)

Climate is sub-alpine temperate continental, according to the IPCC. **SD**: standard deviation.

Soil type	Baseline C stock ± SD (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Year)	More information
Neutral soils (pH 6.5–7.6) with thick organic layers. Histosols and hiperhumics Leptosols.	198.9 ± 66.9	0.10	15	Soils without apparent degradation, with arboreous cover greater than 80 percent.
Very acid soils (pH 4.2–4.5), with accumulation of humus and / or oxides in the lower layers. Albic Podzols.	32.8 ± 16.3	0.66		Soils poor in phosphorus and rich in aluminum. They are not attractive for conventional agriculture.
Acid soils (pH 4.6–5.9), with migration of clays and cations towards the lower horizons.	85.5 ± 34.9	1.10		Soils with the largest total area in the Javor Mountain region. The greatest potential for recovery in these soils is found

Soil type	Baseline C stock ± SD (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Year)	More information
Humic Acrisols, hyperdistric Luvisols and humic Cambisols, Umbrisols.				in areas with the least agricultural or livestock aptitude.
Neutral or slightly alkaline soils (pH 6.2–7.6), with strong accumulation of carbon on the surface. Hiperhumic Phaeozems, Rendzic Clayic Phaeozems.	225.9 ± 50.1	1.29		Soils with high natural productivity but subject to degradation process. The greatest restoration potential is in the border area, between the parcels degraded and the conserved forests.
Neutral soils (pH 6.1–7.2) with moderate accumulation of carbon on the surface. Humic clayic Phaeozems, abruptic clayic Luvisols.	105.3 ± 30.1	1.81		Soils with moderate original productivity. The potential for carbon restoration is only in the most degraded soils of this type.

The carbon stocks were estimated and represented from the quantitative information on organic carbon, stoniness, structure, texture and bulk density of the genetic horizons of 55 soil profiles, distributed on the surface of Javor Mountain, considering a depth of study of 30 cm (Table 22).

The potential for soil organic carbon stock change was estimated from the carbon information available in soil profiles with the same genetic origin and located in landscapes similar in their relief, drainage patterns, meso-climate, physiognomy and structure of the vegetation cover, but with different anthropogenic intervention and level of degradation (deforestation and/or erosion). To maximize the homogeneity of the study units, direct photointerpretation of the total area of Javor Mountain was developed out on a 1: 50 000 editing scale (Figure 2).

In this way, it was possible to spatially represent the changes in taxonomy, underground and superficial carbon stocks, as well as the annual erosion rate according with criteria established by the USDA for the Revised Universal Soil Loss Equation (RUSLE). Finally, to calibrate the carbon restoration extreme values, data taken from Soil erosion map of the Republic of Srpska (Tošić, Dragićević and Lovrić, 2012) on maximum carbon recovery rates in various reforestation periods were considered.

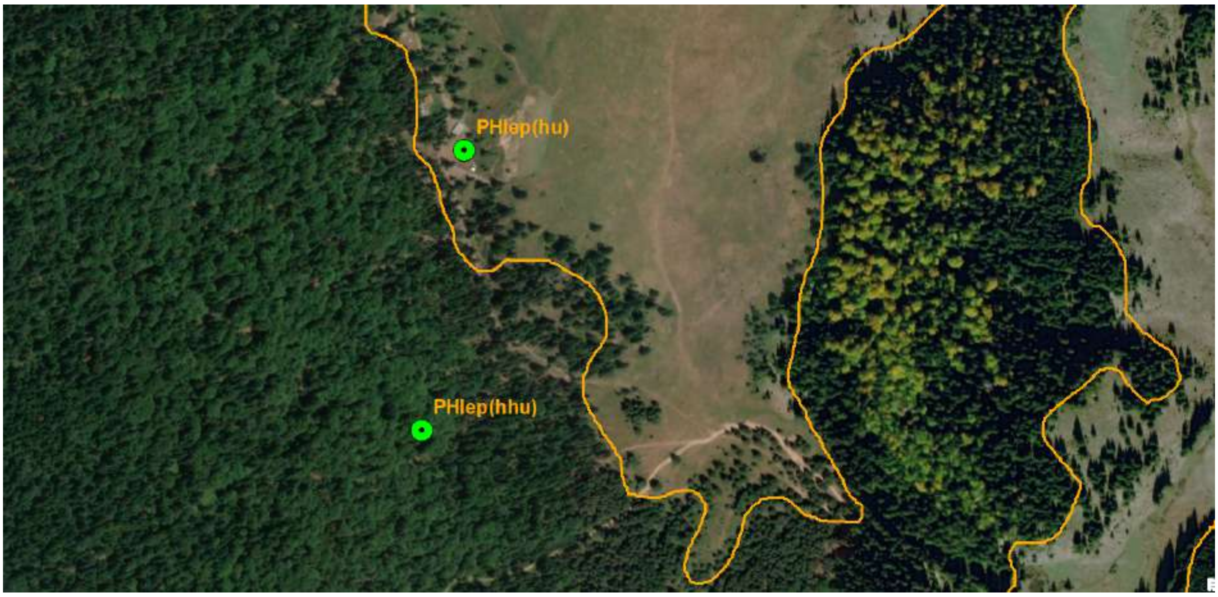


Figure 1. Variation of the carbon stock between the conserved matrix (283 tC/ha/yr) and the area of change (181 tC/ha/yr) for soils of the Epileptic Phaeozem type (Hiperhumic and Humic, respectively) in the same physiographic landscape

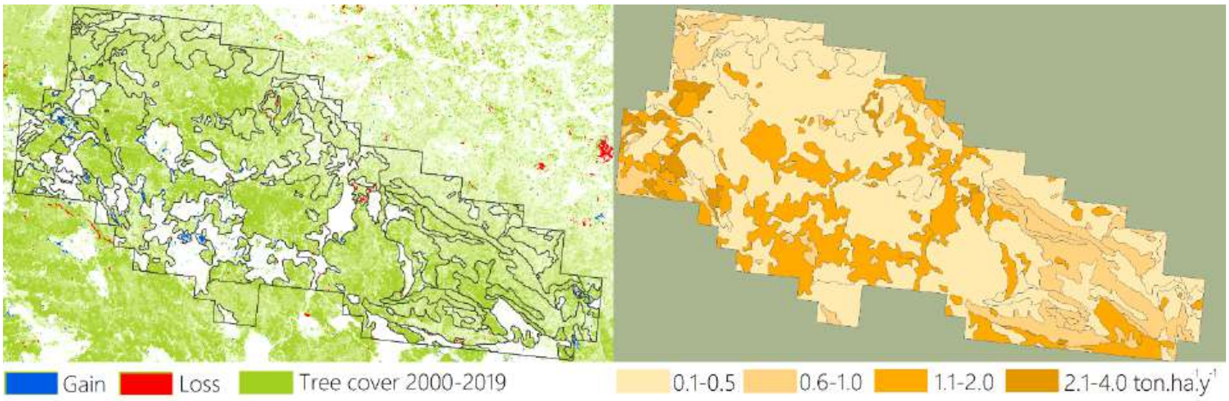


Figure 2. Degradation and reforestation 2000-2019 (left) and water erosion Rate (right) in the Javor Mountain soil units (right)

6. Other benefits of the practice

6.1. Benefits for soil properties

Reforestation in mountain conditions will, contribute to soil protection against erosion at the first place. Later, organic matter will accumulate on the forest floor jointly with development of roots, which will result in increase of soil organic carbon. The protection from soil erosion is another benefit where organic residue behaves as physical barrier to raindrops, improve water infiltration and protect topsoil (Kapovic *et al.*, 2013). Reforestation can also successfully restore the nitrogen reserves of the soil (Silver *et al.*, 2005).

6.2 Minimization of threats to soil functions

Table 23. Soil threats

Soil threats	
Soil erosion	Greater tree cover reduces the runoff speed at the surface and stabilizes the soil due to root propagation (Bochet and Garcia-Fayos, 2004; Tosic *et al.*, 2013; Kapović-Solomun, Ljuša and Eremija, 2014).
Nutrient imbalance and cycles	The leaching level of potassium is very low due to the high content of clays present in the soils of this region (between 45–72 percent of the total) (Kapović *et al.*, 2013)
Soil biodiversity loss	Reforestation contributes to preservation and improvement of the total number of fauna and flora species (Kapovic, 2013).
Soil sealing	The organic materials deposited and humidified on the soil surface (leaves, branches, bark) avoid the sealing and improve water infiltration (FAO, 2019).
Soil compaction	Increasing organic matter on the soil surface improves the porosity and bulk density of the aggregates (Kapović and Knežević, 2010).
Soil water management	Greater forest cover intercepts more rainwater and stimulates better infiltration within the fertility islands.

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

Conversion of degraded forests or bare land into the vegetation cover will contribute to an increase of forest cover and quality, and consequently forest production.

6.4 Mitigation of and adaptation to climate change

Previously, in 2006, according to the IPCC Guidelines, the amount of kg of carbon dioxide and even oxygen per year could be calculated from the difference in carbon stocks (at the same study site). Forest has a beneficial effect on climate change mitigation due to C storage in wood biomass in addition to soil.

6.5 Socio-economic benefits

Reforested land will have multiple environmental functions such as growth of forests of high quality that can be used for recreation. Indeed, these reforested forests are open public forests used by local people for hiking and other types of recreation.

6.6 Other benefits of the practice

In general, reforestation increases the average height, diversity and vigor of forests, from a degraded beech forest into a high beech, fir and spruce forest of good quality, that protects soils from further degradation. Average costs of afforestation range 1000-1500 EUR/ha, depending on terrain characteristics.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 24. Soil threats

Soil threats	
Nutrient imbalance and cycles	The pH of the soils in this region is in general very acid (except soils on limestones and carbonate parent material on the south of the Javor Mountain). Higher acidity may delay the phosphorus cycle and its absorption rate (Carreira *et al.*, 1997). Acidification of soils depends also on type of the forest, where domination of coniferous species can lead to further deterioration of soil characteristics. However, reforestation with coniferous species can underpin acidification process.
Soil acidification	pH of the between 4.3–5.4 in soils with reforestation practices, at altitudes between 1000–1300 meters above sea level, is a normal acidity level, so any reforestation program does not have a negative impact.

7.2 Other conflicts

Average costs of afforestation range 1000-1500 EUR/ha, depending on terrain characteristics, so for developing countries such as Bosnia and Herzegovina, there could be financial obstacles for the implementation of this measure (PFE, 2018).

8. Recommendations before implementing the practice

Soil type identification and careful selection of tree species adapted to the particular site condition, should be done before Afforestation/Reforestation, to ensure better success. Also, vitality of seedling is very important and care measures for young trees.

9. Potential barriers for adoption

Table 25. Potential barriers to adoption

Barrier	YES/NO	
Economic	Yes	Public forest enterprises are usually more focused on commercial Afforestation/Reforestation than environmental and bareland in higher altitudes are neglected from sustainable forest management (Kapovic and Knezevic, 2010).
Institutional	Yes	The institutions have not implemented measurement systems for the main soil properties and exhaustive monitoring of hydrometeorological information in the country (Kapovic Solomun, 2020).
Knowledge	Yes	There is insufficient information to diagnose the potential for carbon restoration on land distributed at altitudes below 1000 meters (Kapovic and Eremija, 2017).
Other	No	During the civil War, significant amounts of soil data were destroyed, including Soil Erosion Map of the Socialistic Republic of BIH, developed between 1979 and 1985 (Lazarević, 1986). This information was reconstructed only for the RS territory in 2012.

Photo

Photo 15. Typical landscape of Javor Mountain, Republic of Srpska, Bosnia and Herzegovina

References

Carreira, J.A., García-Ruiz, R., Liétor, J. & Harrison, A.F. 2000. Changes in soil phosphatase activity and P transformation rates induced by application of N- and S-containing acid-mist to a forest canopy. *Soil Biology and Biochemistry*, 32(13): 1857–1865. https://doi.org/10.1016/S0038-0717(00)00159-0

FAO. 2019. *Trees, forests and land use in drylands: the first global assessment*. Full report. FAO Forestry Paper No. 184. Rome. Page 2. (also available at: http://www.fao.org/3/ca7148en/ca7148en.pdf)

IUSS Working Group WRB. 2015. *World Reference Base for Soil Resources 2014, update 2015 International soil classification system for naming soils and creating legends for soil maps*. World Soil Resources Reports No. 106. FAO, Rome.

Kapović, M. & Knežević, M. 2010. *Characteristics of black soil on Javor mountain limestones in the Republic of Srpska.* First Serbian Forestry Congress, Faculty of Forestry, November 11th – 13th 2010, pp: 257-263. Belgrade, Serbia.

Kapovic Solomun, M. 2013. *Soils of Javor Mountain in the Republic of Srpska*. University of Belgrade, Faculty of Forestry, Doctoral Dissertation.

Kapović-Solomun, M., Ljuša, M. & Eremija, S. 2014. Variability of dystric cambisol in Olovsko forest-economic area. *Journal Forestry*, 85-95, Belgrade.

Kapović, M., Tošić, R., Knežević, M. & Lovrić, N. 2013. Assessment of soil properties under degraded forests: Javor mountain in Republic of Srpska - a case study. *Archives of Biological Sciences*, 65(2): 631–638. https://doi.org/10.2298/ABS1302631K

Kapović Solomun, M. & Eremija, S. 2017. *Soils of Javor Mountain*. National Monograph, Faculty of Forestry, University of Banja Luka, pp. 0-267.

Kapović Solomun, M. 2020. *Drought Management Plan of the Republic of Srpska*. Ministry of Agriculture, Forestry and Water Management and UNCCD, Banja Luka.

Lazarevic, R. 1986. SR Bosnia and Herzegovina's Erosion Map. Erosion Professional Newsletter, 14: 87–97. (also available at: http://www.udruzenjebujicara.com/)

PFE. 2018. Forest managament base for "Hanpjesacko" forest management unit, Public Forest Enterprize Sume Republike Srpske a.d. Sokolac, Sokolac [online]. [Cited 20 december 2020] https://sumerepublikesrpske.org/index.php/strana-2/func-startdown/1851/

Silver, W.L., Thompson, A.W., Reich, A., Ewel, J.J. & Firestone, M.K. 2005. Nitrogen Cycling in Tropical Plantation Forests: Potential Controls on Nitrogen Retention. Ecological Applications, 15(5): 1604–1614. https://doi.org/10.1890/04-1322

Tošić, R., Kapović-Solomun, M., Lovrić, N. & Dragićević, S. 2013. *Assessment of Soil erosion potential using RUSLE and GIS: a Case study of Bosnia and Herzegovina*. Fresenius Environmental Bulletin 22(12).

Tošić, R., Dragićević, S., & N. Lovrić. 2012. Assessment of soil erosion and sediment yield changes using erosion potential method - Case study: Republic of Srpska – BIH. *Carpathian Journal of Earth and Environmental Sciences,* 7(4): 147–154. (also available at: http://www.ubm.ro/sites/CJEES/viewIssue.php?issueId=19)

6. Natural afforestation of abandoned mountain grasslands along the Italian peninsula

Guido Pellis[1], Giuseppe Scarascia Mugnozza[2], Tommaso Chiti[1,2]

[1]Foundation Euro-Mediterranean Center on Climate Change (CMCC), Division on Impacts on Agriculture, Forests and Ecosystem Services (IAFES), Viterbo, Italy

[2]Department for Innovation in Biological, Agro-food and Forest systems (DIBAF), University of Tuscia, Viterbo, Italy

1. Related practices and hot-spot

Afforestation, Natural regeneration; Forests, Mountain soils

2. Description of the case study

Natural forest regeneration is a biological process which can be assisted through human interventions (FAO, 2019; Shono *et al.*, 2020). It can be a cost-effective approach to afforestation, reforestation and revegetation processes (as defined in Decision 16/CMP.1 (UNFCCC, 2006)). It differs from more intensive, tree-planting based approaches to restore forest ecosystems due to the absence of site preparation (Guidi *et al.*, 2014; Pellis *et al.*, 2019) and the time that woody plant species need to reach a new equilibrium (or steady state). This time interval can vary according to bioclimatic parameters of the sites where this process occurs. The regeneration of natural forests is a carbon sink process that leads to changes in the carbon stocks of all ecosystem carbon pools: biomass (above- and below-ground), dead organic matter (dead wood and litter), and soil. It is of particular interest as this process, which occurs on marginal lands formerly used as grassland, has been taking place worldwide for decades and represents a widespread land-use change (LUC) phenomenon (Archer, 2010). It was described and studied in several part of the world, for example, in North America (see for example Archer, 2010; Pinno and Wilson, 2011), in Latin America (see for example Chazdon *et al.*, 2016), in Africa (Chiti *et al.*, 2018) and in Asia (see for example Liu *et al.*, 2019). After the Second World War, the progressive abandonment of rural and mountain territory – by people who moved to urban areas – has caused a significant development of natural forest regeneration over abandoned grasslands in wide European areas (Höchtl, Lehringer and Konald, 2005; Zimmermann *et al.*, 2010) and in particular in the Mediterranean countries where, according to Fuchs *et al.* (2013), it accounts for more than 7.5 percent of the total surface. This is likely to be the cause of the process in other parts of the world, especially in developed countries. According to UN (2019) estimates, about 70 percent of the population will live in urbanized areas in 2050 (currently, this value corresponds to 55 percent), and, simultaneously, the number of people living in rural areas is expected to decrease. Therefore, it is legitimate to think that the process of marginal land abandonment will continue. On the opposite, the maintenance of rural areas and landscapes is one of the objectives of the European Union common agricultural policy (CAP). Its application, by means of Rural development programmes, can lead to the restoration of abandoned pastures and croplands (Fino *et al.,* 2020), limiting the spread of this phenomenon in Europe.

Here we present a national case study, which considers the Italian territory. Natural forest regeneration, which is extensively occurring along the entire peninsula, was investigated from East to West along the Alpine Mountain chain, and from North to South along the Apennine one. Several sites, with at least two stages from grassland currently managed (i.e. pastured or mowed) to abandoned grasslands affected by natural forest regeneration, were identified in different parts of the country. The sites were chosen to be representative of different potential vegetation types (e.g. conifer vs. broadleaves) and soil substrates. In each stage, the soil organic carbon (SOC) stock was determined in order to estimate and report CO_2 emission or removal due to this land-use change. On the basis of these results, it is possible to determine the mitigation potential offered by the natural forest regeneration along the Italian peninsula.

3. Context of the case study

Italy is located in the Southern part of European continent and it is nearly completely included in the Mediterranean basin, an area where, according to Giorgi and Lionello (2008), climate change effects are going to be particularly pronounced (e.g. rising temperatures and precipitation reduction).

The main characteristics of the Italian territory are strong landscape heterogeneity, with high mountain chains a few tens of kilometers from the sea (especially the Apennines, where the highest peaks nearly reach 3 000 m a.s.l.), and a long history of human settlements that have significantly modified the landscape (e.g. agricultural and pastoral land uses).

According to the IPCC classification maps, the Italian territory is mainly included in temperate climate zones (Figure 3) with only some areas of the Alps included in the Polar Moist zone. High activity clay soils are the most represented in Italy, while small patches of low activity clay, sandy and volcanic soils also occur in the country (Figure 3).

There are two major mountain ranges (Alps and Apennines) that cross the Italian peninsula: the first connects the Eastern to Western part of the Northern territory delimiting Italy from the central European countries and the latter follows the main peninsular inland territory from North to South. These mountain areas, especially the hardest to reach and least productive ones, are the most affected by land abandonment due to people movement to urban centers. However, grassland abandonment followed by natural forest regeneration has been observed also on hilly lands only a few kilometers from the Mediterranean coast. According to the last Italian National Inventory Report (Vitullo and Pellis, 2020), from 1990 to 2018, nearly 2 000 ha of grassland has been subjected to land conversion into forest, mainly due to natural forest regeneration.

In this context, six research projects which focus on natural forest regeneration over previous grasslands (managed or unmanaged) were selected because they deal with the effect of this process on SOC stocks. Overall, 15 study sites were identified from these six studies. Figure 3 shows the sites location along the Italian peninsula considering the climate and pedological subdivision. More details about the sites are summarized in Table 26.

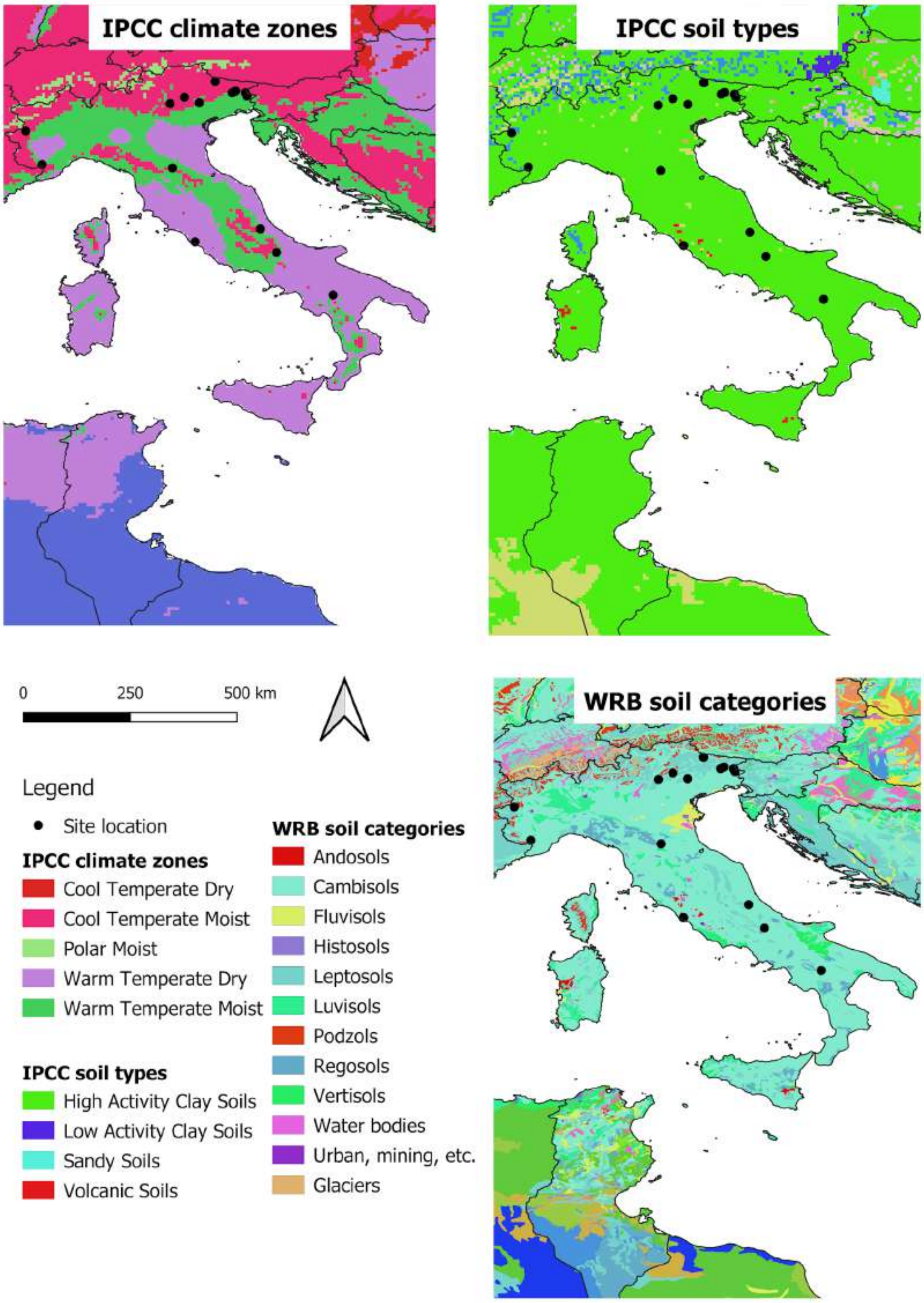


Figure 3. Location of the considered sites over the Italian territory. Panel (A) shows the IPCC climate zones distribution; panel (B) shows the IPCC soil types ones, while panel (C) show the FAO WRB soil classification (FAO, 2020)

4. Possibility of scaling up

The Table 26 database is not easily used to scale up the results. The main problem is that natural forest regeneration is a spontaneous process due to an absence of management practices rather than their presence. This means that the process is not directly manageable (predictable, adjustable) in space and time. However, some parameters may be indicative of a greater or lesser propensity (or probability) of a grassland area to be colonized by tree species, such as the distance of these from inhabited centers and their accessibility to human management; while other parameters can influence the speed of the natural forest regeneration, such as the climatic and pedological characteristics of the sites.

In addition, the considered data set has another limitation, which consists in the small number of observations related to this specific LUC, which does not cover all the possible soil/climate combinations present in Italy.

At present, the methodology used in the National GHG emissions inventory for the LULUCF sector is based on a mixed approach (Vitullo and Pellis, 2020), where the surfaces subject to transition on a statistical basis within the regional territories are known, but with the specific distribution of these conversions still unknown within the regional territory (Approach 2 described in Cai *et al.*, 2014).

5. Impact on soil organic carbon stocks

On the basis of the dataset limitation previously described, it is not possible to clearly explain the effect of the natural forest regeneration over abandoned grasslands on SOC stock changes for the whole Italian peninsular territory. However, Table 26 summarizes the results obtained for the selected sites in the upper 30 cm of the mineral profile (usually called topsoil), which indicates a mean annual SOC stock loss of 0.167 tC/ha with an extremely large uncertainty (standard deviation of 1.217 t/ha) when forests naturally regenerate on abandoned grasslands. This result is obviously affected by the lack of widespread and proportional coverage of the study sites according to the ecopedological conditions of the national territory. The large uncertainty around the mean value confirms how natural forest regeneration on grassland does not equally or concordantly affect the stock of SOC in the topsoil at each site (Table 26). Indeed, within a limited and homogeneous area, SOC stock changes due to natural forest regeneration on former grasslands can be positive, negative or statistically non-significant as demonstrated by several studies (e.g. Guo and Gifford, 2002; La Mantia *et al.*, 2013; Pellis *et al.*, 2019).

According to recent literature, the difference in the SOC stock changes among the considered sites is manly explained by their climatic conditions, but also by their forest vegetation types (i.e. conifers or broadleaves-dominated). Guo and Gifford (2002) and Alberti *et al.* (2011) showed that the dominant climatic driver for SOC stock changes along natural forest regeneration is the mean annual precipitation, while other works (see Poeplau *et al.*, 2011; La Mantia *et al.*, 2013; Pellis *et al.*, 2019) demonstrated that the SOC stock change correlation is significant and more robust with temperature parameters (mean annual temperature and average minimum winter temperature). Moreover, Pellis *et al.* (2019), suggested that the SOC stock change is, or may be, influenced in a not negligible way by the type of forest vegetation, a parameter that, in turn, depends on sites' climate.

According to Jobbágy and Jackson (2000) the type of vegetation (grasslands, shrublands and forests) has an influence on the SOC stock vertical distribution along the profile, even below 30 cm (topsoil threshold). This result can be explained by several input and output processes affecting SOC which can occur not only on surface layers (e.g. aboveground litter deposition and microbial mineralization), but also in deeper layers where root system colonization is responsible, for example, of the deposition of the belowground litter and the release of root exudates (Sollins, Homann and Caldwell, 1996). Several authors (e.g. Poeplau *et al.*, 2011; Pellis *et al.*, 2019) pointed out that SOC stock change from grassland to forest can lead to results that significantly differs if subsoil (below 30 cm depth) is considered or not. Although the IPCC guidelines (2006) and the subsequent refinement (IPCC, 2019) suggest to estimate SOC stocks (and their changes due to LUC) at least in the topsoil (0-30 cm), the results obtained for the areas considered in this case study (see Pellis *et al.,* 2019) suggest that the SOC accumulation due to natural forest regeneration can be much larger when considering also the subsoil compartment, as reported also by other studies (Don *et al.*, 2007, Poeplau and Don, 2013)

Table 26. Potential of additional carbon storage in the different areas along the Italian Peninsula

The values in brackets in the columns "Baseline C stock" and "Additional C storage" are standard deviation estimates. For all studies, soils are considered as High activity clay acceding to the IPCC (2006)

Location	Climate zone	Baseline C stock	Additional C storage	Duration	Depth	Mean coordinate (WGS 84)	Mean elevation	MAP	MAT	Soil type	More info[£]	Reference
	IPCC (2006)	tC/ha	tC/ha/yr	years	cm	Long. / Lat. Decimal degree	m a.s.l.	mm/yr	°C	IUSS WRB (2015)		
Castello Tesino	Cool tempe-rate moist	148.44 (21.17)*	-0.08 (0.39)*	72	30	11.650 / 46.125	1700	1286	4.6	Phaeozem	*Picea abies* (L.) H.Karst., *Larix decidua* (Mill.)	Pellis *et al.* (2019)
Mel		75.87 (7.58)*	+0.30 (0.18)*	72	30	12.071 / 45.969	1250	1670	6.5	Cambisol	*P. abies* (L.) H.Karst.	
Chianocco		74.75 (5.03)*	-0.13 (0.07)*	72	30	7.202 / 45.177	1110	967	8.4		*Fagus sylvatica* L., *Pinus sylvestris* L.	
Firenzuola		91.96 (6.38)*	+0.53 (0.38)*	78	30	11.320 / 44.140	890	1620	10.1		*Quercus cerris* L., *Quecus pubescens Willd.*	
Farindola	Warm tempe-rate moist	116.08 (3.72)*	+0.63 (0.15)*	80	30	13.783 / 42.433	1090	1136	9.8	Phaeozem	*F. sylvatica* L.	
Pignola	Cool tempe-rate moist	84.97 (15.58)*	+0.93 (0.50)*	70	30	15.819 / 40.583	1040	957	11		*Q. cerris* L.,	
Vastogirardi	Warm tempe-rate Moist	169.10 (6.79)*	-0.51 (0.26)*	70	30	14.228 / 41.819	1090	1039	10.0	Chernozem	*Q. cerris* L., F. *sylvatica* L.	Pellis (2017)

Location	Climate zone	Baseline C stock	Additional C storage	Duration	Depth	Mean coordinate (WGS 84)	Mean elevation	MAP	MAT	Soil type	More info[£]	Reference
	IPCC (2006)	tC/ha	tC/ha/yr	years	cm	Long. / Lat. Decimal degree	m a.s.l.	mm/yr	°C	IUSS WRB (2015)		
Tolfa	Warm Tempe-rate Dry	74.42 (22.24) **	+0.04 (0.35)**	70	30	11.952 / 42.067	120	1049	13.7	Cambisol	*Q. cerris* L., *Fraxinus ornus* L., *Acer monspessulanum* L.	Zanini (2017)
Danta di Cadore	Cool tempe-rate moist	91.10 (4.70)*	-0.05 (0.07)*‡	NA	30	12.512 / 46.568	1483	1043	3.5	Cambisol	*P. abies* (L.) H.Karst., *L. deciduas* (Mill.)	Fino *et al.* (2020)
Lavarone		86.08 (4.64)*	-0.62 (0.12)*‡	N.A.	30	11.251 / 45.946	1150	1278	7.2		*P. abies* (L.) *H.Karst., F. sylvatica* L.	Guidi *et al.* (2014)
Chiusa Pesio		115.7 (0.0)*	-0.53 (0.21)*	23	30	7.664 / 44.238	900	1430	12.0	Cambisol*	NA	Alberti *et al.* (2011)
Forgaria del Friuli		99.4 (19.3)*	-1.14 (0.70)*	28	30	12.993 / 46.251	800	2055	13.0			
Trasaghis	Warm tempe-rate moist	80.8 (7.1)*	-0.58 (0.16)*	48	30	13.084 / 46.299	1000	2130	13.0			
Faedis		84.5 (10.0)*	-0.78 (0.23)*	47	30	13.401 / 46.160	800	2265	13.0			
Taipana	Cool tempe-rate moist	106.7 (16.0)*	-0.51 (0.32)*	67	30	13.358 / 46.249	600	2415	8.0	Leptosol*		

£ This information refers to dominant tree species present in the afforested stages. Plant nomenclature follows The Plant List (2013); * Number of soil samples considered (n=3), ** Number of soil samples considered (n=4); ‡ The authors did not specify the minimum age of the forest stand. We assumed that a mature forest in the climate condition of these sites is at least 70 years old.

6. Other benefits of the practice

6.1. Benefits for soil properties

Apart from the contrasting effects on the increase of SOC, natural forest regeneration can have a potential impact also on other soil properties, depending on the location where the process is taking place (e.g. topography) and the climatic conditions of the area. Below the main changes occurring at the different soil properties in relation to the process of natural forest regeneration observed in the investigated areas are reported. Changes following natural forest regeneration can be positive, negative or neutral. However, it has to be noted that positive and neutral effects of natural regeneration are undoubtedly more abundant than negative effects as observed also by Eldridge *et al.* (2011).

Physical properties

According to Panagos *et al.* (2015), forest expansion (especially afforestation and re-vegetation), at European level, is a good practice to reduce soil water erosion especially in territories characterized by high soil loss rates. Indeed, according to the same authors, forest lands are characterized by the lowest soil loss rate when compared to other land uses, including pastures and shrub-herbaceous vegetation. However, these estimates are significantly influenced also by other parameters as the percentage of ground coverage and slopes steepness. These parameters may vary from site to site and, in particular situations, it is possible that they determine a higher surface runoff in wooded lands than in grassland as demonstrated by Archer (2017). In our study sites, no measurement was directly performed to investigate the impact of natural forest regeneration on water infiltration and aggregate stability. In term of soil compaction and the related changes in soil bulk density, it no significant change due to the natural forest regeneration processwas observed in all investigated sites (Pellis *et al.*, 2019).

Chemical properties

Natural forest regeneration can result in lower soil pH, higher soil carbon and nitrogen pools and higher potential nitrogen mineralization (Eldridge *et al.*, 2011). This is mainly due to the changes in plant species composition which substantially affects litter quality and its chemical composition (Guidi *et al.*, 2014). Accordingly, results from this case study indicate that the variation of pH along the natural forest regeneration process was marked and typically in the direction of higher acidity in the older forest stages. This variation can be possibly related to the switch from herbaceous to perennial vegetation since broadleaves and, particularly, evergreen trees can produce a litter, which strongly contribute to soil acidification as also observed by Hiltbrunner *et al.*, (2013). Concerning nitrogen concentration, results from this case study indicate a general increase in the nitrogen pool, particularly below 30 cm depth, while, for the cation exchange capacity, it was observed either an increases or no any significant change along the natural forest regeneration process (Pellis, 2017).

Biological properties

Natural forest regeneration fundamentally alters the ecological processes and tends to reduce the plant richness and biodiversity which characterize managed and semi-natural grassland ecosystems, particularly in settings where the number of encroaching woody species is low. In temperate climates, colonization of grasslands by woody species often threaten the maintenance of endemic grassland biodiversity (Plantureux, Peeters and McCracken, 2005; Habel *et al.*, 2013; Archer *et al.*, 2017). However, regenerating forests also support communities of native flora and fauna that depend on forest habitat, and at the landscape level, grasslands surrounded by other land uses including woodlands and forests have the highest number of species (Plantureux, Peeters and McCracken, 2005).

The natural forest regeneration process substantially alters the composition of soil organisms, favoring fungal communities to the detriment of bacterial ones, and the activity of the soil microbial community (Hollister *et al.*, 2010). For most of the areas included in this study, Pellis (2017) estimated the microbial biomass and main enzymatic activities related to C and N showing a progressive decrease, although not always significant, of both microbial biomass and enzymatic activity, especially in the upper soil layer (0-5 cm depth). According to Pellis (2017), these changes may be mainly caused either by the variation of plant species composition and by the variation of site microclimatic characteristics related to forest canopy closure. Plant species composition change is, probably, the dominant factor affecting soil biochemical properties as vegetation influences both the quantity and quality of the organic matter that reaches the soil (Trasar-Cepeda, Leiros and Gil-Sotres, 2008). Indeed, woody plant species, and specifically conifers, release a less palatable organic substance for microbial communities (Chabrerie *et al.*, 2003), which are characterized by chemically complex and recalcitrant molecules, such as lignin (Lucas-Borjas *et al.*, 2010).

6.2 Minimization of threats to soil functions

The effect of natural forest regeneration on minimizing soil threats cannot be defined uniquely, since apart from a possible increase in SOC, most of the main soil properties (e.g. texture, bulk density) observed in the investigated areas remain unchanged (Pellis *et al.*, 2019). Some important negative changes (e.g. soil erosion) reported in the scientific literature were not investigated within the framework of this study. Table 27 reports the possible changes occurring at the soil as a result of natural forest regeneration on the investigated areas, focusing on the possible soil threats.

Table 27. Soil threats

Soil threats	
Soil erosion	General decrease which can vary from site to site.
Nutrient imbalance and cycles	Increases or no changes occurring at the N pool. Increases or no changes in cation exchange capacity (Pellis, 2017).
Soil acidification	Decreases in soil pH due to a change in organic inputs from herbaceous, richer in cellulose, to inputs from perennial vegetation richer in lignin (e.g. conifers) (Pellis, 2017).

Soil threats	
Soil biodiversity loss	Reduction of microbial biomass (Pellis, 2017).
Soil compaction	No change in soil compaction or a possible decrease of bulk density from grassland to forest (Pellis *et al.*, 2019).

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

Since natural forest regeneration is not a voluntary practice but is mainly due to land abandonment the resulting woody vegetation is generally not used for any productive purpose (e.g. Food/Fuel/Feed/Timber). However, in cases where management of these newly formed forests is introduced, there is a positive impact since the wood could be used to produce fuel wood, timber and charcoal. In the investigated areas the above ground biomass represents a large pool ranging from 95 tC/ha to 167 tC/ha (Pellis *et al.,* 2019), and the use of this biomass for different purposes could have positive impact on production and incomes for the owner of the land.

6.4 Mitigation of and adaptation to climate change

Natural forest regeneration can have diverse impacts on the soil organic carbon pool, leading to both emissions and removals (Jackson *et al.*, 2002). In general, the conversion from grassland to forestland is considered as a climate change mitigation measure and afforestation is in fact included in many nationally determined contributions under the Paris Agreement (e.g. China, India, Honduras, Senegal, etc.). On the other hand, its potential highly depends on the local features of the grassland where the process is taking place (e.g. topography), and local environmental conditions, especially the precipitation rate. In this case study, the process of natural forest regeneration is always leading to an increase of C at an ecosystem level, due to the presence of perennial aboveground biomass. However, the overall mitigation potential related to the process of natural forest regeneration greatly depends on the contribution of the soil during this process. In the investigated areas, the overall mitigation potential at an ecosystem level can vary from 160 tC/ha (e.g. Mel and Danta di Cadore areas, **Table 26**) to about 300 tC/ha (e.g. Firenzuola area **Table 26**), mainly due to positive variations occurring at the SOC pool.

6.5 Socio-economic benefits

The process of natural forest regeneration observed in the investigated areas, lead to the formation of new forests, which in case start to be managed can provide socio-economic benefits due to the wood that can be possibly used for different purposes. The main socio-economic benefits possibly provided by these newly formed forests could be related to the concept of Ecosystem services. In particular, these forests can provide provisioning services (e.g. row material used for energy), supporting services (e.g. soil formation) and regulating services (e.g. climate regulation). However, among these services, only the provisioning ones can represent an income for the owner of the area, while the other two types of services provide benefits for the society. Since the investigated areas are currently abandoned, no economic benefit is obtained while only benefits for the society are actually observed (e.g. climate regulation).

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Since no management is carried out in the areas affected by natural forest regeneration, it is not possible to identify specific tradeoffs, or conflict with other practices.

7.2 Increases in greenhouse gas emissions

During the process of natural forest regeneration, no management activity is carried out so that there are no greenhouse gases (GHG) emissions related to this practice in the investigated areas. Due to the abandonment of the grazing land activities a decrease in methane emission is expected to occur during the process. Methane emissions in grazing land are mainly related to the emissions due to enteric fermentation and manure, which do not occur during the process due to the abandonment of the grazing land management. As a consequence, during natural forest regeneration there are no emissions of GHG different from CO_2 (e.g. possible emissions from soil).

7.3 Conflict with other practice(s)

No conflict with other practices can be related to this process since it is not a voluntary practice but occur naturally after abandonment of grazing land management in mountain areas.

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

As specified in section 6.3 there is not any negative impact on production for the investigated areas. This involuntary practice leads to the formation of new forests, which are not managed, but in case management is introduced the practice can lead only to positive impacts (e.g. fuel wood production).

8. Potential barriers for adoption

No type of barrier is expected to prevent the occurrence of natural forest regeneration, as this corresponds to an involuntary process that occurs independently after land abandonment (Archer *et al.*, 2017). However, an important aspect that needs to be considered is that this process is occurring in areas classified as agricultural land, which have a different legislation compared to forests land. In many Italian regions it is allowed the restoration of grasslands interested by natural forest regeneration. In terms of climate change mitigation, this would mean the immediate release of all the CO2 stored in the aboveground biomass during the process of natural forest regeneration leading to an increase in atmospheric CO2 concentration.

Photo

Photo 16. Forest expansion over abandoned mountain grasslands at the Castello Tesino area located on the Italian Alps

References

Alberti, G., Leronni, V., Piazzi, M., Petralla, F., Mairota, P., Peressotti, A., Piussi, P., Valentini, R., Gristina, L., La Mantia, T., Novara, A. & Rùhl, J. 2011. Impact of woody encroachment on soil organic carbon and nitrogen in abandoned agricultural lands along a rainfall gradient in Italy. *Regional Environmental Change*, 11: 917–924. https://doi.org/10.1007/s10113-011-0229-6

Archer, S.R., Andersen, E.M., Predick, K.I., Schwinning, S., Steidl, R.J. & Woods, S.R. 2017. Woody Plant Encroachment: Causes and Consequences. *In* D.D. Briske (Ed.) *Rangeland Systems: Processes, Management and Challenges*, pp. 25–84. Springer Series on Environmental Management. Cham, Springer International Publishing. https://doi.org/10.1007/978-3-319-46709-2_2

Archer, S.R. 2010. Rangeland Conservation and Shrub Encroachment: New Perspectives on an Old Problem. *In* Toit, J.T.D., Kock, R., Deutsch, J.C. (Eds.) *Wild Rangelands*, pp. 53–97. John Wiley & Sons, Ltd. https://doi.org/10.1002/9781444317091.ch4

Cai, B., Zarate do Couto, H. T., Dong, H., Federici, S., Garivait, S., Hassan, R., Lasco, R., O'Brien, P., Roelandt, C., Sanz Sánchez, M. J., Wagner, F. & Zhu, J. 2014. Generic methodologies for area identification, stratification and reporting. *In* Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Jamsranjav, B., Fukuda, M., Troxler, T. (Eds.) *2013 Revised Supplementary Methods and Good Practice Guidance Arising from the Kyoto Protocol*, pp. 2.15-2.24. IPCC Task Force on National Greenhouse Gas Inventories Technical Support Unit. Published: IPCC, Switzerland.

Chabrerie O., Laval K., Puget P., Desaire S. & Alard D. 2003. Relationships between plant and soil microbial communities along a successional gradient in a chalk grassland in north-western France. *Applied Soil Ecology*, 24: 43-56. https://doi.org/10.1016/S0929-1393(03)00062-3

Chazdon, R.L., Broadbent, E.N., Rozendaal, D.M.A., Bongers, F., Almeyda Zambrano, A.M., Aide, T.M., Balvanera, P., Becknell, J.M., Boukili, V., Brancalion, P.H.S., Craven, D., Almeida-Cortez, J.S., Cabral, G. A.L., de Jong, B., Denslow, J.S., Dent, D.H., DeWalt, S.J., Dupuy, J.M., Durán, S.M., Espírito-Santo, M.M., Fandino, M.C., César, R.G., Hall, J.S., Hernández-Stefanoni, J.L., Jakovac, C.C., Junqueira, A.B., Kennard, D., Letcher, S.G., Lohbeck, M., Martínez-Ramos, M., Massoca, P., Meave, J.A., Mesquita, R., Mora, F., Muñoz, R., Muscarella, R., Nunes, Y.R.F., Ochoa-Gaona, S., Orihuela-Belmonte, E., Peña-Claros, M., Pérez-García, E.A., Piotto, D., Powers, J.S., Rodríguez-Velazquez, J., Romero-Pérez, I.E., Ruíz, J., Saldarriaga, J.G., Sanchez-Azofeifa, A., Schwartz, N.B., Steininger, M.K., Swenson, N.G., Uriarte, M., van Breugel, M., van der Wal, H., Veloso, M.D.M., Vester, H., Vieira, I.C.G., Vizcarra Bentos, T., Williamson, G.B. & Poorter, L. 2016. Carbon sequestration potential of second-growth forest regeneration in the Latin American tropics. *Science Advances,* 2(5): e1501639. https://doi.org/10.1126/sciadv.1501639

Chiti, T., Rey, A., Jeffery, K., Lauteri, M., Mihindou, V., Malhi, Y., Marzaioli, F., White, L.J.T. & Valentini, R. 2018. Contribution and stability of forest- derived soil organic carbon during woody encroachment in a tropical savanna. A case study in Gabon. *Biology and Fertility of Soils*, 54: 897-907. https://doi.org/10.1007/s00374-018-1313-6

Don, A., Schumacher, J., Scherer-Lorenzen, M., Scholten, T. & Shulze, E.-D. 2007. Spatial and vertical variation of soil carbon at two grassland sites – implications for measuring soil carbon stocks. *Geoderma*, 141: 272–282. https://doi.org/10.1016/j.geoderma.2007.06.003

Eldridge, D., Bowker, M.A., Maestre, F.T., Roger, E., Reynolds, J.F. & Whitford, W.G. 2011. Impacts of shrub encroachment on ecosystem structure and functioning: towards a global synthesis. *Ecology Letters,* 14: 709-722. https://doi.org/10.1111/j.1461-0248.2011.01630.x

FAO. 2020. *FAO SOILS PORTAL – Harmonized World Soil Database v 1.2* [online]. [Cited 15 July 2020]. www.fao.org/soils-portal/soil-survey/soil-maps-and-databases/harmonized-world-soil-database-v12/en/

FAO. 2019. *Restoring forest landscapes through assisted natural regeneration (ANR) – A practical manual.* Bangkok. 52 pp. License: CC BY-NC-SA 3.0 IGO. (also available at www.fao.org/3/ca4191en/CA4191EN.pdf)

Fuchs, R., Herold, M., Verburg, P.H. & Clevers, J.G.P.W. 2013. A high-resolution and harmonized model approach for reconstructing and analysing historic land changes in Europe. *Biogeosciences*, 10: 1543–1559. https://doi.org/10.5194/bg-10-1543-2013

Fino, E., Blasi, E., Perugini, L., Pellis, G., Valentini, R. & Chiti, T. 2020. Is soil contributing to climate change mitigation during woody encroachment? A case study on the Italian alps. *Forests,* 11(8): 887. https://doi.org/10.3390/f11080887

Giorgi, F. & Lionello, P. 2008. Climate change projections for the Mediterranean region. *Global and Planetary Change*, 63: 90–104. https://doi.org/10.1016/j.gloplacha.2007.09.005

Guidi, C., Vesterdal, L., Gianelle, D. & Rodighero, M. 2014. Changes in soil organic carbon and nitrogen following forest expansion on grassland in the Southern Alps. *Forest Ecology and Management*, 328: 103–116. https://doi.org/10.1016/j.foreco.2014.05.025

Guo, L.B. & Gifford, R.M. 2002. Soil carbon stocks and land use change: a meta analysis. *Global Change Biology*, 8: 345–360. https://doi.org/10.1046/j.1354-1013.2002.00486.x

Habel, J.C., Dengler, J., Janišová, M., Török, P., Wellstein, C. & Wiezik, M. 2013. European grassland ecosystems: threatened hotspots of biodiversity. *Biodiversity and Conservation*, 22: 2131–2138. https://doi.org/10.1007/s10531-013-0537-x

Hiltbrunner, D., Zimmermann, S. & Hagedorn, F. 2013. Afforestation with Norway spruce on a subalpine pasture alters carbon dynamics but only moderately affects soil carbon storage. *Biogeochemistry*, 115(1): 251–266. https://doi.org/10.1007/s10533-013-9832-6

Höchtl, F., Lehringer, S. & Konold, W. 2005. "Wilderness": what it means when it becomes a reality – a case study from the southwestern Alps. *Landscape and Urban Planning*, 70: 85–95. https://doi.org/10.1016/j.landurbplan.2003.10.006

Hollister, E.B., Schadt, C.W., Palumbo, A.V., Thomas, J.A. & Boutton, T.W. 2010. Structural and functional diversity of soil bacterial and fungal communities following woody plant encroachment in the southern Great Plains. *Soil Biology and Biochemistry,* 42: 1816-1824. https://doi.org/10.1016/j.soilbio.2010.06.022

Intregovernamental Panel on Climate Change (IPCC). 2006. *2006 IPCC Guidelines for National Greenhouse Gas Inventories*. National Greenhouse Gas Inventories Programme, Eggleston H.S., Buendia L., Miwa, K., Ngara, T. & Tanabe, K. (Eds). IGES, Japan.

Intregovernamental Panel on Climate Change (IPCC). 2019. *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories*. Calvo Buendia, E., Tanabe, K., Kranjc, A., Baasansuren, J., Fukuda, M., Ngarize, S., Osako, A., Pyrozhenko, Y., Shermanau, P. & Federici, S. (Eds). IPCC, Switzerland.

Jackson, R.B., Banner, J.L., Jobbágy, E.G., Pockman, W.T. & Wall, D.H. 2002. Ecosystem carbon loss with woody plant invasion of grasslands. *Nature,* 418: 623–626. https://doi.org/10.1038/nature00910

Jobbágy, E.G. & Jackson, R.B. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications*, 10(2): 423–436. https://doi.org/10.1890/1051-0761(2000)010[0423:TVDOSO]2.0.CO;2

Liu, Y.-H., Cheng, J.H., Schmid, B., Tang, L.-S. & Sheng, J.-D. 2019. Woody plant encroachment may decrease plant carbon storage in grassland under future drier conditions. *Journal of Plant Ecology*, 13: 213-223. https://doi.org/10.1093/jpe/rtaa003

La Mantia, T., Gristina, L., Rivaldo, E., Pasta, S., Novara, A. & Rühl, J. 2013. The effect of post-pasture woody plant colonization on soil and aboveground litter carbon and nitrogen along bioclimatic transect. *iForest*, 6: 238-246. https://doi.org/10.3832/ifor0811-006

Lucas-Borjas M.E., Bastida F., Nicolás C., Moreno J.L., del Cerro A. & Andrés M. 2010. Influence of forest cover and herbaceous vegetation on the microbiological and biochemical properties of soil under Mediterranean humid climate. *European Journal of Soil Biology*, 46: 273-279. https://doi.org/10.1016/j.ejsobi.2010.05.003

Panagos, P., Borrelli, P., Poesen, J., Ballabio, C., Lugato, E., Meusburger, K., Montanarella, L. & Alewell, C. 2015. The new assessment of soil loss by water erosion in Europe. *Environmental Science & Policy*, 54: 438–447. https://doi.org/10.1016/j.envsci.2015.08.012

Pellis, G. 2017. *Soil organic carbon, soil microbial communities and plant biomass along woody encroachments over abandoned pastures in Italy*. Dipartimento per la Innovazione nei Sistemi Biologici Agroalimentari e Forestali, Università degli studi della Tuscia di Viterbo (PhD Thesis).

Pellis, G., Chiti, T., Rey, A., Curiel Yuste, J., Trotta, C. & Papale, D. 2019. The ecosystem carbon sink implications of mountain forest expansion into abandoned grazing land: The role of subsoil and climatic factors. *Science of the Total Environment*, 672: 106-120. https://doi.org/10.1016/j.scitotenv.2019.03.329

Pinno, B.D. & Wilson, S.D. 2011. Ecosystem carbon changes with woody encroachment of grassland in the northern Great Plains. *Écoscience*, 18(2): 157-163. https://doi.org/10.2980/18-2-3412

Plantureux, S., Peeters, A. & McCracken, D. Biodiversity in intensive grasslands: Effect of management, improvement and challenges. *Agronomy Research*, 3(2): 153-164.

Poeplau, C. & Don, A. 2013. Sensitivity of soil organic carbon stocks and fractions to different land-use change across Europe. *Geoderma*, 192: 189–201. https://doi.org/10.1016/j.geoderma.2012.08.003

Poeplau, C., Don, A., Vesterdal, L., Leifeld, J., Van Wesemael, B., Schumacher, J. & Gensior, A. 2011. Temporal dynamics of soil organic carbon after land-use change in the temper- ate zone – carbon response functions as a model approach. *Global Change Biology*, 17: 2415–2427. https://doi.org/10.1111/j.1365-2486.2011.02408.x

Shono, K., Chazdon, R., Bodin, B., Wilson, S. & Durst, P. 2020. Assisted natural regeneration: harnessing nature for restoration. *Unasylva*, 252(71): 71-81.

Sollins P., Homann P. & Caldwell B.A. 1996. Stabilization and destabilization of soil organic matter: mechanisms and controls. *Geoderma*, 74: 65-105. https://doi.org/10.1016/S0016-7061(96)00036-5

The Plant List. 2013. *The Plant List Version 1.1* [online]. [Cited 15 July 2020]. http://www.theplantlist.org/

Trasar-Cepeda C., Leiros M.C. & Gil-Sotres F. 2008. Hydrolytic enzyme activities in agricultural and forest soils. Some implications for their use as indicators of soil quality. *Soil Biology and Biochemistry*, 40: 2146-2155. https://doi.org/10.1016/j.soilbio.2008.03.015

United Nation Framework Convention on Climate Change (UNFCCC). 2006. ANNEX Definitions, modalities, rules and guidelines relating to land use, land-use change and forestry activities under the Kyoto Protocol. Decision 16/CMP.1 – Land use, Land-use change and forestry.

United Nations (UN), Department of Economic and Social Affairs, Population Division. 2019. World Urbanization Prospects 2018: Highlights (ST/ESA/SER.A/421). (also available at: https://population.un.org/wup/Publications/Files/WUP2018-Highlights.pdf)

Vitullo, M. & Pellis, G. 2020. Land Use, Land Use Change and Forestry. *In* Romano, D., Arcarese, C., Bernetti., A., Caputo, A., Contaldi, M., Cordella, M., De Lauretis, R., Di Cristofaro, E., Gagna, A., Gonella, B., Morocci, F., Taurino, E., Vitullo, M. (Eds.) *Italian Greenhouse Gas Inventory 1990-2018 – National Inventory Report 2020*. pp. 250-297. Institute for Environmental Protection and Research (ISPRA), Italy.

Zanini, M. 2017. *Variazione degli stocks di carbonio in funzione della dinamica prateria-foresta matura nei monti della Tolfa (Roma).* Facoltà di Scienze Matematiche Fisiche e Naturali, Dipartimento dei Biologia ambientale, Sapienza Università di Roma (PhD Thesis).

Zimmermann, P., Tasser, E., Leitinger, G. & Tappeiner, U. 2010. Effects of land-use and land- cover pattern on landscape-scale biodiversity in European Alps. *Agriculture Ecosystem & Environment*, 139: 13–22. https://doi.org/10.1016/j.agee.2010.06.010

7. Afforestation of vineyards in Italy

Chiara Ferré[1], Roberto Comolli[1], Gianni Facciotto[2], Sara Bergante[2]

[1]Department of Earth and Environmental Sciences, Milano Bicocca University, Italy

[2]Council for Agricultural Research and Agricultural Economy Analysis CREA –Research Centre for Forestry and Wood, Casale Monferrato, Italy

1. Practice(s) used

Afforestation of agricultural land

2. Description of the case study

The effect of conversion from vineyard (VY) to tree plantation (TP) on soil organic carbon (SOC) stocks was investigated by sampling paired plots in a hilly area of Monferrato (Italy).

The study area includes a VY and a nearby 30-year-old TP for wood production that was established in the winter 1985-1986 on an area formerly cultivated as vineyard. Eight poplar clones (*Populus x canadensis* I-214, Luisa Avanzo, Neva, Ongina, Panaro, Zero*; Populus x canadensis x P. maximoviczii* Eridano*; Populus alba* Villafranca) were consociated with some timber species (wild cherry - *Prunus avium* L.-, European ash – *Fraxinus excelsior* L., manna ash – *Fraxinus ornus* L.-, himalayan cedar – *Cedrus deodara* (Roxb) G.Don.-). In contrast with the VY, where the soil was deeply ploughed (till about 70 cm) before planting the vine and then annually surface tilled, at TP soil tillage involved the first ten-planting years only. Before the tree establishment, the soil has been ploughed to a depth of 30 cm and the harrowing has been carried out twice a year for three years and then annually for the next seven years, after which no further tillage has been carried out. The study area (3 ha) extends along a slighty-wavy slope (average gradient of 15 percent).

The impact of land use change on SOC stock was evaluated testing for autocorrelation among the model residuals. Soil sampling was performed from 0-10 cm, 10-40 cm and 40-70 cm layers at 61 and 69 points in the VY and the TP respectively, to characterize spatial distribution of SOC stock and other soil properties. At TP the organic horizons were sampled and analyzed for OC content determination.

Statistical analyses showed significant (p-value < 0.05) differences between the investigated land uses up to 70 cm in depth.

The VY showed in the 0-70 cm layer a SOC stock of 59.2±23.6 t/ha, a weighted average pH value of 7.9±0.1, and a C:N ratio of 7.8±0.7. TP was characterized by higher SOC stock (74.4±21.1 t/ha) and C:N ratio (8.8±1.0) and lower pH value (7.5±0.5) than VY; the SOC stock of the organic layer was 10.2 t/ha.

Our study showed that: (1) 30 years of TP changed SOC stock, resulting in an increase of 26 percent in the first 70 cm, which becomes 43 percent if the organic layers were included; (2) soil acidification and change in SOM type were also observed in TR compared to VY; and (3) the spatial distribution of soil properties in the VY was affected by erosive and depositional dynamics unlike the TR where vegetation controls erosion.

3. Context of the case study

The process of abandonment of the agricultural sector that characterized Italy since the 1960s, especially in the less fertile mountain and hilly areas, continued in the 1980s. In the study area, the bulk wine that was produced did not have a large market and many farmers abandoned the vineyards. Today, on the contrary, there is a recovery of vineyards, with construction of modern and technological cellars and the support of expert winemakers. However, tree plantations, that take the place of arable land, are quite widespread in the region (17 700 ha; INARBO Project, 2017) also promoted by financial support in rural development plans.

The investigated tree plantation, that substituted for a vineyard, extends along a slightly-wavy slope on the hilly area of Rosignano Monferrato, Piedmont, Italy (45.09°N, 8.42°E). Among selected trees, poplars, being a fast-growing species, were planted with the aim of favoring the growth of the valuable tree species; they should have been cut after about 10 years of planting but the poplar cutting was not carried out to prevent the breakage of other trees due to the reduced planting distance.

As monitored by the long-term meteorological station near the study site, the yearly average rainfall is 877 mm and the mean air temperature is 11.7 °C. The slope is exposed to northwest, not so favorable for obtaining high quality grapes. We relied on space for time substitute study using a paired plot design. The common pedological origin of soils within the study area was verified and confirmed by comparability of soil texture and carbonates content of the deeper layer. According to the WRB classification (IUSS Working Group, 2015) the soils are Calcaric Cambisols (Loamic).

4. Possibility of scaling up

It is a context-specific case study, however, tree plantations on former arable land are quite widespread in the region.

5. Impact on soil organic carbon stocks

The tree plantation was characterized by a SOC of 74.4±21.1 t/ha (0-70 cm), higher than that of the nearby vineyard, which represents the SOC stock before afforestation (baseline SOC stock). In addition, OC stock in the organic horizons (not present in the vineyard) was 10.2 t/ha (Ferré *et al.*, 2019;

Table 28).

Table 28. Additional SOC potential from afforestation of a hilly area of Monferrato (Italy)

30 years of tree plantation resulted in a SOC stock increase of 26 percent in the first 70 cm, representing an additional SOC storage potential of 0.51 tC/ha/yr.

Location	Climate zone	Soil type	Baseline SOC stock (tC/ha)	Additional SOC storage (tC/ha/yr)	Duration (Years)	Depth (cm)	More information	Reference
Piedmont, Italy	Warm temperate moist	Calcaric Cambisol (Loamic)	59.2 ±23.6	0.51	30	0-70	Baseline stock is the SOC stock of the vineyard, representing the land use before afforestation	Ferré *et al.* (2019)

6. Other benefits of the practice

6.1. Benefits for soil properties

An increase in SOC stock in the mineral soil (from 59±23 to 74±21 t/ha) and the formation of an organic layer were observed.

Physical properties

We observed a slight improvement of the soil structure; the bulk density in the first 10 cm slightly decreased compared to VY (significant difference between tree plantation –1.2±0.1 g/cm^3- and vineyard -1.3±0.2 cm^{-3} was found.

Biological properties

The formation of an organic layer was observed; the main humus forms were Mull and Amphi, both characterized by a biomacrostructured A horizon.

6.2 Minimization of threats to soil functions

Table 29. Soil threats

Soil threats	
Soil erosion	The spatial distribution of soil properties in vineyard were affected by erosion and deposition dynamics unlike in tree plantation where vegetation reduced erosion. While at tree plantation the spatial distribution of SOC and pH were quite homogeneous, at vineyard there were variations along the slope with accumulation of SOC at the change of slope gradient and at the foot of the slope and with increase in pH values in the lower part of the slope, likely linked to redistributionof soil carbonates.
Soil biodiversity loss	We observed the formation of organic horizons and different humus forms like Mesomull, Oligomull, Dysmull, Leptoamphi and Eumacroamphi, characterized by different features and biological activity.
Soil compaction	The bulk density slightly decreased (see Section. 6.1).

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

The afforested area is suitable for timber exploitation. The large part of tree plantations of the region is for timber and biomass energy as they are mainly poplar plantations, but polycyclic tree plantations are also common. The tree plantation showed also production of truffles.

6.4 Mitigation of and adaptation to climate change

Tree plantations favour C sequestration. SOC storage increased (organic matter input increased, soil temperature and thus OC mineralization decreased and soil tillage was carried out in the initial phase of plantation only) and soil erosion decreased.

6.5 Socio-economic benefits

On slope with exposure not favorable for obtaining high quality grapes as the investigated one, the planting of valuable species may be a possible option to the vineyard.

6.6 Other benefits of the practice

Tree plantation can favor the presence of fauna that helps to control parasites of the vineyard.

Tree plantation benefits people well-being by acting as a buffer zone between cropland (vineyard may require recurring pesticide applications to control *Peronospora* and leafhoppers (*Empoasca flavescens*)) and the small hilly villages.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 30. Soil threats

Soil threats	
Soil acidification	Soil acidification (pH average difference of 0.4) was observed linked to a change in SOM type.

8. Recommendations before implementing the practice

It is important to carefully choose the tree species to be cultivated among those typical of the area taking also into account the soil properties.

9. Potential barriers for adoption

Table 31. Potential barriers to adoption

Barrier	YES/NO	
Economic	Yes	Currently the land use change from vineyard to forest is hampered mainly by economic reasons, since wine production is more profitable than timber exploitation and biomass energy production although, on the slopes with not optimal exposures for the vineyard or in areas with other limitations to the vine cultivation, forestry could be a possible alternative to the vineyard. At the moment in Italy the establishment (ranging 487-2591 € ha^{-1} depending on type of plantation and tree number) and initial management of tree plantation are quite completely financed by the regional rural development plans. Costs of vineyard removal and soil preparation are excluded only (InBioWood Project, 2018).
Institutional	No	Tree plantation introduction (land use change from cropland to tree plantation) is promoted and financed by rural development plans.
Legal (Right to soil)	Yes	Permanent conversion from agricultural land to forestry use; there is a worry of farmers for the conversion. As long as the agricultural use remains, they can choose and rotate tree plantation with any other crop.

Photos

Photo 17. The vineyard, representing the land use before afforestation (9.9.2019)

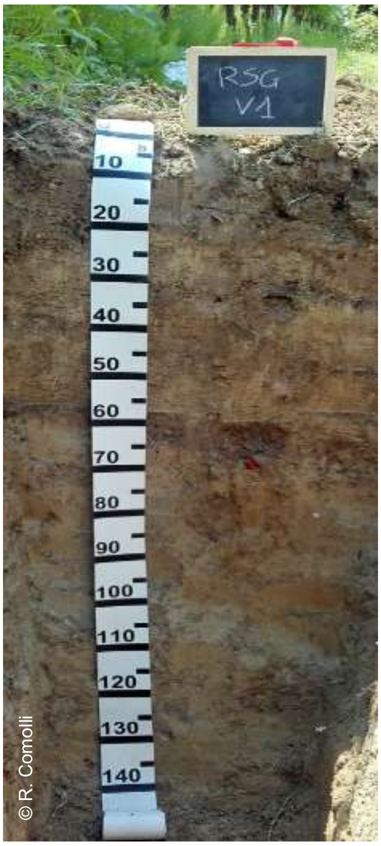


Photo 18. Calcaric Cambisols (Loamic, Aric, Ochric) at vineyard (1.07.2016)

Photo 19. Afforestation of vineyard: the tree plantation (9.09.2019)

Photo 20. Calcaric Cambisols (Loamic, Humic) at tree plantation (1.07.2016)

References

Ferré, C., Facciotto, G., Bergante, S. & Comolli, C. 2020. Effects of conversion from vineyard to tree plantation on humus forms, soil organic carbon stock and other soil properties. Oral presentation at European Geosciences Union (EGU), 4-8 May 2020.

INARBO Project. 2017. Inventario arboricoltura da legno in Italia www.reterurale.it/flex/cm/pages/ServeBLOB.php/L/IT/IDPagina/18647 by CREA - Research Centre for Forestry and Wood in the frame of National rural network (Mipaaft) with the support of FederlegnoArredo.

InBioWood Project. 2018. LIFE12 ENV/IT/000153 - LIFE+ InBioWood - Increase Biodiversity through Wood Production.

IUSS Working Group. 2015. World Reference Base for Soil Resources 2014, Update 2015. World Soil Resources Reports 106, FAO, Rome. ISBN 978-92-5-108369-7.

8. Afforestation by planting in bench terraces: Kalimanska watershed, Grdelica gorge, Southeastern Serbia

Sara Lukić, Snežana Belanović Simić, Milan Knežević, Predrag Miljković, Aleksandar Baumgertel, Jelena Beloica, Milica Caković

Faculty of Forestry, University of Belgrade, Belgrade, Serbia

1. Related practices

Afforestation/reforestation, bench terraces

2. Description of the case study

Land degraded by erosion is characterized by unfavorable conditions for the establishment of vegetation, but afforestation of these sites can create ecosystems able to provide various services (such as significant carbon sequestration potential). Forest restoration by afforestation or reforestation of degraded sites increase the C storage potential both in biomass and in soils. Choosing the right afforestation method may influence the afforestation success in the years after the stand establishment.

This case study was conducted on a degraded area suffering from severe erosion after years of extensive agriculture, in order to evaluate long-term effects of afforestation through planting Black Pines (*Pinus nigra* Arnold) in bench terraces (so called "gradone" afforestation method) in the Kalimanska watershed, Grdelica gorge, Southeastern Serbia. This practice was also used in several afforested sites in southern Europe (Serbia, Italy, North Macedonia, etc.) (Lukić *et al.,* 2015a; Mercurio and Schirone, 2015). Considering the extensive agriculture as previous land use led to extreme degradation and erosion, the selection of Black pine for planting was considered then as the best choice for erosion control in the given conditions because of its pioneering and xerothermic features. Soil carbon stock and soil carbon accumulation rate were selected as two main indicators to evaluate the efficiency of the afforestation method.

Trees were planted in bench terraces in several locations in the Grdelica gorge 60 to 70 years ago. The method of planting in bench terraces requires a specific site preparation including terracing along the contour line by making cuts and fills and forming the flat part of the bench terrace with a counter slope to promote the accumulation of water and organic matter and to prevent soil loss (Figure 4). The method of planting in bench terraces also involves special mechanical soil preparation in the planting pit. The root of the planted seedling is surrounded by soil from the surface layer, while the rest of the pit should be filled with chipped material from deeper layers of the soil (Djorović, Isajev and Kadović, 2003).

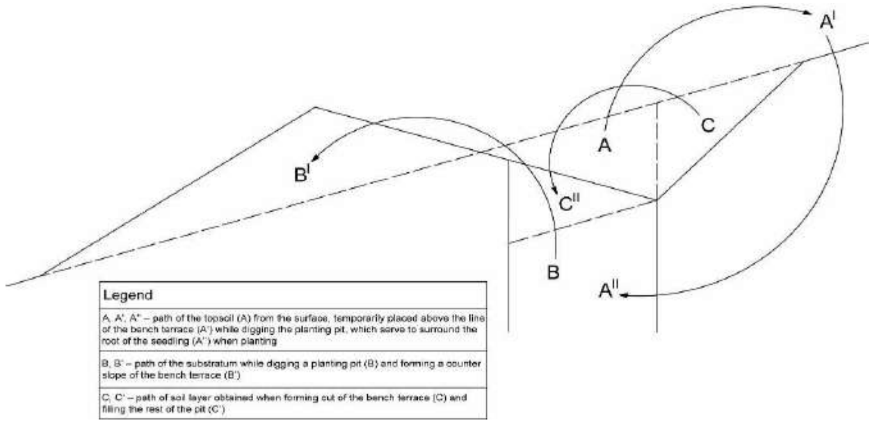


Figure 4. Formation of bench terrace and planting technique (Lukić *et al.,* 2015a)

In this case-study, the seedlings of black pine were planted on the formed counter slope of bench terraces along the contour line, in 40 × 40 cm pits, with a spacing of 1-2 m between the seedlings and a distance of 8-10 m between two adjacent bench terraces depending on the slope angle.

3. Context of the case study

The area of the Grdelica Gorge covers up to 1 784.34 km^2 and extends between 42°22′ and 42°55′ N and 21°19′ and 20°00′ E. It is characterized by its developed hydrographic network with 137 torrential streams and a total catchment area of 1 700.33 km^2. The area of the Grdelica Gorge is a site marked by specific conditions and vulnerable ecosystems requiring appropriate management methods. Climate is continental with an average annual air temperature of 10.9°C and an annual precipitation average of 672 mm (RHOS, 1949-2011). The harsh conditions in the area of the Grdelica Gorge are reflected in the erodibility of the parent rock displaying various degrees of weathering. In the 1950s afforestation was carried out by pit planting as well as in bench terraces (gradone) in this area. Kalimanska watershed environmental conditions represent a wider area of Grdelica gorge. In Kalimanska watershed (16.14 km^2) afforestation was applied by planting black pines in bench terraces (gradone) (Andrejević, 1959). This location as sample plot was singled out for the purpose of this report.

The parent rock of the sample plots are schists. The observed soil type is Leptosol (Dystric) (WRB, 2015) of light mechanical composition and poorly expressed or unexpressed structure. Sample plots were placed on terrains with a slope of over 30 percent, in warm exposures (SW - S), at altitudes ranging from 600 to 990 m a.s.l.

4. Possibility of scaling up

Planting in bench terraces is applicable in various sites conditions in geological and pedo-climatic context. It was applied in wide range of unfavorable environmental conditions of the mountain areas. The only constraint for application of bench terraces is related to the sites of potential landslides.

The site conditions define the species selection for planting. Species selection further affects the magnitude of benefits on soil properties, minimizing soil treads (e.g. foster nutrient balance and cycles, soil acidification and soil biodiversity loss), production (Food/Fuel), etc.

5. Impact on soil organic carbon stocks

The additional C storage was considerably higher in soil within the bench terraces compared to soil between adjacent bench terraces. The uncertainty of additional C storage potential in bench terraces was considerable higher than in soil between bench terraces (82.6 ± 27.4 tC/ha and 29.7 ± 5.5 tC/ha, respectively) which could be attributed to a specific intensive site preparation for planting in bench terraces. The baseline C stock was measured in 1956 but humus was determined by Kotzman's method. In studies from 1977 and 2015, humus was determined by Tyurin's method. Since the results obtained by these two methods are not comparable, that baseline C stock was calculated based on the study from 1977 (Table 32).

Table 32. Additional soil C storage value in sample plots after 60 years of afforestation

<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 (Years)</th><th>Depth (cm)</th><th>More information</th><th>Reference</th></tr>
<tr><td rowspan="2">Kalimanska watershed</td><td rowspan="2">Cool Temperate Moist</td><td rowspan="2">Leptosol (Dystric)</td><td rowspan="2">10.05</td><td>1.21</td><td rowspan="2">60</td><td rowspan="2">0-20</td><td>in planting line of bench terraces</td><td rowspan="2">Velašević (1977); Lukić et al. (2015a)</td></tr>
<tr><td>0.34</td><td>between adjacent bench terraces</td></tr>
</table>

6. Other benefits of the practice

6.1. Benefits for soil properties

Soil development was affected by afforestation in bench terraces through the formation of a humus-accumulative A horizon and an increase in soil depth.

Physical properties

Increased infiltration, reduction of soil losses and retention of organic matter affect soil aggregation. Soil bulk density (BD) decreased between 1977 to 2013. In 1977 BD ranged between 1.48-1.46 t/m3 and in 2013, in the planting line of bench terraces BD amounted 1.30-1.35 t/m3, while between adjacent bench terraces BD it ranged between 1.32-1.41 t/m3 (Velašević, 1977; Lukić, 2013).

Chemical properties:

CEC increases with the increase of organic matter content. The total capacity of cation exchange increased in period from 1977 to 2013. In 1977 the total capacity of cation adsorption was in range of 11.80-13.72 and in 2013, within the bench terraces the total capacity of cation adsorption was 17.88-20.95, while between adjacent bench terraces it was 12.88-14.08 (Velašević, 1977; Lukić, 2013).

Biological properties:

No data

6.2 Minimization of threats to soil functions

Table 33. Soil threats

Soil threats	
Soil erosion	Decrease in soil loss (terracing of the terrain and improving the infiltration in bench terraces lines) (Lukić, 2013; Braunović, 2013; Baumgertel *et al.*, 2018).
Nutrient imbalance and cycles	Organic matter increase (Lukić, 2013).
Soil acidification	Depends on planted species. Black locust performs beneficial effect on a decrease in the acidity of the surface soil layers and an increase in pH of the 0-5 cm layer compared to the same soil layer under black pine. (Lukić *et al.*, 2015b).
Soil biodiversity loss	Depends on planted species. Ammonifying bacteria and actinomycetes increased in soils under common laburnum and silver lime in contrast to soils under black locust and black pine, where enhanced mineralization processes and slower renewal of the humus horizon have been observed (Velašević *et al.*, 1977).
Soil water management	The specific sediment transport decreased from 1421.05 m3/ha2/yr (in 1953) to 364.39 m3/ha2/yr (in 2016) as a result of the biological, biotechnical and technical erosion control works performed in the torrential watersheds of Grdelica Gorge including Kalimanska watershed (Braunović, 2013; Kostadinov *et al.*, 2018).

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

The method was applied with the primary goal of protection against physical degradation by erosion. It may have a positive effect on the production of food (medicinal and aromatic plants and fruits (blueberries, blackberries, fruit trees...) and on the production of firewood obtained within the regular care measures of the stand. The results of the floristic research conducted 60 years after afforestation in the forests planted by Black Pine in bench terraces showed the emergence of medicinal and aromatic plants (*Hypericum perforatum*, *Asperula odorata*, *Stachys officinalis*, *Ahillea millefolium* etc.) and edible plants (*Fragaria vesca*, *Rubus hirtus*) (Lukić, 2013).

6.4 Mitigation of and adaptation to climate change

This method directly affects the increase in soil resistance to erosion processes by increasing its infiltration capabilities and organic matter content, consequently increasing C storage in soil and in biomass. Also, this technique enables better survival of seedlings in the first years after afforestation in extremely degraded sites and it could be considered as climate adaptive afforestation technique.

6.5 Socio-economic benefits

Demographic-economic analysis of erosion processes and sustainable soil management in Grdelica gorge showed that depopulation in the area of Grdelica Gorge resulted in the decrease of soil erosion. Also, the increase in age of the remaining population affected the change in agricultural practices. In the households of older people, traditional arable practices without conservation measures were substituted with orchards and grasslands as better management practices in hilly regions. Sustainable soil management should comprise both the application of erosion control measures including afforestation and planning of agricultural production in line with principles of sustainable development, which should result the improvement of human wellbeing in this area (Zlatić, 1998; Babović 2016). From the aspect of direct benefit for the local people, the application of this practice (planting in bench terraces) provides the torrential floods risk reduction, obtain non-timber forest products as well as the employment opportunities in forestry.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 34. Soil threats

Soil threats	
Soil acidification	The method of planting in bench terraces is not directly in conflict with soil acidification, but species selected for planting may have adverse impact (Table 33).
Soil biodiversity loss	The method of planting in bench terraces is not directly in conflict with soil biodiversity, but species selected for planting may have adverse impact (Table 33).

7.2 Conflict with other practice(s)

Livestock breeding: Excludes grazing on treated areas.

The areas treated by planting in bench terraces are initially prone to degradation and grazing has negative impact because livestock contribute the soil compaction and harm both young seedlings and herbaceous vegetation, which slow down cover closing and reduces the effects of the method applied.

8. Recommendations before implementing the practice

It is necessary to:

- Analyze the conditions of the habitat beforehand in order to exclude the possibility of applying lower cost afforestation methods. Since planting in bench terraces requires large investments, it is recommended for planting on extremely degraded sites. If the site conditions allow application of the lower-cost methods with satisfying success in seedlings survival and canopy closing fast enough it is recommended to select the lower-cost afforestation methods.
- Exclude grazing on forested areas. Disruptions of young seedlings and herbaceous vegetation by livestock slow down growing rate of vegetation, reduce degradation control effect and carbon accumulation.

9. Potential barriers for adoption

Table 35. Potential barriers to adoption

Barrier	YES/NO	
Economic	Yes	Large investments in establishment of bench terraces (Đorović *et al.* 2003) which mostly requires human labor to perform the works properly in mentioned conditions.
Legal (Right to soil)	No	The Law on Soil Protection (Official Gazette of RS 112/2015). Obligations to bringing protection plans and soil protection measures were defined in the Law on Soil Protection (Official Gazette of RS 112/2015), including the obligations to reclamation of degraded areas (Article 15, 18 and 22).

Photos

Photo 21. Bench terraces (gradone) in Kalimanska watershed

Photo 22. Soil profile between adjacent bench terraces (gradone)

References

Andrejević, M. 1959. Are the bench terraces novelty for our country? *Šumarstvo,* 5-6: 268-272.

Babović, S. 2016. *Influence of anthropogenic factors on intensity of erosion in southeastern Serbia.* University of Belgrade Faculty of Forestry. PhD dissertation.

Baumgertel, A., Lukić, S., Belanović Simić, S. & Miljković, P. 2018. *Effects of ameliorative afforestation on the erodibility factor and soil loss in the Grdelica Gorge*. Bulletin of the Faculty of Forestry, University of Banja Luka 278:37-46

Braunović, S. 2013. *Effects of erosion control works on the state of erosion in Grdelička klisura and Vranjska kotlina*. University of Belgrade Faculty of Forestry. PhD dissertation.

Djorović, M., Isajev, V. & Kadović, R. 2003. *Systems of afforestation and grass cover in erosion control.* Faculty of Forestry University of Banja Luka, Republic of Srpska, Bosnia and Hertzegovina.

Kostadinov, S., Braunović, S., Dragićević, S., Zlatić, M., Dragović, N. & Rakonjac, N. 2018. Effects of erosion control works: Case study—Grdelica gorge, the South Morava River (Serbia). *Water*, 10: 1094. https://doi.org/10.3390/w10081094

Lukić, S. 2013. *The effects of ameliorative afforestations in Grdelička gorge and Vranjska valley*. University of Belgrade Faculty of Forestry. PhD dissertation.

Lukić, S., Kadović, R., Knežević, M., Beloica, J., Đukić, V. & Belanović Simić, S. 2015a. Soil carbon accumulation as a response to the afforestation method used in the Grdelica gorge in southeastern Serbia. *In* V. Ivetić, D. Stanković (Eds.) *Proceedings: International conference Reforestation Challenges*. 03-06 June 2015, Belgrade, Serbia. Reforesta. pp. 104-116.

Lukić, S., Pantić, D., Belanović Simić, S., Borota, D., Tubić, B., Djukić, M., Djunisijević- Bojović, D. 2015b. Effects of black locust and black pine on extremely degraded sites 60 years after afforestation - a case study of the Grdelica Gorge (southeastern Serbia). *iForest*, 9: 235-243. https://doi.org/10.3832/ifor1512-008

Mercurio, R. & Schirone, B. 2015 Black pine reforestation of the Appenines in the Abruzzi region (central Italy): Perspectives and management *In* V. Ivetić, T. Ćirković-Mitrović (Eds.) *Book of Abstracts: International conference Reforestation Challenges*. 03-06 June 2015, Belgrade, Serbia. 36

RHOS. (1949-2011). Meteorological Annual Reports. Republic Hydrometeorological Office of Serbia, Belgrade, Serbia

Soljanik, I. 1955. Research afforestation at Grdelica gorge. *Šumarstvo,* 12: 741-756.

Velašević, V. 1979. *Study of ecosystems' disorders and degradation of the environment.* (Research Project Report V-11), University of Belgrade Faculty of Forestry, Serbia

WRB IUSS Working Group. 2015. *World Reference Base for Soil Resources 2014, update 2015. International soil classification system for naming soils and creating legends for soil maps*. World Soil Resources Reports No. 106. FAO, Rome.

Zlatić, M. 1998. Demographic-economic aspects of erosion processes and sustainable soil management in hilly-mountainous regions, Serbia. *In IAHC and WASWAC: Headwaters: water resources and soil conservation*. AA. Balkema/Rotterdam/Brookfield, 391-398.

9. Conservation of degraded forests of central and western Spain

Juan F. Gallardo

Prof. and Senior Researcher, retired. C.S.I.C., IRNASa, Salamanca, Spain

1. Related practices

Forest conservation through community protection, Silvo pastoralism.

2. Description of the case study

This case-study describes some characteristics of deciduous oak forests in Central/Western Iberian Peninsula, giving data on carbon sequestration if they are used as open forests hosting grazing activities (sylvopastoral systems), with complementary activities as hunting or beehives. A comparison is made with European chestnut plantation used for timber production.

Low-productive deciduous forests are abundant in the Central Western and Northern Iberian Peninsula. Climax forests are mainly associated with *Quercus pyrenaica* (a deciduous, broadleaf oak), and other oak species (e.g. *Q. robur, Q. rotundifolia* in drier sites) (Rivas-Martínez and Loidi-Arregui, 1999). Although oaks are still dominant, because of their low productivity they have been replaced in some areas by *Castanea sativa* (European chestnut) and, lately, by *Pinus pinaster* (Maritime pine). These forests were of high importance for charcoal production before the appearance and dissemination of butane gas in the 1960s.

As a consequence of the cessation of charcoal production, these managed forests were abandoned and slowly became high forests (i.e. forests reproduced by seeds). The lack of active management (cleaning) and local interests in the exploitation of such forests make them more prone to both natural and human-induced fires, which are recurrent every summer (Chas-Amil, Touza and Presteman, 2010). Although the impact of forest fires on SOC contents is not always significant, fires have major effects on soil erosion and biodiversity losses.

Conservation activities have therefore been initiated, including the formation of open high forests enabling grazing activities (sylvopastoralism), and the potential replacement of oaks by Maritime pine after large fires or on very degraded soils, or chestnuts used for timber production. Provincial authorities have also supported forest management activities such as thinning, clearing, pruning or the establishment of open areas for grazing or recreation.

Nowadays, deciduous-oak forests cover about 6 000 km^2 of the Western Iberian Peninsula, in addition to other 1 300 km^2 of chestnut coppices and orchards, with variable tree density. Small villages (there are very few big cities in the area) persist inside this large region, where older people live and look after these forests, also performing rudimental agroforestry activities. Forest conservation is therefore seen as a means of maintaining economic activities in these rural areas.

3. Context of the case study

The case-study includes four monitoring sites (Navasfrías, Villasrubias, Fuenteguinaldo and San Martín Trevejois) located in the "*Sierra de Gata*" Mountains (Central Western Spain), very close to the Portuguese border. The climate is Mediterranean sub-humid with long dry periods during summer and classified as Warm temperate Moist according to the IPCC classification (2019). Annual rainfall ranges between 750 to 1 600 mm/yr and mean annual temperature is ca. 12 ºC. Soils are mainly *cambic Umbrisols*, with some *umbric Lithosols* scattered (WRB, 2015) in the area. Studied forests are open forests with relatively low tree density (400 to 1 000 trees/a).

4. Possibility of scaling up

The case-study can be scaled up from northern and north-eastern Spain to some developed countries of central and western Europe with similar or related climates and soils, and socioeconomic standards, where large territories are abandoned by the widespread processes of industrialization and urbanization.

5. Impact on soil organic carbon stocks

These forests have a considerable potential of C sequestration, mainly due to the relatively high rainfall, moderate temperatures (mountain area) and dominant acidic soils. C sequestered by the systems increase with the rainfall that, in turn, promotes soil acidity (soil acidity limits microbial activity, resulting in an accumulation of SOC; Gallardo-Lancho, 2015) making all these soils rich in SOM (*Umbrisol*s, showing very dark colors). The amount of sequestered SOC significantly depends on the soil depth, and the main limitations for C sequestration are soil erosion and rock outcrops.

The increase of forest rotation allows a high value of C sequestered by the system, even if the forests are thinned (Table 36). Rotation time for chestnuts is about 20-25 year, and about 80 for deciduous oaks. Table 36 shows some actual values of C pool in four selected forests. Ages of these deciduous oak forests ranged 40-60 year when the base lines were established, and around 22 years for the chestnut coppice. Total organic C sequestered in these open deciduous-oak forests ranged between 110 to 165 tC/ha, while the potential C sequestration of the chestnut coppices before timber can reach 300 tC/ha. Beside this, SOC contents range between 60 to 130 tC/ha for the oak forests and up to 250 tC/ha in the chestnut coppice. Therefore, these forests can be considered as C pools (and sinks) (Bravo, 2007), considering that SOC is scarcely affected by forest fires. Other authors recorded a total of 228 tC/ha for deciduous-oak forests of 80 year and 270 tC/ha for Maritime-pine forests of 99 year. The aboveground biomass measured were 155 tC/ha for deciduous-oak forests of 80 year, and 159 tC/ha for Maritime-pine forests of 99 year, i.e. both have similar values (Bravo, 2007; Díaz-Pinés *et al.*, 2011; Alvarez *et al.*, 2013).

Table 36. Carbon and potential carbon storage increase in five selected forests of the Sierra de Gata region

Baseline stocks correspond to stocks when the practice was implemented.

C storage potential represents storage in the whole system (soil + biomass).

Location	Precipitations (mm/yr)	Soil type	Baseline C stock (tC/ha) (optimal)	Additional C storage (tC/ha/yr)	Duration of Rotation (Years)	More information	References
Navasfrías	1500	Cambic Umbrisol & Umbric Leptosol	160	1.0	80	Deciduous Oak Forest, thinning &grazing	Gallardo (2000)
Villasrubias	900		120	1.0			Gallardo and González (2004b)
Fuentegui-naldo	750		110	1.2			
San Martín Trevejo	1200	Cambic Umbrisol	300	5.0	22	Chestnuts for timber	Gallardo and González (2004a)
Sierra Guadarrama	850	Umbrisol & Cambisol	83	0.7	80	Deciduous oak forests	Gracia *et al.* (2005)
			150	0.8	60	Scotch pine	Bravo (2007)

6. Other benefits of the practice

6.1. Benefits for soil properties

Owing to the high contents of SOM, bulk density (and erosion) is usually low. The main achievement attained with these open forests is the control of erosion and preservation of natural areas. From the point of view of production, soil characteristics are paradoxically more adverse when the rainfall increase beyond 1 000 mm/yr, because of the increased soil acidity (Gallardo, 2000). In fact, erosion symptoms indicate certainly that the original SOC values have significantly decreased (in general, for human activities or overgrazing). More information on soil properties is available on Table 37.

Table 37. Soil properties of the monitoring sites

Stand litter content corresponds to the superficial layer of organic residues on the topsoil.

Soil properties where measured and monitored from 1992 to 2000 and are available in Gallardo (2000).

	Bulk density (t/m^3)	pH	Soil total N (gN/kg)	C/N	SOC (tC/ha)	Stand litter content (tC/ha)
Navasfrías	0.73	4.9	4.6	21	130	5.3
Villasrubias	0.84	4.6	4.0	20	90	5.8
Fuenteguinado	1.0	5.4	3.4	13	60	4.6
San Martín de Trevejo	0.96	5.1	2.1	19	200	7.2

6.2 Minimization of threats to soil functions

Table 38. Soil threats

Soil threats	
Soil erosion	When oak forests or chestnut coppices are preserved the risk of soil erosion diminishes, even in pronounced slopes (Gallardo-Lancho, 2000); furthermore, these areas have usually low cattle load which limits the risk of erosion.
Nutrient imbalance and cycles	Although this has not been specifically assessed, it is expected that the reduced erosion and improved soil properties resulting from the forest community management also have positive impacts in terms of nutrient balance and cycles.
Soil contamination / pollution	There is no soil contamination on the study sites. Soil contamination has only been found in the surroundings of mines (deposition of atmospheric dusts).
Soil biodiversity loss	Limited human impact usually preserves high biodiversity.
Soil compaction	Limited because density of cattle is usually very low.
Soil water management	Small areas are irrigated for local vegetable production and summer meadows.

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

In Portugal, chestnut-fruit collection is a popular secondary activity as important as potato production. Maintaining and conserving the deciduous oak forest by introducing cattle (or goats) in open forest allows to keep producing on these areas.

6.4 Mitigation of and adaptation to climate change

Projected climate change can improve the production of these oak forests because the lowering of soil acidity (reducing exchangeable Al^{3+} content) at the sites with excess of water fluxes, leading to a diminution of the soil leaching and losses of bio-elements (Gallardo, 2002). As an example, when the rainfall is higher than 1 500 mm/yr annual production of the oak forests is 2.6 tC/ha/a; when the rainfall is about 750 mm/yr, production increases to 4.2 tC/ha/yr. For comparison, mean annual production of chestnut is about 5.7 tC/ha/yr.

6.5 Socio-economic benefits

Maintaining of the landscape favors tourism activities. A limited market could be established based in local products (e.g. honey, chestnuts, or meat), besides the production of these forests and husbandry.

6.6 Other benefits of the practice

Landscapes of these forest offer an incomparable sightseeing and fresh atmosphere.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 39. Soil threats

Soil threats	
Nutrient imbalance and cycles	The turnover of bio-elements is moderately quick (about 3–5 years, depending on the mean annual temperatures). In some sites there is serpentine (excess of Mg).
Soil acidification	Risk of soil acidification in the wetter sites.

7.2 Increases in greenhouse gas emissions

No risks of other GHG emissions, except in small patches (e.g. stables).

7.3 Other conflicts

If forests are not conserved, they are likely to be abandoned, therefore increasing the risks of forests fires, whether spontaneous (uncontrolled, by increase of dry organic residues), or 'social' (voluntary provoked; Chas-Amil, Touza and Presteman, 2010). Likewise, legal restrictions to manage forests can lead to social issues increasing the threats to these forests. Indeed, when local people do not agree with these legal restrictions to use local natural resources, a known (and easy) solution is burning forests. These 'social' fires are unfortunately quite frequent in all the area (Spanish Central Range, Galicia, and Northern Portugal; Chas-Amil, Touza and Presteman, 2010).

8. Recommendations before implementing the practice

- Regulations on forest management should be decided in agreement with the landowners and the local populations;
- A solution to avoid the risk of social fires would be to include areas of meadows in appropriate sites, to enable local farmers to raise their cattle in the area, or to perform their farming activities.
- Dissemination of forest managements is advisable although this management does not require sophisticated knowledge;
- Although fertilization of open high forests is not common and not recommended, in some places, liming is recommended to increase grass production on very acidic soils, or excess of Mg (serpentine).

9. Potential barriers for adoption

Table 40. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	No	Open forests are usually considered as cultural heritage.
Cultural	No	Open forests of central Spain have a long story of management and inclusion of farming activities, therefore their management does not present any cultural issue.
Social	No	These areas concur with a limited development, depending economically on the incomes from the Central government or the European Union's supports.
Economic	No	These activities produce economic incomes for inhabitants.
Institutional	Yes	National and local legislation for public and private forests (e. g. Spanish "*Ley de Montes*").
Legal (Right to soil)	Yes	National legislations on forests and protected areas sometimes go against local interests.
Knowledge	No	With adequate instruction, appropriated management can be adopted by the high mean-age of the inhabitants of these areas.
Natural resource	Yes	These forests are located sometimes inside natural or protected areas, but other places are not considered because of the low tree density, disperse spots of different species, or scattered little forests.
Other	Yes	These areas are frequently visited by tourists or by hunters; but these activities are not sometimes allowed by the legislation of the Protected areas.

Photos

Photo 23. Different forests (left, Quercus rotundifolia, autumn and early summer; middle, Castanea sativa, summer; and right, Pinus pinaster, just after a forest fire) located at the district of 'Sierra de Gata' Mountains in Central-Western Spain Distances among the three plots are few kilometers; each soil (Umbrisol) is shown below each forest species)

References

Álvarez, S., Ortiz, C., Díaz-Pinés, E. & Rubio, A. 2013. Evolución de los reservorios de carbono de masas forestales dominadas por *Quercus pyrenaica vs Pinus sylvestris*. *In II Workshop sobre Mitigación de Emisiones de Gases de Efecto Invernadero Provenientes del Sector Agroforestal.* Aula Dei, Zaragoza, Spain. pp. 187-189.

Bravo, F. (Ed.). 2007. *El papel de los bosques en la mitigación del cambio climático*. Fundación Gas Natural, Barcelona. 320 pp. ISBN: 978-84-611-6599-5.

Carvalho, J.P. (Ed.). 2005. *O Carvalho negral.* AGRO, Vila Real (Portugal). 208 pp. ISBN: 972-669-624-0.

Chas-Amil, M.L., Touza, J. & Prestemon, J.P. 2010. Spatial distribution of human-caused forest fires in Galicia (NW Spain). pp. 247–258. Paper presented at FOREST FIRES 2010, 28 May 2010, Kos, Greece. (also available at http://library.witpress.com/viewpaper.asp?pcode=FIVA10-022-1).

Díaz-Pinés, E., Rubio, A., Van Miegroet, H., Montes, F. & Benito, M. 2011. Does tree species composition control soil organic carbon pools in Mediterranean mountain forests? *Forest Ecology and Management*, 262(10): 1895–1904. https://doi.org/10.1016/j.foreco.2011.02.004

Gallardo-Lancho, J.F. 2000. Biogeochemistry of Mediterranean forest ecosystems: A case study. In J.M. Bollag & G. Stotzky (Eds.) *Soil Biochemistry*, Marcel Dekker, New York. 10: 423-460.

Gallardo-Lancho, J.F. 2015. *La materia orgánica del suelo*. SiFyQA (Ed.), Salamanca, Spain. ISBN: 978-84-937437-7-2.

Gallardo-Lancho, J.F. & González-Hernández, M.I. 2004a. Sequestration of C in a Spanish chestnut coppice. *Invest. Agrar: Sist. Recur. For.*, vol. extra, 108-113.

Gallardo Lancho, J.F. & González Hernández, M.I. 2004b. Sequestration of carbon in Spanish deciduous oak forests. *Advances in Geoecology* (No.37): 341–351.

Gracia, C., Gil, L. & Montero, G., 2005. Impactos sobre el sector Forestal. In: *Evaluación preliminar de los impactos en España por efecto del cambio climático*. Impactos del Cambio Climático en España, Informe final Proyecto ECCE. Ministerio de Medio Ambiente-UCLM, Madrid, Spain. Páginas 399-436.

IPCC. 2019. Consistent representation of lands. *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories*, p. 55. (also available at: https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch03_Land%20Representation.pdf).

Rivas-Martínez, S. & Loidi-Arregui, J. 1999. Bioclimatology of the Iberian Peninsula & Biogeography of the Iberian Peninsula. *Itinera Geobot.*, 13: 41-47 & 49-67.

TRAGSA. 2009. Robledales de *Quercus pyrenaica* y robledales de *Quercus robur* y *Quercus pyrenaica* del Noroeste ibérico. Ministerio del Medio Ambiente, Madrid. ISBN: 978-84-491-0911-9.

WRB. 2015. *World reference base for soil resources 2014, update 2015 International soil classification system for naming soils and creating legends for soil maps*. IUSS Working Group WRB. World Soil Resources Reports No. 106. Rome, Food and Agriculture Organization of the United Nations. 192 pp. (also available at http://www.fao.org/3/a-i3794en.pdf).

10. Straw mulch and biochar application in recently burned areas of Algarve (Portugal) and Andalusia (Spain)

Sergio Alegre Prats[1,3], Franciscus Verheijen[1], Agustin Merino[2], Jose Antonio Gonzalez-Perez[3], Jose Maria De la Rosa[3]

[1]Centre for Environmental and Maritime Studies (CESAM), Dept. Environment and Planning, University of Aveiro, Aveiro, Portugal

[2]Escuela Politécnica Superior, Soil Science and Agricultural Chemistry, University of Santiago de Compostela, Lugo, Spain.

[3]Instituto de Recursos Naturales y Agrobiología de Sevilla, Consejo Superior de Investigaciones Científicas (IRNAS-CSIC), Seville, Spain

1. Related practices and hot-spots

Restauration of forested areas affected by wildfires, straw mulching, biochar; Forests

2. Description of the case study

High severity wildfires consume or alter soil organic matter (OM), thereby causing a drastic alteration of the carbon cycle and usually are followed by strong runoff and erosion events (Shakesby, 2011). Post-fire straw mulching has been used to mitigate post-fire soil erosion (Fernandez and Vega, 2016), both at small and micro-catchment scales (Prats *et al.*, 2019b); however, no studies have assessed the effects of the combined application of a layer of straw, which hinders soil erosion, over a layer of biochar which can restore the soil organic carbon lost to the atmosphere during the wildfire.

The effects of novel post-fire straw mulch/pine biochar treatments on the yield and fate of eroded sediments were assessed from two sites that burned at high (SW Portugal) and moderate severity (SW Spain). Just after the wildfire, 24 erosion plots were installed and three treatments were applied in 1 m^2 plots (with 4 replicates) in each burned area (**Photo 24**): untreated (burned plots without treatment); straw mulch (1 t/ha); and straw mulch + biochar (1+15 t/ha). Total Carbon contents were measured for the soil, the treatment materials and the eroded sediments with a Flash 2000 HT elemental micro-analyser.

The treatment effects in mitigating soil erosion and C losses were monitored for the most intense rainfall period in each study area. Literature about post-fire soil erosion indicates that the most intense rainfall event causes half the erosion of the first post-fire year (González-Hidalgo *et al.* 2009; Prats *et al.*, 2019b). A first publication is being prepared and will be sent to a journal very soon (Prats *et al.*, 2020).

The average erosion on the untreated plots for the two sites was 3.6 t/ha, and the mulch and mulch+biochar treatments reduced this figure in 69 and 70 percent, respectively (Prats *et al.*, 2020). The sediments, from the mulch+biochar treatment, were slightly enriched in organic carbon, which contained some biochar and were less dense and higher in water holding capacity. Chemical analysis indicated no clear alteration of the OM quality of the untreated and mulch-treated sediment, but the sediments produced in the mulch+biochar were enriched in refractory carbon (Prats *et al.*, 2020). Yet, the amount of eroded OM was lower for the mulch+biochar than the untreated plots for both sites. These results indicate that applying these treatments can restore the soil carbon stocks that were lost by high severity wildfires through: 1) an increase the labile OM fraction, due to the straw application, 2) an increase in the refractory OM fraction due to the application of biochar and 3) the retention of the native OM fractions of the soil due to the mitigation of erosion (Prats *et al.*, 2020).

3. Context of the case study

The Portuguese burned site is located in the Algarve region, near the Monchique hamlet (37°17′37.2″N; 8°33′31.3″W) (SW Portugal). The hillslope (steepness, 13°; east aspect, N110E) was affected by a high severity wildfire that destroyed 27 000 ha of shrubland and forest on 29th August 2018. The vegetation was composed of cork oaks (*Quercus suber* L.) and old eucalyptus stumps (*Eucalyptus globulus* Labill.), which were logged the year before the wildfire for paper pulp production. The climate is Mediterranean with oceanic influence, humid meso-thermal "Csb" (according to the Köppen classification), with prolonged dry and warm summers. Long-term mean annual rainfall and temperature were 1234 mm and 14.1 °C (SNIRH, 2018). Monchique soil parent material was sienite, a coarse-grained ultrabasic intrusive igneous rock, which resulted in 35-40 cm-depth, gravelly-sandy Eutric Cambisols (WRB, 2015).

The Spanish burned site is located in the Andalusia region, near the village of Casares (36°26′34.7″N; 5°14′09.5″W) (SW Spain). A moderate-severity wildfire (16-22 July 2018) consumed 230 ha of Mediterranean forest and shrubland in a steep hillslope (26°; east aspect, N90E). Pre-fire vegetation was composed of sparse *Quercus suber* L. and *Pinus pinaster* Aiton. and other Mediterranean trees of agronomic interest (*Ceratonia siliqua, Olea europaea*) with a dense shrubland understory of *Pistacia lentiscus, Arbutus unedo*, *Calicotome villosa, Cistus ladanifer, Chamaerops humilis* and annual plants (*Psoralea bituminosa, Phlomis purpurea, Rubia peregrina* and *Daphne gnidium*). The climate is Mediterranean "Csa" (Köppen classification), with prolonged dry and hot summers. Long-term mean annual rainfall and temperature averaged 868 mm and 17 °C. The soil parent material was composed of Palaeozoic slates and schists, which derived in a 35 cm-depth gravelly-loamy Eutric Cambisols WRB, 2015), with a high stone content and consequently, high bulk density.

4. Possibility of scaling up

This is a context-specific case study. The C-sequestration results can be scaled up to global scales, but first the experiment needs to be repeated on a representative sample of environmental conditions, before responsible upscaling can be considered. In the case of post-fire mulch, there already exists a large body of research, which makes the knowledge gap before sustainable upscaling much smaller than for the new mulch+biochar, which has potential benefits.

5. Impact on soil organic carbon stocks

The potential for C storage was calculated according to the measured C-stock baseline, immediately after the wildfire, minus the C stock one year after the fire. The C stock one year after the fire was calculated as the C-stock baseline plus the gains due to the application of the treatment materials (straw, biochar) minus the losses due to soil erosion. The mean annual soil erosion for the first post-fire year was 8 t/ha (Prats *et al.*, 2019b).

The potential soil C-storage for the Algarve site for the untreated plots resulted in the loss of 0.8 tC/ha (Table 41). The mulch and mulch+biochar treatments increased the C storage in 0.2 tC/ha and 12 tC/ha, respectively (Table 41). In the Andalusia site, the potential soil C-storage for the untreated plots resulted in the loss of 0.5 tC/ha (Table 41). The mulch and mulch+biochar treatments increased the C storage in 0.3 tC/ha and 12.4 tC/ha, respectively (Table 41).

The mulch and mulch + biochar treatments consistently increased the potential for soil C storage due to the effective reduction in soil erosion and the C gains due to the application of the treatment materials.

Table 41. Treatments effect on soil C storage

Location	Climate zone	Soil type	C stock baseline (tC/ha)	C stock one year post-fire (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Year)	Depth (cm)	More info	Reference
Algarve						1	0-10	Ongoing project	Prats *et al.* (2019b, 2020)
untreated	Temperate Csb	Eutric cambisol (sandy)	35.3	34.5	-0.8				
mulch			35.3	35.5	+0.2				
mulch+biochar			35.3	47.3	+12.0				
Andalusia									
untreated	Temperate Csa	Eutric cambisol (loamy)	22.6	22.1	-0.5				
mulch			22.6	22.9	+0.3				
mulch+biochar			22.6	35.0	+12.4				

6. Other benefits of the practice

6.1. Benefits for soil properties

All measurements were made on sediments eroded from the untreated, mulch and mulch+biochar treatments, so the physical and chemical properties are limited to the sediments.

For the mulch treatment:

Physical properties : increase in water holding capacity, water infiltration.

Chemical properties: slight enrichment in labile forms of soil organic matter.

For the mulch+biochar treatment:

Physical properties: increase in water holding capacity, water infiltration, decrease in bulk density.

Chemical properties: increase in pH, enrichment in labile and refractory forms of soil organic matter, which increase soil quality and will likely impact soil fertility.

6.2 Minimization of threats to soil functions

Table 42. Soil threats

Soil threats	
Soil erosion	Covering the soil with 1 t/ha mulch reduced soil erosion by 60-70 percent compared to untreated plots. Higher effectiveness can be achieved by increasing the straw application rate to 2 t/ha (Fernandez and Vega, 2016).
Nutrient imbalance and cycles	Increase in C storage, due to biochar addition will also increase the chances to store forest N, P and K (Campos *et al.*, 2020).
Soil contamination / pollution	It is recommended the use of certified EBC (2020) or IBI (2020) biochars.
Soil acidification	Not likely, as biochar increased soil pH (Campos *et al.*, 2020).
Soil biodiversity loss	Not likely, as biochar increased microbial biodiversity (Paneque *et al.* 2016).
Soil water management	Straw and biochar increase water infiltration and retention (Prats *et al.*, 2019a).

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

It is expected that an increase in soil carbon storage (as well as a lack of soil erosion) will increase soil quality, and hence a higher soil fertility for timber or fuel production.

6.4 Mitigation of and adaptation to climate change

Biochar is already considered a useful organic ameliorant for the restoration of degraded soils (Paneque *et al.*, 2016). In addition, the use of biochar to increase soil carbon has been found to be an effective way to increase C sequestration, especially if compared to the typical straw application. This is because biochars have a much longer residence time in the environment before they get degraded by microorganisms (decades to centuries; De la Rosa *et al.*, 2018). Thus, the addition of biochar to soils can be seen as a reservoir to store CO_2. It is recognized that the application of the treatment itself will require some emissions to the atmosphere, which should be considered and minimized by using straw and biochar produced locally.

6.5 Socio-economic benefits

Economically, forest and agricultural local residues are available to be converted into biochar, reducing the residue managing costs. The production of biochar via slow pyrolysis allows the generation of energy and the reduction of a residue, which may pose important socio-economic benefits to the local communities. It also will help to reduce fuel loads and hence, wildfire risk. Applying biochar will contribute to the long-term stabilization of carbon and will increase soil quality in degraded areas, especially in the case of burned areas in the Mediterranean regions at high risk of desertification.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 43. Soil threats

Soil threats	
Soil erosion	It is recommended that biochar is not applied alone in the soil surface but protected by a straw cover against rain splash detachment, to keep the biochar in place.
Soil contamination / pollution	The feedstock materials should be known, as some feedstocks can be toxic.
Soil biodiversity loss	The feedstock materials should be known, as biochars from unknown origin or not certified can be toxic to soil biodiversity.

7.2 Increases in greenhouse gas emissions

Several considerations can be made to infer the implications of the treatments on GHG emissions after 1 year of its application. In the case of the mulch+biochar treatment, the equivalent CO_2 stored in the burned soil is 28.2 t CO_2 /ha considering that:

a) the mean treatment C sequestration for the two study areas increased in 12.8 tC/ha as compared to the untreated area,
b) excluding the details of CO_2 emissions during biochar production and considering that an appropriate pyrolysis technology is used that takes advantage of the methane generated during the pyrolysis to transform it into energy (Campos *et al.*, 2020),
c) excluding the reductions in CO_2 emissions due to the lack of restoration operations (road restoration, emergency hillslope stabilization), as a result of lower post-fire flooding and siltation risk in mulch-treated areas (Robichaud, Rhee and Lewis, 2014),
d) considering that C sequestration is equivalent to a low, conservative figure of 2.2 t CO_2 per ton of biochar (Galinato, Yoder and Granatstein, 2011), and
e) assuming that C sequestration corresponded all to pyrogenic-like materials (the applied straw, biochar and the wildfire-charred not-eroded soil (Prats *et al.,* 2020).

Assuming that a low-emission heavy-duty vehicle, emitting CO_2 at a rate of 350 g CO_2 per kilometer (EU, 2020), can carry all the feedstocks (1 t straw + 15 t biochar) in a single travel and that the treatment will be hand-applied in one hectare of burned area, the feedstock transportation from a distance of 500 km will amount 0.18 t CO_2/ha, which is less than 1 percent of the C sequestration from the mulch+biochar treatment.

In the case of the mulch treatment, the equivalent CO_2 stored in the burned soil is 1.9 t CO_2/ha considering that:

a) the mean C sequestration for the mulch treatment corresponded to an increase in 1 t/ha as compared to the untreated area,
b) excluding the details of CO_2 emissions during biochar production,
c) excluding the reductions in CO_2 emissions due to the lack of restoration operations,
d) considering that C sequestration is equivalent to a low, conservative figure of 2.2 t CO_2 per ton of straw mulch (Galinato, Yoder and Granatstein, 2011), and
e) assuming that C sequestration corresponded all to pyrogenic-like materials (the wildfire-charred not-eroded soil).

Assuming that the same vehicle will equally carry the feedstock (1 t straw) from a distance of 500 km to the burned study area, the truck transportation will consume an 8 percent of the C sequestration from the mulch treatment.

7.3 Conflict with other practice(s)

This practice is rather in conflict with the usual "non-management" of burnt areas in Mediterranean burned areas. Some practices have been implemented with success (post-fire forest residue mulching, or straw mulching, (Fernandez and Vega, 2016; Prats *et al.*, 2019b) but they are rarely implemented, in spite of their high effectiveness. This is basically due to the lack of knowledge about the effects of wildfires on carbon sequestration and its mitigation.

8. Recommendations before implementing the practice

The feedstock materials should be known, because they can introduce seeds from undesirable exotic plant, such as eucalyptus or acacia. Biochar can be toxic, and thus, all biochars should be certified (EBC, 2020; IBI, 2020), as it was the case of our study case.

9. Potential barriers for adoption

Table 44. Potential barriers to adoption

Barrier	YES/NO	
Cultural	Yes	Stakeholders and forest owners in the area are still not aware of the advantages of protecting burned soils with a layer of mulch.
Social	Yes	Stakeholders and forest owners in the area are still not aware of the advantages of protecting burned soils with a layer of mulch.
Economic	Yes	Post-fire mulch is considered an expensive technique if its C sequestration potential is not considered.
Knowledge	Yes	Longer monitoring periods on different geologies, slope angles, precipitation regimes are needed to unveil the mean residence times of biochar and straw in the environments, and the implications for forest productivity.

Photos

Photo 24. Experimental untreated (a), straw mulch (b) and straw+biochar (c) mulched plots in the Monchique (Portugal) burned study site, as well as detail pictures of the soil surface for these treatments for Monchique (d, e, f) and Casares (g, h, i) burned sites

Acknowledgements

This research was carried out in the framework of the Research contract (CDL-CTTRI-88-ARH/2018 REF.-138-88-ARH/2018) for the first and fourth authors, funded by national funds (OE), through the Portuguese Foundation for Science and Technology (FCT/MCTES), in the scope of the framework contract foreseen in the numbers 4, 5 and 6 of the article 23, of the Decree-Law 57/2016, of August 29, changed by Law 57/2017, of July 19. Thanks are also due for the financial support to CESAM (UID/AMB/50017/2019), to FCT/MCTES through national funds, and the cofunding by the FEDER, within the PT2020 Partnership Agreement and Compete 2020.

References

Campos, P., Miller, A.Z., Knicker, H., Costa-Pereira, M.F., Merino, A. & De la Rosa, J.M. 2020. Chemical, physical and morphological properties of biochars produced from agricultural residues: Implications for their use as soil amendment. *Waste Management*, 105: 256–267. https://doi.org/10.1016/j.wasman.2020.02.013

De la Rosa, J.M., Rosado, M., Paneque, M., Miller, A.Z. & Knicker, H. 2018. Effects of aging under field conditions on biochar structure and composition: Implications for biochar stability in soils. *Science of the Total Environment*, 613-614: 969-976. https://doi.org/10.1016/j.scitotenv.2017.09.124

European Parliamentary Research Service, (EU). 2020. *CO_2 emission standards for heavy-duty vehicles*. [online]. [Cited 15 June 2020]. https://www.europarl.europa.eu/RegData/etudes/BRIE/2018/628268/EPRS_BRI(2018)628268_EN.pdf

European Biochar Certificate (EBC). 2020. The European Biochar Certificate [online]. [Cited 15 June 2020]. https://www.european-biochar.org/en

Fernandez, C. & Vega, J.A. 2016. Are erosion barriers and straw mulching effective for controlling soil erosion after a high severity wildfire in NW Spain? *Ecological Engineering*, 87: 132–138. https://doi.org/10.1016/j.ecoleng.2015.11.047

Galinato, S.P., Yoder, J.K. & Granatstein, D. 2011. The economic value of biochar in crop production and carbon sequestration. *Energy Policy*, 39(10): 6344–6350. https://doi.org/10.1016/j.enpol.2011.07.035

González-Hidalgo, J.C., Batalla, R.J., Cerdà, A. & De Luis, M. 2009. Contribution of the largest events to suspended sediment transport across the USA. *Land Degradation and Development*, 21: 83–91. http://dx.doi.org/10.1002/ldr.897

International Biochar Initiatives (IBI). 2020. Biochar Standards. [online]. [Cited 15 June 2020]. https://biochar-international.org/characterizationstandard/

Paneque, M., De la Rosa, J.M., Franco-Navarro, J.D., Colmenero-Flores, J.M. & Knicker, H. 2016. Effect of biochar amendment on morphology, productivity and water relations of sunflower plants under non-irrigation conditions. *Catena*, 147: 280-287. http://dx.doi.org/10.1016/j.catena.2016.07.037

Prats, S.A., Gonzalez-Pelayo, O., Silva, F.S., Bokhorst, K.J., Baartman, J.E.M., & Keizer J.J. 2019b. Post-fire soil erosion reduction with forest residue mulching at catchment scale. *Earth Surface Processes and Landform*, 44: 2837-2848. http://dx.doi.org/10.1002/esp.4711

Prats, S.A., Malvar, M.C., Coelho, C.O.A. & Wagenbrenner, J.W. 2019a. Hydrologic and erosion responses to compaction and added surface cover in post-fire logged areas: isolating splash, interrill and rill erosion. *Journal of Hydrology,* 575: 408-419. http://dx.doi.org/10.1016/j.jhydrol.2019.05.038

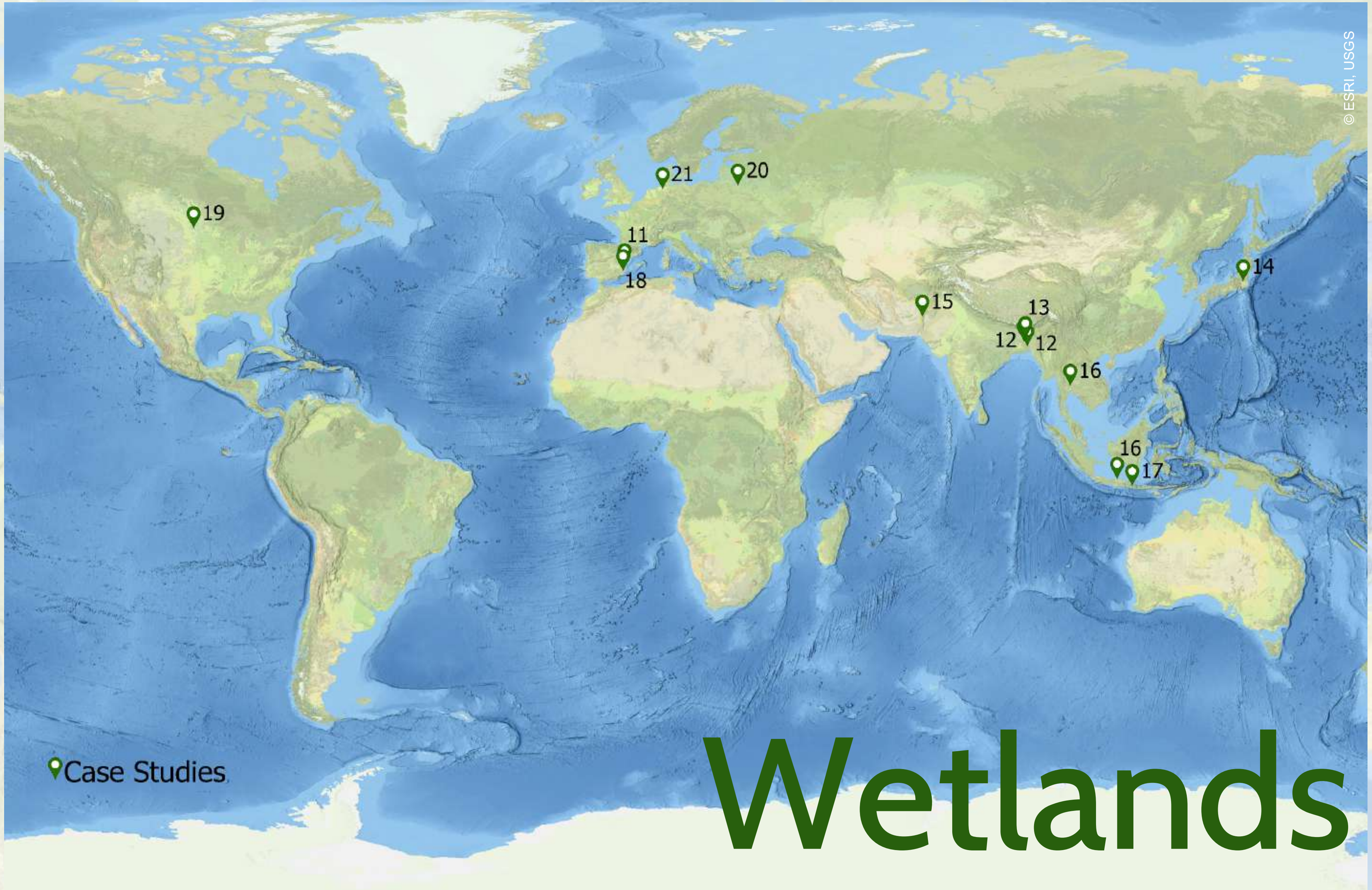
Prats, S.A., Merino, A., Gonzalez-Perez, J.A., Verheijen, F. & De la Rosa, J.M. 2020. Can straw-biochar mulching mitigate erosion of wildfire-degraded soils under extreme rainfall? *Science of the Total Environment*. https://doi.org/10.1016/j.scitotenv.2020.143219

Robichaud P.R., Rhee H. & Lewis S.A. 2014. A synthesis of post-fire Burned Area Reports from 1972 to 2009 for western US Forest Service lands: trends in wildfire characteristics and post-fire stabilization treatments and expenditures. *International Journal of Wildland Fire*, 23: 929–944. http://dx.doi.org/10.1071/WF13192.

Shakesby, R.A. 2011. Post-wildfire soil erosion in the Mediterranean: review and future research directions. *Earth-Science Reviews*, 105:71–100. https://doi.org/10.1016/j.earscirev.2011.01.001

Serviço Nacional de Informação sobre Recursos Hídricos, (SNIRH). 2018. Dados climatológicos sintetizados [online]. [Cited 15 December 2020] http://snirh.apambiente.pt

World Reference Base IUSS Working Group (WRB). 2015. *World Reference Base for Soil Resources 2014, update 2015, International Soil Classification System for naming soils and creating legends for soil maps.* World Soil Resources Reports No. 106. FAO, Rome.



Wetlands

Case Study ID	Region	Title	Practice 1	Practice 2	Practice 3	Duration
11	Europe	Management of Rice straw in Mediterranean wetlands, Spain	Mulching (straw)			7
12	Asia	Conservation Agriculture in intensive rice-based cropping systems in the Eastern Gangetic Plain	Conservation agriculture	Strip tillage	Mulch (straw)	5
13	Asia	Long term fertilization in a subtropical floodplain soil in Bangladesh	Integrated soil fertility management	Manure	Chemical fertilization	42
14	Asia	Organic rice cultivation with internal nutrient cycling in Japanese Andosols	Organic farming	Organic matter additions (straw residues)	Crop rotation	4, 8 and 12
15	Asia	Conservation tillage to tackle smog issue and improve carbon sequestration in rice-wheat cropping system in Pakistan	Conservation agriculture	Reduced tillage	Mulching	1
16	Asia	Water regimes in rainfed rice-paddies in Indonesia and Thailand	Alternate wetting and drying	Water sheet management		1 and 3

Case Study ID	Region	Title	Practice 1	Practice 2	Practice 3	Duration
17	Asia	Mangrove restoration in abandoned ponds in Bali, Indonesia	Mangrove restoration			10
18	Europe	Management of Common Reed (Phragmites australis) in Mediterranean wetlands, Spain	Wetland conservation	Mulching		NA
19	North America	Preserving Soil Organic Carbon in Prairie Wetlands of Central North America	Wetland conservation	Wetland restoration		NA
20	Europe	Biomass from reeds as a substitute for peat in energy production in Lida region, Grodno oblast, Belarus	Paludiculture			NA
21	Europe	Sphagnum farming for replacing peat as horticultural growing media, Lower Saxony, Germany	Paludiculture			NA

11. Management of rice straw in Mediterranean wetlands, Spain

Sara Ibáñez-Asensio, Héctor Moreno-Ramón

Department of plant production, Universitat Politècnica de València, Valencia, Spain

1. Practice(s) used

Rice straw management: incorporation into soil versus straw removal or burning in a marsh area.

2. Description of the case study

Rice is one of the most common crops in Spanish Mediterranean wetlands (Doñana, Delta del Ebro and Albufera de Valencia - the most important Spanish RAMSAR[6] sites). All these areas have the same environmental characteristics (coastal wetlands or deltas with similar geological material, hydraulic functioning and rice field management, among others) and normally accumulate SOC, but this situation is changing dramatically within the current climate change situation.

In these human-impacted wetlands, organic carbon accumulation depends on the water management regime and crop management, which are the most important factors the rice straw management. Farmers have three options with rice straw: burning, incorporation and removal.

In the past, straw used to be burned after harvest, causing respiratory problems in nearby municipalities, or increased GHG emissions. The incorporation of straw into rice fields or its removal for use in livestock are, up to now, the most widely used alternatives. In recent years, the regional government have been promoting rice straw extraction, even providing subsidies for it.

However, much of the straw is not collected (Photo 25) and eventually it is incorporated into the soil. In this case, once crushed, the straw remains mixed with the soil for at least 30-40 days before the winter flooding of the rice fields that takes place between October and February. The authorities can modify this usual situation

[6]Ramsar, The Convention on Wetlands is the intergovernmental treaty that provides the framework for the conservation and wise use of wetlands and their resources. More information on: https://www.ramsar.org/.

when they consider that there is a high incidence of some pest or disease, authorizing the burning of straw in plots (Photo 26).

All these actions have modified the rate of SOC accumulation in the area because it has been necessary to adapt the management to regional, national and European environmental regulations. In general, in the 15 500 ha of Albufera of Valencia, a homogeneous and standard management of rice straw has not been applied, and we can find plots where the straw is removed, incorporated or, if it is allowed, occasionally burned.

Another important item is water management, because rice-growing area is flooded between 8 and 9 months per year (including the cultivation cycle and winter flooding for migratory birds (Photo 27)). In this situation, water leads to slower accumulation of organic carbon and flooding conditions that favour CH_4 emissions. On the contrary, in dy fields, there is an oxidation of organic matter (losses) and an increase of CO_2 and N_2O emissions.

More specifically, during the rice cycle (May to September), soil is flooded with a 10- to 15-cm sheet of water, and this sheet is removed 1 or 2 times during the months of June and July to promote robust stem development and allow treatments to eliminate weeds. To date, no specific study has been published in the area for quantifying the impact on emissions from the reduction of water inputs in rice fields or as a function of water management strategies. Data from Delta del Ebro, located a little further north of the Albufera, indicated that intermittent irrigation led to significant higher N_2O and CO_2 emissions than continuous Irrigation. In that regard, continuous irrigation can significantly mitigate the integrative greenhouse effect caused by CH_4 and N_2O from paddy fields (Maris, Teira-Esmatges and Català, 2016). However, Jégou and Sanchis-Ibor (2019) found that Albufera has lost 40 percent of its water input in 30 years and that this gradual decrease in the amount and quality of water has been modifying the traditional management of water. In addition, soils and water are undergoing a salinization process and, consequently, are altering the decomposition process of organic matter (Moreno-Ramón *et al.*, 2015).

3. Context of the case study

The Albufera of Valencia is a coastal wetland of 21120 ha located at the east of the Iberian Peninsula (Spain) (Photo 28). It was declared a Natural Park in 1986 by the regional government and it has been classified as RAMSAR wetland and Special Protection Area for birds, among others special protection figures that reveal its importance as a Mediterranean wetland.

Rice in this area began to be cultivated in the 15th century when the Muslims developed a large network of channels and ditches to distribute the water from the rivers into the rice production area. Nowadays, the marsh area surrounding a central lagoon is cultivated and produces more than 12 104 tons of rice. The Mediterranean climate determines the crop cycle, which begins with the sowing in May and ends in September with the harvesting. Average rainfall is around 578 mm whereas annual evapotranspiration reaches 861 mm and average temperature is 17ºC. These data reveal the semiarid climate conditions with two periods of rainfall (spring and autumn seasons). Soils are carbonated, usually saline and with a moderate organic carbon content in surface as result of crop management. In addition, they are classified as Entisols and Aridisols (Moreno-Ramón *et al.*, 2015) due to the alluvial character of the area and the presence of a saline water table intrusion whose origin is caused by the proximity to the Mediterranean Sea.

4. Possibility of scaling up

This strategy can be used in many other coastal wetlands which have rice management in marsh areas.

5. Impact on soil organic carbon stocks

Albufera de Valencia and its entire surface area of 15 500 ha of paddy fields has a potential for carbon accumulation. The average SOC value in the profile is 30.7 g/kg (0-80 cm) and has an irregular profile in soil depth due to the presence of buried organic horizons that are related to the natural formation process of the wetland for the past centuries (Moreno-Ramón *et al.*, 2015). Regarding the management of rice fields in which the surface is tilled and turned in the preparation of the fields before sowing, the average value of SOC on the surface (0-20 cm) is greater than the average of the profile (32.2 g/kg). This value corroborates the outcomes presented by Kögel-Knabner *et al.* (2010) and Cui *et al.* (2014) who revealed that the amount of organic inputs after harvest (rice straw) is the main reason for the high values in the upper soil layer in relation to the normal content of hydric soils.

The recent abandonment of the practice of rice straw burning and its incorporation into the soil is also being translated into an increase of carbon in the soil. Data from a continuous study that is currently being carried out by Moreno-Ramón and Ibáñez-Asensio (not published yet) shows that between 2010 and 2017, the average content of SOC in the first 20 cm of the soil was 25.29 g/kg to 45.85 g/kg respectively in 42 control points distributed throughout the entire area (Table 45). Therefore, the carbon stock in the topsoil of the hydric soils in Albufera of Valencia increased by 54.77 tC/ha on average between the years, according to the partial data of temporal SOC research in the first 20 cm and a bulk density of 1.40 t/m^3.

Sanchis (2015) also identified this trend with statistically significant differences between rice straw incorporation and other types of waste management. Plots with straw incorporation registered SOC values of around 16.40 g/kg after the application compared to straw removal, which registered values between 11 - 11.7 g/kg. In the same vein, Verdejo (2016) also recorded that in paddy rice plots where the straw was removed from field, the average SOC at 0-20 cm was 25.8 g/kg, while those in which its SOC content was incorporated reached an average of 33.50 g/kg. Ribó *et al.* (2017) presented a different situation in a technical report in reference to an eight-year study in paddy fields that determined that in the last year SOC was similar and not statistically significant regardless of the treatment that the plots had undergone (burning, incorporation and removal). However, they do not have intermediate data that can guide us on the annual variation or on the initial situation.

Table 45. Evolution of SOC stocks in the 7-year trial

Location	Climate zone(MAP, MAT)	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration (Years)	Depth (cm)	More information	Reference
Albufera Valencia, Eastern Spain	Mediterranean (578 mm; 17 ºC)	Typic Fluaquent, Thapto-Histic Hydraquent, Typic Hydraquent and Typic Aquisalid pH: 7.91 1.52 t/m³ 25.29 gC/kg	76.74 (min: 24.15, max: 123.29)	7.82	7 (2010-2017)	0-20	Average data of 42 points (superficial samples surrounding the lagoon)	Moreno-Ramón and Ibáñez-Asensio (unpublished)

6. Other benefits of the practice

6.1. Benefits for soil properties

Crushing of the rice straw before straw incorporation favors its decomposition and facilitates adequate soil aeration. The incorporation of straw always supposes a contribution of organic matter to rice fields, which is translated into an important source of nutrients for the soil. Applying rice straw to paddy fields improves soil conditions because promotes nitrogen mineralization and nutrients availability and favors plant growth. In addition, soil organic carbon addition increases soil microbial biomass, soil enzyme activities and soil quality (Kongchum, 2005; Chivenge *et al.*, 2020; Zhou *et al.*, 2020).

On the other hand, when the rice straw burning is done in the Natural Park according to the legal directives each year, there are also a beneficial effect over soil because an important amount of mineral salts are incorporated into soil. In any case, generally in the Albufera, most of the straw is neither collected nor burned, incorporated into the soil.

6.2 Minimization of threats to soil functions

Table 46. Soil threats

Soil threats	
Soil erosion	Removed rice straw can be used in other near areas to stop soil erosion (González-Prieto *et al.*, 2018).
Nutrient imbalance and cycles	Rice straw incorporation can reduce the amount of soil fertilization applied during the field preparation for cultivation.

Soil threats	
Soil salinization and alkalinization	When rice straw is not burned, the input of salts can increase the salinization in the area where seawater intrusion affects the lowest fields and the nearest fields to the lagoon and the seashore is reduced. Incorporation of straw can increase mineral salt addition to the soil.
Soil contamination / pollution	There are no smoke problems on the nearest municipalities when rice straw is removed or incorporated.
Soil biodiversity loss	When rice straw is incorporated into soil, it acts as substrate for promoting the microfauna that favors the decomposition of organic matter (Schimdt *et al.*, 2015).
Soil compaction	When rice straw is incorporated, organic matter favors soil structure and therefore avoid soil compaction.

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

The study by Ribó *et al.* (2017) concludes that rice yields did not vary according to the type of straw management in short or medium periods. More specifically, after eight campaigns, the performance data of the final one did not show significant differences: 8.69 t/ha for burning, 8.45 t/ha for incorporation and 8.43 t/ha for rice straw removal. Chivenge *et al.* (2020) explain that at short-term period, straw removal does not show a difference and in a continuous flooding water regime, there is not an increase in rice yield. However, straw incorporation provides basic food for many micro-invertebrates that promote soil biodiversity.

6.4 Mitigation of and adaptation to climate change

In relation to GHG emissions, Sanchis (2015) concluded that rice straw removal was the management with the lowest emissions (5248 kg CO_2eq/ha). At the case of straw incorporation, GHG emissions reached 11708 kg CO_2eq/ha, but if straw is left on soil surface the outcomes are different. In that case, water management is very relevant because if paddy fields are flooded immediately, GHG reached 8440 kgCO_2eq/ha, whereas if paddy fields are not flooded the GHG reached 7802 KgCO_2eq/ha. According to FAOSTAT data in 2017, the average GHG emissions due to rice in Spain was 10584 kgCO_2eq/ha. Therefore, the key to reduce GHG emissions when the straw is incorporated into soil is to extend the period of dry fields after the harvest, until the straw has decomposed. Furthermore, this favors organic matter to decompose into much more stable forms of organic carbon in the soil (Jiang *et al.*, 2019). Verdejo (2016) applying the DNDC model with real data from the paddy fields (soil, climatic data, fertilization, water regime, etc.) concluded that plots without rice straw (removal) produced 5586 kgCO_2eq/ha, whereas plots with rice straw deposition on surface produced 6903 kgCO_2eq/ha.

6.6 Other benefits of the practice

The abandonment of the practice of burning rice straw clearly reduces the environmental pollution generated by the burning and formation of fumes.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 47. Soil threats

Soil threats	
Nutrient imbalance and cycles	The decomposition of the straw stubble in flood conditions that generally occurs some years due to the delay of the campaign owing to the rains causes the decomposition of organic matter in the fields. This situation causes the water on the surface of the fields to have a blackish color (Photo 29) as a result of dissolved organic matter, which reaches the lake through the ditches, increasing eutrophication problems.
Soil salinization and alkalinization	Burn straw rice (Photo 26) can increase soil salinity because an important amount of mineral salts that are incorporated into soil due to the direct mineralization.
Soil compaction	For rice straw elimination it is necessary to use heavy machinery, and wet soil conditions facilitate the compaction process (friction resistance is lower than soil is dry).

7.2 Conflict with other practice(s)

The Natural Park presents a winter flooding period to favor those migratory birds can rest on their journey to warmer areas. This action sometimes causes straw to be degraded in a lack of oxygen (anaerobiotic conditions) that cause the death of fish in some parts of the lagoon (recipient vessel of the waters of the rice fields) and therefore the fishing activity suffer the loss of catches.

8. Recommendations before implementing the practice

It is important to control the water management and the straw decomposition rate according to the climatic conditions of the area. It is normal to wait 30-40 days once the straw is incorporated to favor the aerobic decomposition. This action in addition, favors the reduction of water inputs, in areas where there is water scarcity.

9. Potential barriers for adoption

Table 48. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	The use of machinery in wetland areas is always difficult due to the instability of soil that depends on the proximity of the water table to the soil surface.
Cultural	No	Farmers have assumed that burning is the best way to solve the problem of rice straw. Traditionally this has been done and there is no economic cost generated.
Economic	Yes	Crushing the straw, incorporating it and collecting it has associated costs that farmers do not want to bear due to the low profitability of the crop.
Legal (Right to soil)	Yes	There is a regulation that prevents the burning of the straw and forces its incorporation or removal, but it is usually modified by the regional government.
Knowledge	Yes	Farmers must be made aware of the consequences and benefits of each practice.

Photos

Photo 25. Rice straw in paddy fields after harvesting

Photo 26. Rice fields burned

Photo 27. Rice fields inundation during winter months for migratory birds

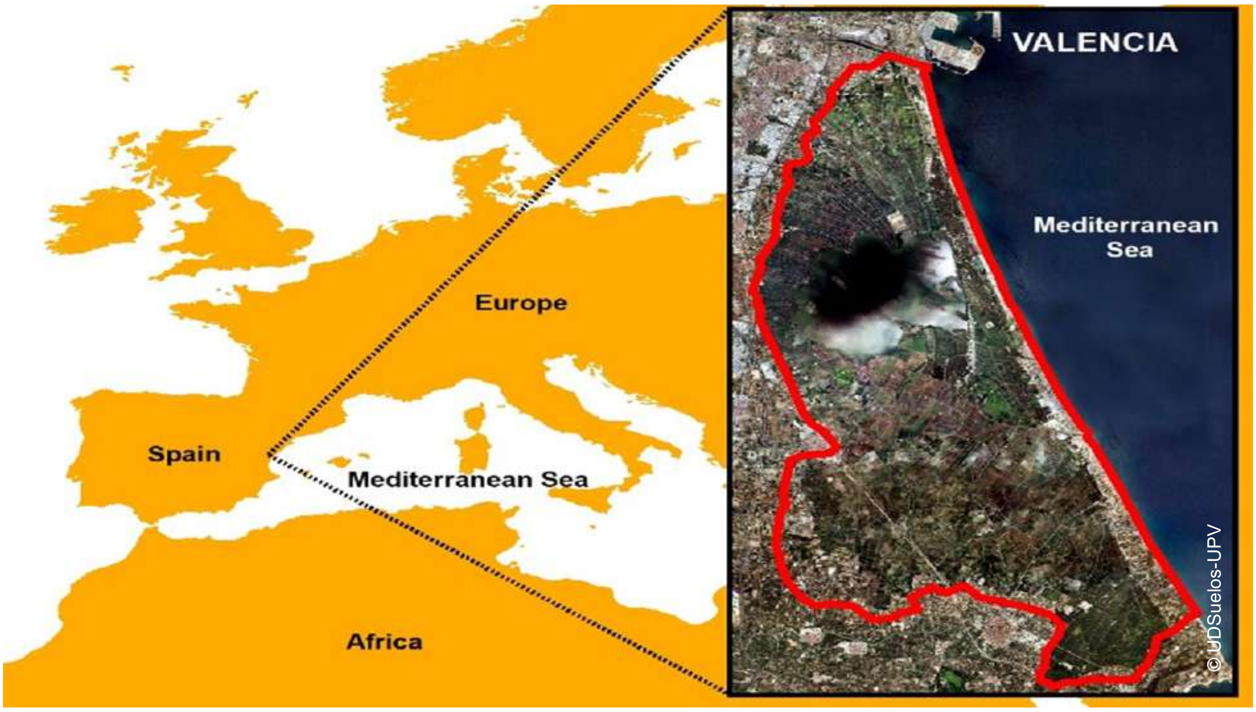


Photo 28. Location of Albufera of Valencia area with the boundaries of the Natural Park

Photo 29. Disolved organic matter in surface water owing to rice straw descomposition in paddy fields

References

Chivenge, P., Rubianes, F., Van Chin D., Va Thach, T., Tien, V., Romasanta, R.R., Van Hung, N. & Van Trinh, M. 2020. Rice straw incorporation influences nutrient cycling and soil organic matter. *In Sustainable Rice Straw Management*, pp.131-144. Springer.

Cui, J., Li, Z., Liu, Z., Ge, B., Fang, C., Zhou, C. & Tang, B. 2014. Physical and chemical stabilization of soil organic carbon along a 500-year cultivated soil chronosequence originating from estuarine wetlands: temporal patterns and land use effects. *Agr Ecosyst Environ.*, 196: 10-20. https://doi.org/10.1016/j.agee.2014.06.013

González-Prieto, S.J, Martín, A., Carballas, T. & Díaz-Raviña, M. 2018. *Guía de actuaciones en una zona quemada.* Andavira Ed. Santiago de Compostela. Pp 15

Jégou, A. & Sanchis-Ibor, C. 2019. The opaque lagoon. Water management and governance in l'Albufera de València wetland (Spain). *Limnetica*, 38(1): 503-515. https://doi.org/10.23818/limn.38.29

Jiang, Y., Qian, H., Huang, S., Zhang, X., Wang, L., Zhang, L., Shen, M., Xiao, X., Chen, F., Zhang, H., Lu, C., Li, C., Zhang, J., Deng, A., Groenigen, K.J. van & Zhang, W. 2019. Acclimation of methane emissions from rice paddy fields to straw addition. *Science Advances*, 5(1): eaau9038. https://doi.org/10.1126/sciadv.aau9038

Kögel-Knabner, I., Amelung, W., Cao, Z., Cao, Z., Fiedler, S., Frenzel, P., Jahn, R., Kalbitz, K., Kölbl, A. & Schloter, M. 2010. Biogeochemistry of paddy soils. *Geoderma*, 157: 1-14. https://doi.org/10.1016/j.geoderma.2010.03.009

Kongchum, M. 2005. *Effect of plant residue and water management practices on soil redox chemistry, methane emission and rice productivity*. Pp 1-189. Faculty of the Louisiana State University and Agricultural and Mechanical College.

Maris, S.C., Teira-Esmatges, M.R. & Català, M.M. 2016. Influence of irrigation frequency on greenhouse gases emission from a paddy soil. *Paddy and Water Environment*, 14(1): 199–210. https://doi.org/10.1007/s10333-015-0490-2

Moreno-Ramón, H., Marqués-Mateu, A., Ibáñez-Asensio, S. & Gisbert, J.M. 2015. Wetland soils under rice management and seawater intrusion: characterization and classification. *Spanish Journal of Soil Science*, 5(2): 111–129. https://doi.org/10.3232/SJSS.2015.V5.N2.02

Ribó, M., Albiach, R., Pomares, F. & Canet, R. 2017. *Alternativas de gestión de la paja de arroz en la Albufera de Valencia.* Nota técnica IVIA, (mayo), 1-9. (also available at: http://www.ivia.gva.es/documents/161862582/162455759/Nota+t%C3%A9cnica_Alternativas+de+gesti%C3%B3n+de+la+paja+de+arroz+en+la+Albufera+de+Valencia.pdf/cc127504-cf3c-4142-9345-d33e5c56c649)

Sanchis, E. 2015. *Emisiones de gases en el cultivo del arroz: efecto de la gestión de la paja*. TFM. Universitat Politècnica de València. Pp 1-76. (also available at: https://riunet.upv.es/bitstream/handle/10251/47780/01-Memoria.pdf?sequence=1&isAllowed=y)

Schmidt, A., Auge, H., Brandl, R., Heong, KL., Hotes, S., Settele, J., Villareal, S. & Schädler, M. 2015. Small-scale variability in the contribution of invertebrates to litter decomposition in tropical rice fields. *Basic and Applied Ecology*, 16: 674–680. https://doi.org/10.1016/j.baae.2015.01.006

Verdejo, J.V. 2016. *Efecto del manejo de la paja de arroz sobre los Gases de Efecto Invernadero (Albufera de Valencia).* TFC. Universitat Politècnica de València. Pp 1-133.

Zhou, G., Gao, S., Lu, Y., Liao, Y., Nie, J. & Cao, W. 2020. Co-incorporation of green manure and rice straw improves rice production, soil chemical, biochemical and microbiological properties in a typical paddy field in southern China. *Soil and Tillage Research*, 197: 104499. https://doi.org/10.1016/j.still.2019.104499

12. Conservation agriculture in intensive rice-based cropping systems in the Eastern Gangetic Plain

Md Khairul Alam[1,2], R.W. Bell[2], M.E. Haque[2], M.A. Kader[2,3]

[1]Soil Science Division, Bangladesh Agricultural Research Institute, Gazipur, Bangladesh

[2]Land Management Group, College of Science, Health, Engineering and Education, Murdoch University, Australia

[3]School of Agric. & Food Techn, Alafua Campus, The University of the South Pacific, the Independent State of Samoa

1. Related practices

Conservation agriculture, rice straw residue retention, strip tillage

2. Description of the case study

In this case study the results of long-term research on carbon (C) sequestration potential and life cycle analysis to estimate greenhouse gas emissions under conventional tillage (CT) and strip planting (SP) in combination with low residue (straw) retention and increased residue retention in rice-based cropping systems of the Eastern Gangetic Plain are presented.

In the Eastern Gangetic Plains (EGP), farmers are beginning to follow Conservation Agriculture (CA) practices to grow dry season crops but for rice they follow the traditional soil puddling for rice establishment. Accordingly, the benefits accrued by following zero tillage or strip planting are lost by the puddling of wet soil. The non-puddled transplanting of rice, a novel practice of rice crop establishment, involves minimal soil disturbance so that CA can be practiced in rice-based cropping systems. For rice, first the soil is strip-tilled, for example with a two-wheeled tractor operated Versatile Multi-crop Planter (VMP) (Haque *et al.*, 2016), then the land is soaked overnight, and finally rice seedlings were transplanted within the strip.

Verification of the benefits of the CA system in the EGP (two locations of northwestern Bangladesh) involved long term studies on carbon (C) sequestration potential and life cycle analysis to estimate greenhouse gas (LCA GHG) emissions under conventional tillage (CT) and strip planting (SP) in combination with low residue (straw) retention (LR, business-as-usual/current practice) and increased residue retention (HR) in Calcareous Brown

Floodplain soil (Alipur, Durgapur, Rajshahi) and Grey Terrace soil (Digram, Godagari, Rajshahi). The cropping systems studied were mustard-irrigated rice-monsoon at Alipur and wheat-jute-monsoon rice at Digram site. The novel establishment practices increased total organic carbon (TOC) in 0-30 cm of soil by 26 percent (19.4 t/ha to 24.5 t/ha). That is, after 5 years, practicing SP for upland/dry season crops and non-puddling for rice accumulated an additional 5.1 t TOC/ha. The rice transplanted after non-puddling of soil with residues retained at a minimal rate decreased CO_2eq emissions by 31 percent for the actual LCA GHGs emissions and by 20 percent for the total LCA GHGs emissions in comparison with conventional puddled transplanting. By contrast, non-puddling with increased residue retention reduced the actual LCA GHG emissions by 16 percent in comparison with the current conventional practices. The accurate estimation of improvement in actual LCA GHG emissions depended on the long-term assessment of increases in soil TOC.

3. Context of the case study

The EGP covers about 34.6 million ha of mostly low-lying alluvial plains in Eastern India, Nepal, and Bangladesh. Most of the agricultural land grows a monsoon season crop of wetland, rainfed rice. This is followed in the dry season by mostly irrigated fields crops such as wheat, maize, mustard and irrigated rice (Boro). Then in the early wet season, another crop such as jute may be grown with irrigation to supplement rainfall. Presently in the EGP, the cropping intensity is around 200 percent. However, the intensification occurs at the cost of soil health. The present case study is conducted in the Subtropical climate of the EGP. The climate is favorable for quick decomposition of soil organic matter (SOM) which is a major impediment for SOM enrichment (Ross, 1993). On the top of that, farmers employ excess tillage for dry season/upland crop establishment and puddle the soil for rice crop establishment that accelerates SOM degradation (Alam *et al.*, 2018; Alam, Biswas and Bell, 2016). Over time, the breakdown of soil structure caused by puddling of soils creates a plough layer that hardens when dry at 10-20 cm soil depth leading to shallow rooting depth, waterlogging risk and soil crack development in the dry season. Preparation of the rice-puddled lands for upland/dry season crops requires applying extra tillage, excess labor and fuel that incur farm income losses (Sharma, Datta and Redulla, 1988; Sharma, Tripathi and Singh, 2005). Soil degradation and yield stagnation have been reported in many studies conducted in the Indo-Gangetic Plains (Alam *et al.*, 2017; Kukal and Aggarwal, 2003). The puddled soil for rice in a rice-based system has downsides in terms of methane (CH_4) emissions, while upland crops under intensive tillage emits increased N_2O and CO_2 (Pathak *et al.*, 2011). However, Xu *et al.* (2013) saw the potential for carbon sequestration in paddy soils with improved management, while Lal (2004), Wang, Li and Alva (2010), Alam, Biswas and Bell, (2016) and Das *et al.* (2013) proposed that sequestration of SOC as an effective option for restoration of degraded soil structure, enhancement of soil fertility and emission mitigation from rice soils. The case study relates to the Sub-Tropical/Tropical moist zone (according to the IPCC climate zones on the active Ganges floodplain and Meghna estuarine floodplain) on the following soils: Calcareous Brown Floodplain (Aeric Eutrochrept) and Grey Terrace soils (Aeric Albaquepts) (USDA-SCS, 1975).

4. Possibility of scaling up

The adoption of CA is being replicated in different parts of the EGP including in Bangladesh. Based on the larger number of on-farm trials over a three-year period in four districts of Bangladesh, Haque and Bell (2019) reported that strip-based non-puddled transplanting of rice provided similar or greater yield, reduced cultivation cost and increased gross margin by 69 percent for Boro rice and 67 percent for Aman rice. In the early demonstrated areas, some farmers have adopted non-puddling rice followed by strip planting of dry season crops while retaining moderate level of crop residue. Due to farmers' mindset and preference for soil puddling followed by transplanting of rice seedling, adoption of CA in rice-based systems is slow. The concept of non-puddled rice followed by strip planted dry season crops is relatively new and most farmers, researchers, extension workers and policy planners are not aware on the technology. On the other hand, research and refinement of this technology is still limited while only limited out-scaling actions were undertaken. Intensive research on CA for smallholder farmers in the EGP that reported by Bell *et al.* (2019). Based on that evidence, it can be confidently asserted that the scaling out of CA (non-puddled rice followed by strip planted dry season crop with moderate amount residue retention) in most of the rice-based areas in EGP is possible with the engagement of private sector and farmers. Policy level action for extension, mass propagation, education and advanced research are essential for out scaling of this technology.

5. Impact on soil organic carbon stocks

Strip planting for dry season crops and strip-tilled non-puddled rice transplanting in combination with increased residue retention sequestered more C in soils under rice-intensive cropping systems than the current farmers' practices (intensive dry tillage for upland dry season crops and wet tillage or puddling for rice in combination with residue removal). The accumulated C storage was 10.8 t/ha at Alipur which was 4.26 t/ha higher than business as usual (current practice) in the 0–10 cm soil layer. Similarly, after 13 crops at Digram, the storage was 10.2 t/ha in 0-10 cm depth which was 3.79 t/ha higher than the current farmers' practice. The increased SOC accumulation rate corresponds to 1.2 and 1.1 tC/ha per year at Alipur and Digram, respectively, more than the current practice. In addition to C stock increase in the 0-10 cm soil depth, the C stock increase under the CA-based novel crop establishment practice in the 10-30 cm soil depth was around 0.7 t/ha (i.e. ~0.2 t/ha/yr) more than the current practice after 14 crops at both the sites (Table 49). The adoption of the non-puddled transplanting of rice in the entire EGP could increase the C storage by 131-145 million tons CO_2eq of C in the soils of the rice-based cropping systems.

Table 49. Evolution of SOC stocks after the 5-year experiments

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha)	Duration (Years)	Depth (cm)	Reference
Alipur, Bangladesh	Sub-Tropical/ Tropical Moist[ǂ]	Calcareous Brown Floodplain (Calcareous Alluvium)	6.5	5.1	5	0-30	Alam *et al.* (2018, 2020a)
Digram, Bangladesh		Grey Terrace Soils (High Barind Tract)	6.4	4.6			

[ǂ]IPCC climate zones

6. Other benefits of the practice

6.1. Benefits for soil properties

Besides SOC accumulation, the novel practices had other benefits in terms of soil physical, chemical and microbial properties. Soil bulk density (BD) at both the sites was reduced significantly relative to current farmers' practice. The lowest BD at both Digram and Alipur was recorded in the strip planted rice soil which was 0.12 g/cm^3 lower than the current farmer's practice. In case of soil porosity, residue retention combined with the CA practice increased porosity values by 4.3-4.6 percent relative to the current practices.

The pHs of soils were slightly higher at both the sites (Alipur and Digram) than the current practice. The pHs of the soils was around 6.4 under current farmer's practice which after 14 crops at Alipur was increased to 6.8 and at Digram increased to 6.7 under the novel CA practice.

After 13-14 crops, minimal soil disturbance combined with residue retention recorded the highest increase in total nitrogen (N) in soils of both the case-study sites. The CA soils recorded 0.33 g N/kg at Alipur and 0.27 gN/kg at Digram more total N than the conventionally managed soil after growing 13-14 crops.

Similarly, response of microbial activities (microbial biomass carbon- MBC) were positive to strip tilled non-puddled rice practice. At Alipur and Digram, the non-puddled rice soils recorded higher amounts of MBC than the business as usual (farmers') practice. The non-puddled practice had 43-49 mg/kg higher MBC than the current farmers' practice.

Previous research on the same sites recorded increased water holding capacity and plant available water content in soils but reduced soil penetration resistance under the CA practice (Islam, 2016).

6.2 Minimization of threats to soil functions

Table 50. Soil threats

Soil threats	
Nutrient imbalance and cycles	The strip tilled non-puddled transplanting of rice and strip planting of dry season crops sequesters C and N in soils by slowing their in-season turnover, decreasing gaseous emissions and by improving the synchrony between crop N demand and N availability in soils (Alam *et al.*, 2018, 2020a).
Soil acidification	The current practice in the case-study areas intensifies acidification of soils in the EGP (Bangladesh) (BARC, 2018). After 5 years, the non-puddled rice practice followed by strip planting with increased residue retention showed increase in pH by 0.4, relative to current farmer's practice (Alam *et al.*, 2018, 2020a), indicating that, over time, the practice has potential to reverse the acidification process.
Soil biodiversity loss	The modified crop establishment practices in line with CA principles increased MBC which might be due to diversified microbes in soil (Alam *et al.*, 2020b). Similarly, Rathore *et al.* (2017) related the increased microbial activity with the diversified microbiome and increased abundance in soil under the minimal soil disturbance with increased residue retention.
Soil compaction	The strip tilled non-puddled transplanting of rice and strip planting of dry season crops reduces soil compaction in the surface soil. Studies at Alipur and Digram, Rajshahi, Bangladesh (Islam, 2016) and at Baliakandi, Rajbari, Bangladesh (Salahin, 2017) showed reduced soil compaction due to practicing CA.

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

The higher grain yields and higher net return for crops in rice-based cropping systems were recorded in strip planting and strip tilled non-puddled rice transplanting practice (in combination with residue retention) compared to the practices farmers currently use (Bell *et al.*, 2019; Islam, 2016; Salahin, 2017) In the first two years, there was no significant variation in rain-fed rice yield using non-puddled transplanting relative practices farmers currently use. Thereafter, significantly higher grain yield of rain-fed rice in non-puddled transplanting was recorded following strip planting with minimal soil disturbance.

6.4 Mitigation of and adaptation to climate change

Strip planting for dry season crops and non-puddled transplanting of rice in combination with increased residue retention after five years decreased greenhouse gas emissions. The on-farm emissions results indicated that the CA practice reduced crop production emissions by around 22 percent relative to current farmers' practice. The strip tilled non-puddled rice transplanting in combination with increased residue retention saved 9.1 and 74.8 kg CH_4/ha/season emission relative to current farmers' practice. Similarly, Alam, Bell and Biswas (2019a), Alam, Biswas and Bell (2016) and Pathak *et al.* (2013) stated that CA can reduce net GHG emission through increased SOC sequestration.

The CA-based crop establishment practice is one of the climate-smart agricultural practices (Rahaman, Rahman and Hossain, 2018). Based on the year 2013-2014, Bell *et al.* (2019) showed that with 2.5 percent adoption of CA and mechanized planting in Bangladesh (which is the current CA adoption level in Asia), the estimated potential gain stands to about USD 20 million year^{-1}. Moreover, it was reported in 2017 that the non-puddling (NP) of rice with and without mechanization was gaining momentum in the EGP area which would help them adapt well with the changing climate.

6.5 Socio-economic benefits

Haque and Bell (2019) showed up to 59 percent greater profit from non-puddled rice on farmers' fields in the EGP. From 50 to 94 percent farmers adopting NP of rice in both the irrigated (Boro) and monsoon (T. aman) seasons reported greater net returns even when there were economic losses due to low rice grain prices. Islam, Hossain and Saleque (2014) recorded the lowest benefit cost ratio (1.42) in the puddled transplanting which they attributed to lower yield at the cost of increased inputs (fuel and machinery pass) and labor requirement for land preparation and seedling transplanting. Salahin (2017) also employed non-puddled transplanting of rice and strip planting for dry season crops and recorded 1.29 benefit: cost ratio for zero tillage under non-puddled condition, 1.51 benefit:cost ratio for strip planting under non-puddled condition and 1.31 for conventional puddling. With these economic profits, farmers adopting the practice will have better socio-economic conditions than farmers with their traditional practices.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 51. Soil threats

Soil threats	
Soil erosion	In the EGP, soil erosion is not a significant threat.
Nutrient imbalance and cycles	Nutrient stratification concentrates P and K and SOM closer to the soil surface in CA. Less straw for animal feed and as household fuel when crop residues retained in fields.

Soil threats	
Soil acidification	Soil acidification appears to reverse in long term studies (Alam *et al.*, 2018, 2020a).
Soil compaction	Repeated dry tillage and wet tillage in rotation increase soil compaction (Parihar *et al.*, 2016). On the other hand, Islam (2016) reported reduced penetration resistance and thereby less soil compaction in CA-managed soils. Increased residue retention in combination with non-puddled rice followed by strip planting decreased penetration resistance by 14–17 percent at 5–15 cm depth and by 16–18 percent at 0–10 cm depth, compared to puddled rice followed by dry tillage with low residue retention. The SOC sequestration and reduced tractor passes may reduce the soil compaction under the practice (Bogunovic *et al.*, 2017). Non-puddled soil is initially firmer and depending on the soil type may require more force for transplanting rice seedling roots.

7.2 Increases in greenhouse gas emissions

For all crops in the mustard-irrigated rice-monsoon rice cropping system, strip planting for dry season crops and non-puddled transplanting of rice with either low or increased residue retention were the best actual life cycle GHG mitigation options. The novel CA-based establishment practice decreased CH_4 emission by from 4.3 to 10.8 kg per ton of rice equivalent yield (REY) relative to farmers' practice, while the same practice also decreased N_2O emission by 9 kg CO_2eq to 15 kg CO_2eq emission per ton of REY. With the accumulation of soil TOC in CA cropping (3.8 - 4.2 tons CO_2eq/ha) after five years, the life cycle GHG savings with the best mitigation practice for 1 ton of rice-equivalent yield were 46 percent relative to current farmer's practice in the EGP. After accounting for sequestered TOC, the non-puddled rice followed by strip planting reduced net CO_2eq life cycle GHG emission by 0.25 ton for each ton of rice equivalent production (mustard-irrigated rice-monsoon rice system) relative to farmers' current practice (Alam, Bell and Biswas, 2019a).

7.3 Negative impact on production

The case study did not observe yield reduction of rice either in the short term or longer term (Bell *et al.*, 2019). At both the sites and in other trials, the yields of non-rice crops of the rice-based cropping systems were the same or higher in the CA practice compared to conventional practice.

8. Recommendations before implementing the practice

The following recommendations should be followed to get good results in this practice:

1. If strip tillage non-puddling cannot be provided due to excessive wetness of the soil, zero tillage non-puddling method can be applied.
2. Farmers should confirm the availability of the machines or attachments to machines (two-wheel tractor with strip tillage attachment) before implementing the practice.
3. If hands/fingers experience fatigue from repeated manual transplanting of rice seedlings due to firmness of soils, soak the soil for one or two more days.
4. Even if the yield is equal or slightly less in the first year relative to their current practice, farmers should not be discouraged as it may be due to inexperience with the techniques.
5. To get better yield result by establishing rice seedlings in this technique in lowlands, it is recommended to have good drainage system.
6. Fertilizer application should be localized (along the strip) as much as possible.
7. Pre- and post-planting weed management should be satisfactory. Switching between herbicides with different modes of action and hand weeding once in a year could be advised.

9. Potential barriers for adoption

Table 52. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	- Delay in timely transplanting due to unavailability of machinery (Johansen *et al.*, 2012). - Drudgery during rice seedling transplanting if the non-puddled soils are hard to penetrate. - Plants/seedlings suffer stresses for establishment if the soils are too firm and resist root proliferation - Non-availability of affordable rice planters.
Cultural	No	There are no cultural barriers for adopting the practice.
Social	No	Farmers who use non-puddled transplanting may be discouraged by neighbors and relatives as the practice is unconventional and by drudgery if the non-puddled soils require more effort to place seedling roots in soils. Mindset is the biggest barrier to adopt this technology.
Economic	No	There are no economic barriers for adopting the practice. Indeed, there is generally increased profitability of non-puddled transplanting and CA. However, the cost of machinery for strip tillage is too high for individual farmers, so they have to rely on service providers. However, the hire cost of the strip tillage is lower than for conventional land preparation and puddling.

Barrier	YES/NO	
Institutional	Yes	The government extension services still favor full tillage, and subsidies for machinery support full tillage machinery rather than those for minimum tillage. Finance institutions are unfamiliar with minimum soil disturbance and hence do not have loan packages to support adopting CA practice.
Legal (Right to soil)	No	There are no legal barriers for adopting the practice.
Knowledge	Yes	Knowledge gap of farmers, researchers, extension workers and policy makers.
Natural resource	No	Increased infiltration in light textured, upland soils may lead to requirement of additional irrigation for rapidly-draining soils and/or yield reduction of irrigated rice (Boro).
Other	Yes	- Delay in transplanting, prolonged inundation and seedling mortality may happen due to lack of drainage in lowland rice. - Early monsoon rain may not allow CA equipment (e.g. zero tillage drill) to enter the field - Altered efficacy of herbicides for control of weeds.

Photos

Photo 30. Two-wheel tractor with Versatile Multi-crop Planter (VMP), sowing through standing residue retained at higher rate at Digram, Rajshahi (top) and low rate (bottom) at Alipur, Durgapur, Rajshahi, Bangladesh

Photo 31. Germination of strip planted mustard and non-puddled rice seedling transplanting at Alipur, Durgapur, Rajshahi, Bangladesh

Photo 32. Performance of non-puddled transplanted rice at Alipur, Durgapur, Rajshahi, Bangladesh

References

Alam, Md.K., Bell, R.W., Haque, M.E., Islam, M.A. & Kader, M.A. 2020a. Soil nitrogen storage and availability to crops are increased by conservation agriculture practices in rice–based cropping systems in the Eastern Gangetic Plains. *Field Crops Research*, 250: 107764. https://doi.org/10.1016/j.fcr.2020.107764

Alam, Md.K., Bell, R.W., Haque, M.E. & Kader, M.A. 2018. Minimal soil disturbance and increased residue retention increase soil carbon in rice-based cropping systems on the Eastern Gangetic Plain. *Soil and Tillage Research*, 183: 28–41. https://doi.org/10.1016/j.still.2018.05.009

Alam, Md.K., Biswas, W.K. & Bell, R.W. 2016. Greenhouse gas implications of novel and conventional rice production technologies in the Eastern-Gangetic plains. *Journal of Cleaner Production*, 112: 3977–3987. https://doi.org/10.1016/j.jclepro.2015.09.071

Alam, M.K., Bell, R.W. & Biswas, W.K. 2019a. Decreasing the carbon footprint of an intensive rice-based cropping system using conservation agriculture on the Eastern Gangetic Plains. *Journal of Cleaner Production*, 218: 259–272. https://doi.org/10.1016/j.jclepro.2019.01.328

Alam, M.K., Bell, R.W. & Biswas, W.K. 2019b. Increases in soil sequestered carbon under conservation agriculture cropping decrease the estimated greenhouse gas emissions of wetland rice using life cycle assessment. *Journal of Cleaner Production*, 224: 72–87. https://doi.org/10.1016/j.jclepro.2019.03.215

Alam, M.K., Bell, R.W., Hasanuzzaman, M., Salahin, N., Rashid, M.H., Akter, N., Akhter, S., Islam, M.S., Islam, S., Naznin, S., Anik, M.F.A., Apu, M.M.R.B., Saif, H.B., Alam, M.J. & Khatun, M.F. 2020b. Rice (Oryza sativa L.) Establishment techniques and their Implications for soil properties, global warming potential mitigation and crop yields. *Agronomy*, 10(6): 888. https://doi.org/10.3390/agronomy10060888

Alam, M.K., Salahin, N., Islam, S., Begum, R.A., Hasanuzzaman, M., Islam, M.S. & Rahman, M.M. 2017. Patterns of change in soil organic matter, physical properties and crop productivity under tillage practices and cropping systems in Bangladesh. *The Journal of Agricultural Science*, 155(2): 216–238. https://doi.org/10.1017/S0021859616000265

BARC. 2018. Fertilizer Recommendation Guide 2018, Bangladesh Agricultural Research Council, Khamarbari, Dhaka, Bangladesh pp. 1-230. (also available at https://msibsri4313.files.wordpress.com/2013/10/frg_2005.pdf).

Bell, R.W., Haque, M.E., Jahiruddin, M., Rahman, M.M., Begum, M., Miah, M.A.M., Islam, M.A., Hossen, M.A., Salahin, N., Zahan, T., Hossain, M.M., Alam, M.K. & Mahmud, M.N.H. 2019. Conservation Agriculture for Rice-Based Intensive Cropping by Smallholders in the Eastern Gangetic Plain. *Agriculture*, 9(1): 5. https://doi.org/10.3390/agriculture9010005

Bogunovic, I., Bilandzija, D., Andabaka, Z., Stupic, D., Rodrigo Comino, J., Cacic, M., Brezinscak, L., Maletic, E. & Pereira, P. 2017. Soil compaction under different management practices in a Croatian vineyard. *Arabian Journal of Geosciences*, 10(15): 340. https://doi.org/10.1007/s12517-017-3105-y

Das, T.K., Bhattacharyya, R., Sharma, A.R., Das, S., Saad, A.A. & Pathak, H. 2013. Impacts of conservation agriculture on total soil organic carbon retention potential under an irrigated agro-ecosystem of the western Indo-Gangetic Plains. *European Journal of Agronomy*, 51: 34–42. https://doi.org/10.1016/j.eja.2013.07.003

Haque, M.E. & Bell, R.W. 2019. Partially mechanized non-puddled rice establishment: on-farm performance and farmers' perceptions. *Plant Production Science*, 22(1): 23–45. https://doi.org/10.1080/1343943X.2018.1564335

Haque, M.E., Bell, R.W., Islam, M.A. & Rahman, M.A. 2016. Minimum tillage unpuddled transplanting: An alternative crop establishment strategy for rice in conservation agriculture cropping systems. *Field Crops Research*, 185: 31–39. https://doi.org/10.1016/j.fcr.2015.10.018

Islam, A.K.M.S., Hossain, M.M. & Saleque, M.A. 2014. Effect of Unpuddled Transplanting on the Growth and Yield of Dry Season Rice (*Oryza sativa* L.) in High Barind Tract. *The Agriculturists*, 12(2): 91–97. https://doi.org/10.3329/agric.v12i2.21736

Islam, M.A. 2016. *Conservation Agriculture: Its effects on crop and soil in rice-based cropping systems in Bangladesh*. Murdoch University. (phd). (also available at https://researchrepository.murdoch.edu.au/id/eprint/36706/).

Johansen, C., Haque, M.E., Bell, R.W., Thierfelder, C. & Esdaile, R.J. 2012. Conservation agriculture for small holder rainfed farming: Opportunities and constraints of new mechanized seeding systems. *Field Crops Research*, 132: 18–32. https://doi.org/10.1016/j.fcr.2011.11.026

Kukal, S.S. & Aggarwal, G.C. 2003. Puddling depth and intensity effects in rice–wheat system on a sandy loam soil: I. Development of subsurface compaction. *Soil and Tillage Research*, 72(1): 1–8. https://doi.org/10.1016/S0167-1987(03)00093-X

Lal, R. 2004. Soil carbon sequestration in India. *Climate Change*, 65: 277–296.

Parihar, C.M., Jat, S.L., Singh, A.K., Kumar, B., Yadvinder-Singh, Pradhan, S., Pooniya, V., Dhauja, A., Chaudhary, V., Jat, M.L., Jat, R.K. & Yadav, O.P. 2016. Conservation agriculture in irrigated intensive maize-based systems of north-western India: Effects on crop yields, water productivity and economic profitability. *Field Crops Research*, 193: 104–116. https://doi.org/10.1016/j.fcr.2016.03.013

Pathak, H., Saharawat, Y.S., Gathala, M. & Ladha, J.K. 2011. Impact of resource-conserving technologies on productivity and greenhouse gas emissions in the rice-wheat system. *Greenhouse Gases: Science and Technology*: 1–17. https://doi.org/10.1002/ghg.27

Pathak, H., Sankhyan, S., Dubey, D.S., Bhatia, A. & Jain, N. 2013. Dry direct-seeding of rice for mitigating greenhouse gas emission: field experimentation and simulation. *Paddy and Water Environment*, 11(1–4): 593–601. https://doi.org/10.1007/s10333-012-0352-0

Rahaman, M.A., Rahman, M.M. & Hossain, Md.S. 2018. Climate-resilient agricultural practices in different Agro-ecological Zones of Bangladesh. *In* W. Leal Filho, ed. *Handbook of Climate Change Resilience*, pp. 1–27. Cham, Springer International Publishing. (also available at https://doi.org/10.1007/978-3-319-71025-9_42-1).

Rathore, R., Dowling, D.N., Forristal, P.D., Spink, J., Cotter, P.D., Bulgarelli, D. & Germaine, K.J. 2017. Crop establishment practices are a driver of the plant microbiota in winter oilseed rape (Brassica napus). *Frontiers in Microbiology*, 8: 1489. https://doi.org/10.3389/fmicb.2017.01489

Ross, S.M. 1993. Organic matter in tropical soils: current conditions, concerns and prospects for conservation. *Progress in Physical Geography: Earth and Environment*, 17(3): 265–305. https://doi.org/10.1177/030913339301700301

Salahin, N. 2017. *Influence of minimum tillage and crop residue retention on soil organic matter, nutrient content and crop productivity in the rice-jute system*. Ph.D. Thesis

Sharma, P., Tripathi, R.P. & Singh, S. 2005. Tillage effects on soil physical properties and performance of rice–wheat-cropping system under shallow water table conditions of Tarai, Northern India. *European Journal of Agronomy*, 23(4): 327–335. https://doi.org/10.1016/j.eja.2005.01.003

Sharma, P.K., Datta, S.K.D. & Redulla, C.A. 1988. Tillage effects on soil physical properties and wetland rice yield. *Agronomy Journal*, 80(1): 34–39. https://doi.org/10.2134/agronj1988.00021962008000010008x

USDA-SCS. 1975. Soil taxonomy : a basic system of soil classification for making and interpreting soil surveys. [Washington] : U.S. Dept. of Agriculture, Soil Conservation Service. [Cited 19 June 2020]. https://trove.nla.gov.au/version/9911712

Wang, Q., Li, Y. & Alva, A. 2010. Cropping systems to improve carbon sequestration for mitigation of climate change. *Journal of Environmental Protection*, 1(3): 207–215. https://doi.org/10.4236/jep.2010.13025

Xu, S.-Q., Zhang, M.-Y., Zhang, H.-L., Chen, F., Yang, G.-L. & Xiao, X.-P. 2013. Soil organic carbon stocks as affected by tillage systems in a double-cropped rice field. *Pedosphere*, 23(5): 696–704. https://doi.org/10.1016/S1002-0160(13)60062-4

13. Long term fertilization in a subtropical floodplain soil in Bangladesh

Mohammed A. Kader[1,2,3], Mohammed J. A. Mian[2], Mohammed M.R. Jahangir[2]

[1]School of Agric. and Food Techn, Alafua Campus, The University of the South Pacific, the Independent State of Samoa

[2]Soil science Department, Bangladesh Agricultural University, Mymensingh, Bangladesh

[3]Land Management Group, College of Science, Health, Engineering and Education, Murdoch University, Australia

1. Related practices

Crop rotations, Chemical fertilization, Cattle manure application, Rice paddy management

2. Description of the case study

A long-term field experiment was established on a young floodplain soil at Bangladesh Agricultural University (BAU) in the Field Lab of the Department of Soil Science, Mymensingh in 1978 with an annual rice (*Oryza spp.*)-rice-pulse crop rotation to evaluate the influence of different fertilizer application on soil fertility and crop productivity. It involves different mineral fertilizer treatments including one with farmyard manure (FYM) and control (control, 100%N, 100%NP, 100%NPK, 100%NPKSzn and 50% N +FYM). Crop rotation and fertilizer rate were adjusted over time based on the current practices and crop demand (Mostofa *et al.*, 2015; Islam *et al.*, 2019a, 2019b). The rate of fertilizer application was updated based on crop requirement in 1982 and 2013 and initial FYM was changed to 50% N+FYM in 1982 and later further changed to 100% NPK + FYM in 2013 (Table 53).

Three crops were cultivated each year during 1978-1982. Rice was grown in two seasons: Transplant Aus (local) (March-June) and T. Aman (July-November), with a leguminous pulse (grass pea sown in the dry winter season: December- February). No fertilizer was applied for grass pea as it is grown as a relay crop with Aman and sown on standing Aman crop before harvest.

With the availability of irrigation facilities and high yielding varieties (HYV), rice was cultivated in two growing seasons from 1983 onwards following a cropping pattern of Boro rice (irrigated winter rice transplanted on mid-January and harvested mid-May)-Fallow-T. Aman (Kader *et al.*, 2017; Begum *et al.*, 2018a). Leguminous pulse was omitted from the yearly cropping pattern to accommodate irrigated Boro rice, as both are cultivated in the same time of the year.

Current application rates of N, P, K, S, and Zn are 180 (120+60), 18 (14+4), 87 (58+29), 14 (8+6) and 1 (1+0) kg/ha/yr, respectively applied as urea, triple super phosphate, potassium chloride, gypsum, and zinc oxide (Mostofa *et al.*, 2015). Cow dung was mixed with rice straw applied once a year 10-15 days prior transplantation of Boro rice at a rate of 5 t/ha fresh material (1-1.5%N, 0.3-0.4%P and 1-1.5%K). The experiment was conducted in a randomized block design with three replications (12m × 6m) (Photo 33).

Table 53. Fertilizer dose and cropping pattern at BAU long-term field experiment and its modification over time

Nutrient element	Dose (kg/ha)						
	1978-1982			1983-2012 (T. Aman)		2013 (Boro)-till date	
	T. Aus	T. Aman	Grass pea	Boro rice	T. Aman	Boro rice	T. Aman
N	60	60	-	90	80	120	60
p	20	20	-	20	20	14	4
K	15	15	-	19	19	58	29
S	-	-	-	30	30	8	6
Zn	-	-	-	5	5	1	-
FYM	15 000	15 000	-	5 000	-	5 000	-

3. Context of the case study

Floodplain is the dominant soil physiography of Bangladesh and represents 80 percent of soil area (9.7 million ha) (Brammer, 1996). The study area has sub-tropical humid climate and is characterized by hot and humid summers and cool winters with an annual mean temperature of 25.8 °C and rainfall of 2 427 mm, 80 percent of which falls between May to September (Begum and Kader, 2018). Floodplain soils are very fertile and intensively cultivated area in Bangladesh and have a cropping intensity of more than 200 percent (Uddin *et al.*, 2019). However, at the beginning of green revolution in Bangladesh during 1970s, farmers did not apply fertilizers or manure to supplement the nutrients taken up by crops, if applied very sporadically and only little N fertilizer. Considering this situation, agricultural scientists were afraid that this intensively cultivated soil would be degraded soon and would be showing different nutrient deficiency symptoms. Thus, a long-term experiment was established in 1978 at Bangladesh Agricultural University (BAU) farm at Mymensingh (24°43' N, 90°25' E), Bangladesh on a loamy, mixed, non-acidic Aeric Haplaquept to demonstrate the effect of balanced fertilization on crop productivity and soil fertility to farmers and policy makers (Mian *et al.*, 1991; Egashira *et al.*, 2005; Kader *et al.*, 2017).

4. Possibility of scaling up

Findings of this experiment have been using for formulation of national fertilizer guide particularly for floodplain soils of Bangladesh. In addition, good fertilizer practices adapted here in this long-term field experiment for maintaining better crop productivity and soil fertility are promoted throughout the region.

5. Impact on soil organic carbon stocks

This 42-year long-term experiment has been running on a loamy, mixed, non-acidic Aeric Haplaquept under tropical wet climate in Bangladesh. Soil organic carbon (SOC) has mostly doubled from initial 7.3 g/kg soil to 14.9-17.0 g/kg depending on treatments including control. However, more SOC build up was observed in NPK and NPK + FYM treatments compared to the others. As a result, higher crop yields were also recorded in these two treatments. The highest accumulation rate of SOC was calculated around 0.51 t/ha/yr in NPK treatment (

Table **54**). SOC build up in NPK + FYM treatment was lower than the NPK treatment because only 50 percent N and no P and K fertilizer was applied in this treatment until 2013. It was also observed that the accumulated SOC in NPK treatment was more stable and less prone to de-composition if present crop management has been changed. Balanced fertilization, particularly K might contributed to the formation of stable organic carbon in plant root system that latter added to soil. This high accumulation of SOC even in control treatment might be related with practicing yearly double rice cropping pattern that kept the field wet around nine months of a year. This anaerobic environment favors SOC accumulation by producing large aquatic biomass as well as reducing the decomposition rate. In addition, obtaining a substantial yield (2-3 t/ha) continuously in control treatment indicates that there are other N sources supporting plant growth other than the inputs through fertilization and atmospheric deposition, and mineralization of residues and SOM. This N could be supplied by microbial N fixation, blue green algae (BGA) or other aquatic biomass (*Azolla*, weeds) that grow in the standing water with the main crop and during the fallow period (Begum *et al.*, 2018a).

Table 54. Carbon storage potential of yearly double rice cropped soil under different long-term nutrient management

Treatments	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	More information
Control	13.4 (0.73%)	0.44	Aquatic biomass helps SOC accumulation
N (100% N)		0.43	-
NP (100% N & P)		0.44	-

Treatments	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	More information
NPK (100% N, P & K)		0.51	Under NPK management SOC increased to 1.61%
NPKSZn (100% N, P, K, S & Zn)		0.44	S and Zn fertilizer was applied later since 1982
NPK+FYM (100% N, P & K+ FYM @ 5 t/ha/yr)		0.46	100% P & K and 50% additional N was applied later since 2013

6. Other benefits of the practice

6.1. Benefits for soil properties

Physical properties

Over time the plough pan become thicker due to continuous puddling for cultivation of two wetland rice per year which reduces the percolation loss of irrigation water. In addition, bulk density of the surface soil declined due to accumulation of SOC.

Chemical properties

Soil pH and available phosphorous content remain mostly stable over time. However, a sharp decline of available K (mostly half) was observed in all the treatments. The depletion of available K was remarkable particularly in treatments without K fertilizer application. Soil total N content was also doubled like SOC that varied from 1.60 g N/kg (control) to 1.78 g N/kg (application of NPK) (Kader *et al.*, 2017, Islam *et al.*, 2019a). Cation exchange capacity (CEC) also increased mostly in balanced fertilized treatments (NPK and/or NPK+ FYM) due to accumulation of more SOC.

Biological properties

Better soil respiration and enzyme activities were observed in balanced fertilized treatments (Islam *et al.*, 2019b).

6.2 Minimization of threats to soil functions

Table 55. Soil threats

Soil threats	
Nutrient imbalance and cycles	Available P remained stable over time while there is a significant increase of soil total N in balanced fertilizer treatments. However, available K was declined sharply in all treatments though it was slightly low where K fertilizer was applied.
Soil salinization and alkalinization	No salinity and alkalinity were developed in any treatment.
Soil acidification	Soil pH remains stable in all the treatments.
Soil sealing	No soil sealing was observed.
Soil compaction	Density of surface soil decreases due to SOC accumulation.

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

Crop yields increased by 30-40 percent in balanced fertilized treatments over the time compared to other treatments (Mostofa *et al.*, 2015).

6.4 Mitigation of and adaptation to climate change

Due to sequestration of more C in balanced fertilized treatments over time will indirectly reduce GHG (Begum *et al.*, 2018a, 2018b).

6.5 Socio-economic benefits

As the balanced fertilized plots provide 30-40 percent higher yield and maintain soil fertility better, thus it is economically much profitable.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 56. Soil threats

Soil threats	
Soil contamination / pollution	There could be some soil contamination. However, it was not studied yet.

7.2 Increases in greenhouse gas emissions

Overall net GHG mitigation 2 t CO_2-eq/ha/yr (equivalent to 0.6 tCeq/ha/yr) achieved when estimated with a 100-year global warming potential for CH_4, N_2O and CO_2 in NPK treatment under double rice cropping system (Begum *et al.*, 2018a; 2018b).

8. Recommendations before implementing the practice

This long-term field experimentation shows that balanced fertilizer application with NPK and/or NPK+ FYM based on soil nutrient status and crop requirement maintains crop productivity and soil fertility as well as increases soil organic carbon under rice-rice cropping pattern in subtropical soils of Bangladesh. Thus, farmers are encouraged to apply balanced fertilizer along with 5 t/ha well decomposed organic fertilizer once a year if available for maintaining their crop productivity and soil fertility.

9. Potential barriers for adoption

Table 57. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	No	No biophysical barriers.
Cultural	No	There are no cultural barriers nowadays though earlier few people thought that quality of product deteriorates due to fertilization.
Social	No	There are no social barriers.
Economic	Yes/No	Sometimes farmers did not apply balanced fertilizer as the price of P and K containing fertilizers are higher than that of N containing fertilizer as government provide high subsidy on N fertilizer.
Institutional	No	Government and other agricultural research organization promote balanced fertilization practices.
Legal (Right to soil)	No	There are no legal barriers for adopting the practice.
Knowledge	Yes	Some farmers are still do not know the importance of balanced fertilization, thus there is still a knowledge gap.

Photos

Photo 33. Experimental layout showing all the unit plots (top) and growing rice crop (bottom)

References

Begum, S.A. & Kader, M.A. 2018. Intercropping short duration leafy vegetables with pumpkin in subtropical alluvial soils of Bangladesh. *South Pacific Journal of Natural and Applied Sciences*, 36(1): 27-35. https://doi.org/10.1071/SP18004

Begum, K., Yeluripati, J., Kader, M.A., Smith, P., Kuhnert, M., Parton, W. & Ogle, S. 2018a. Soil organic carbon sequestration and mitigation potential in a rice cropland in Bangladesh - a modelling approach. *Field Crop Research,* 226: 16-27. https://doi.org/10.1016/j.fcr.2018.07.001

Begum, K., Kuhnert, M., Yeluripati, J., Ogle, S., Parton, W., Kader, M.A. & Smith, P. 2018b. Model based regional estimates of soil organic carbon sequestration and greenhouse gas mitigation potentials from rice croplands in Bangladesh. *Land*, 7 (82): 1-18. https://doi.org/10.3390/land7030082

Brammer, H. 1996. *The Geography of the Soils of Bangladesh*. University Press Limited, Red Crescent Building, 114 Motijheel C/A, P.O. Box 2611, Dhaka 1000, Bangladesh.

Egashira, K., Han, J., Satake, N., Nagayama, T., Mian, M.J.A. & Moslehuddin, A.Z.M. 2005. Field experiment on long-term application of chemical fertilizers and farmyard manure in floodplain soil of Bangladesh. *Journal of Faculty of Agriculture Kyushu University*, 50: 861–870.

Islam, M.M, Hossain, M.F., Mia, M.M., Islam, M.S., Bhuiyan, M.S.H., Talukder, J.A. & Kader, M.A. 2019a. Long-term fertilization effect of organic carbon and total nitrogen on floodplain soil. *International Journal of Advanced Geosciences*, 7(2): 139-141.

Islam, M., Kader, M.A., Hossain, M.S., Bhuiyan, S.C., Talukder, J.A., Rahman, M.M. & Ahmed, F. 2019b. Effect of long term fertilization on soil respiration and enzyme activities in floodplain soil. *International Journal of Research in Agronomy,* 2(2): 29-34.

Kader, M.A., Yeasmin, S., Solaiman, Z.M., De Neve, S. & Sleutel, S. 2017. Response of hydrolytic enzyme activities and N mineralization to fertilizer and organic matter application in two long-term subtropical paddy field experiments. *European Journal of Soil Biology*, 80: 27-34. https://doi.org/10.1016/j.ejsobi.2017.03.004

Mian, M.J.A., Blume, H.P., Bhuiya, Z.H. & Eaqub, M. 1991. Water and nutrient dynamics of a paddy soil of Bangladesh. *Zeitschrift für Pflanzenernährung und Bodenkunde,* 154:93–99.

Mostofa, B., Raihan, M.Z., Hossain, M.F., Farhana, T., Mia, M.M. & Kader, M.A. 2015. Effects of long-term mineral fertilization and manuring on rice-rice cropping pattern in sub-tropical floodplain soil. *Asian Journal of Medical and Biological Research*, 1(2): 222-229.

Uddin, M.J., Hooda, P., Mohiuddin. S., Smith, M. & Waller, M.P. 2019. Land Inundation and Cropping Intensity Influences on Organic Carbon in the Agricultural Soils of Bangladesh. *Catena*, 178: 11-19. https://doi.org/10.1016/j.catena.2019.03.002

14. Organic rice cultivation with internal nutrient cycling in Japanese Andosols

Valensi Kautsar[1,2], Weiguo Cheng[1,3], Keitaro Tawaraya[3], Susumu Yamada[4], Kazunobu Toriyama[3,5], Kazuhiko Kobayashi[6]

[1]The United Graduate School of Agricultural Sciences, Iwate University, Morioka, Japan

[2]Faculty of Agriculture, Stiper Agricultural University, Yogyakarta, Indonesia

[3]Faculty of Agriculture, Yamagata University, Tsuruoka, Japan

[4]Department of Bioresource Development, Faculty of Agriculture, Tokyo University of Agriculture, Tokyo, Japan

[5]Japan International Research Center for Agricultural Sciences, Tsukuba, Japan

[6]Graduate School of Agricultural and Life Sciences, The University of Tokyo, Tokyo, Japan

1. Related practices

Organic farming, rice straw residues management, Crop rotations

2. Description of the case study

Since the 1970s, per capita rice (*Oryza* spp.) consumption in Japan has decreased, and the country is self-sufficient in terms of rice production (Cheng *et al.*, 2018; Coyle, 1981); thus, the agricultural sector has been focused on improving quality rather than quantity (Efferson, 1985). Organic farming could improve the quality of rice grown in Japan while simultaneously maintaining environmental quality (Cheng *et al.*, 2015; Surekha *et al.*, 2010). Organic farming was developed to reduce the negative impacts of agriculture on the environment while sustaining the productivity of the ecosystems. While crop yield in organic farming is usually lower than in conventional farming, under best management practices the organic yield nearly matches conventional yield (Hokazono, Hayashi and Sato, 2009; Seufert, Ramankutty and Foley, 2012). In this case study, organic farming was implemented gradually to replace conventional farming in 1992, through the removal of synthetic inputs and the application of rice plants residues, i.e., straw, rice bran, and the incorporation of winter fallow grasses in farmer fields. In the area, a single-cropping rice system is used, and after harvest, the field is left fallow until the next planting season. Gradually, the soil physico-chemical properties improved while the yield was maintained. Long-term organic farming decreased bulk density by about 17.5 percent compared to conventional. Meanwhile, the chemical properties of the soil recorded a slight increase in electrical conductivity and a decrease in soil pH. The carbon decomposition, nitrogen mineralization potential, and their respective ratios to soil organic carbon or total nitrogen were significantly increased after 8–9 years of organic farming. In addition, the carbon and nitrogen stocks were significantly increased after 12 years of organic farming. Thus, the organic farming practices which depend on internal nutrient cycling confirmed that this farming system provides the benefit of restoring abundant carbon to domesticated soils.

3. Context of the case study

This study was conducted in farmers′ fields in Sagawano, Nogi, Simotsuga District, Tochigi Prefecture, Japan (36°14′N, 139°46′E). Area of fields range from 775 to 2 425 m2, with an average of 1 553 m2 (Figure 5). This region is characterized by a mean annual temperature of 14.4°C and annual precipitation of 1 274 mm based on 20 years of meteorological data from Oyama meteorological station, about 12.5 km from the study site (JMA, 2020). The soil was classified as an Andosols, according to the World Reference Base (FAO, 2014). Under organic farming management, rice straw and rice bran were incorporated into the soil after harvest, and weeds that grew during the fallow season were plowed up to 15 cm into the soil the next season before rice seedlings were transplanted. Rice straw and rice bran were applied annually at 7.4 t/ha and 650 kg/ha, equivalent to 2 960 and 273 kg C/ha/yr and 42 and 16 kg N/ha/yr, respectively (Tanaka, Toriyama and Kobayashi, 2012). The off-season weeds were dominated by foxtail (*Alopecurus aequalis*), a common fallow weed in paddy fields in Japan. In most fields, >90 percent of the weed biomass was foxtail (Sakuraoka *et al.*, 2018). The second-largest off-season weed (<5 percent by biomass) was milk vetch (*Astragalus sinicus*). The total weed biomass varied between fields, with an average of 4 380 kg/ha/yr, equivalent to 1 871 kg C/ha and 43 kg N/ha.

4. Possibility of scaling up

The agricultural system carried out by farmers is feasible to be adapted in other areas because it is easy to imitate, apply, and even develop. Application of rice straw, rice bran after harvest as plant residue, would provide sufficient time for nutrients to be available during the next growing season. This practice is simple and possible to adapt in many areas because of material availability. Meanwhile, fallow weeds which incorporated while plowing as green manure is a specific location advantage in Tochigi with winter conditions that can be tolerated by weeds to grow.

5. Impact on soil organic carbon stocks

The climate in Tochigi Prefecture, Japan is a warm temperate moist climate based on IPCC (2006). Since the initial soil sample was not available, the conventional field in the current year used as a baseline of C stock (Table 58).

Table 58. Evolution of C stocks after 12 years of organic farming

Source: Adapted from Kautsar et al. (2020)

Depth (cm)	Baseline C stock in 1992 (tC/ha)	Duration (Years)	Additional C storage (tC/ha/yr)
0–15	60.3	4	1.7
		8	0.8
		12	0.8
15–20	23.5	4	0.5
		8	0.2
		12	0.1
0–20	83.7	4	2.2
		8	1.0
		12	0.9

6. Other benefits of the practice

6.1. Benefits for soil properties

Increasing the addition of organic matter resulted in lower bulk density. In this study, organic matter application decreased bulk density from 0.67 g/cm^3 on the conventional field to 0.55 g/cm^3 after 12 years of organic farming. The lower bulk density will be beneficial because it lightens tillage, support root development, and provides space for the soil organism's activity (Bouwman and Arts, 2000; Brussaard *et al.*, 2004; Kautsar *et al.*, 2020; Kobayashi *et al.*, 2008; Koolen, 1987; Unger and Jones, 1998). The changes were also noted in the chemical properties of the soil. Soil pH was closer to neutral on both conventional and organic farming; however, in general, pH tends to decrease in plots with longer organic management practices. Soil pH decreased from 6.01 to 5.90 after 4 years of organic farming and again decreased to 5.79 after 12 years. Meanwhile, soil electrical conductivity increased slowly from 195.18 µS cm^{-1} in conventional, to 295.50 µS/cm after 12 years of organic farming.

6.2 Minimization of threats to soil functions

Table 59. Soil threats

Soil threats	
Soil erosion	The presence of rice straw and winter weeds as soil cover will minimize surface runoff, the velocity of runoff, and increase water infiltration into the soil (Won *et al.*, 2012).
Nutrient imbalance and cycles	Organic farming contributes to increase nitrogen and available phosphorus (Kobayashi *et al.*, 2008; Sakuraoka *et al.*, 2018; Tanaka *et al.*, 2012).
Soil biodiversity loss	The shift to organic farming and the addition of residues had positive impacts on soil biodiversity (Brussaard *et al.*, 2004).
Soil compaction	Decreasing the value of bulk density was clear evidence that organic farming could reduce soil compaction as a result of higher biological activity in the soil (Kautsar *et al.*, 2020; Kobayashi *et al.*, 2008).

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

Even though organic farming initially reduced rice yield (Hokazono and Hayashi, 2012), in the long-term (around 12 years) it was able to produce the same or higher yields than conventional field (Kobayashi *et al.*, 2008; Tanaka, Toriyama and Kobayashi, 2012).

6.4 Mitigation of and adaptation to climate change

Organic farming had a lower rice spikelet sterility ratio under heat stress, which may be ascribed to high N supply at heading. Hence, organic farming able to survive in severe weather conditions (Tanaka, Toriyama and Kobayashi, 2011). In addition, the application of organic matter in the long-term could increase soil organic carbon (Kautsar *et al.*, 2020).

6.5 Socio-economic benefits

With the same or even higher amount of rice yields than conventional, organic farming will not affect farmers' profits. Conversely, organic rice is more expensive; thus, it will become a benefit to farmers. Besides, by eliminating the application of chemical fertilizers, farmers will save variable costs every season.

6.6 Other benefits of the practice

Increased nitrogen mineralization was demonstrated in organic agriculture. In addition, based on the illustrations developed by (Tanaka, Toriyama and Kobayashi, 2012), the potential N supply was 41.9 g N/m^2 in organic rice fields.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 60. Soil threats

Soil threats	
Soil salinization and alkalinization	The value of electrical conductivity has been noted to be increasing after 8 years of organic farming, but the value can still be tolerated by rice plants.
Soil acidification	Soil acidity decreases but remains in neutral (Kautsar *et al.*, 2020).

7.2 Increases in greenhouse gas emissions

Organic agriculture has been noted to increase carbon dioxide (CO_2) and methane (CH_4) emissions (Hokazono and Hayashi, 2012; Kautsar *et al.*, 2020).

7.3 Negative impact on production

The application of organic material without being balanced by chemical fertilizers in the early years will cause a decrease in rice yield. Without chemical fertilizers, the yield of 1–2 years of organic farming decreased by 14.5–16.3 percent compared with conventional (Tanaka, Toriyama and Kobayashi, 2012).

7.4 Other conflicts

Generally, the irrigation in agriculture highly depends on the river flows. Considering the irrigation water is important because possibly bring chemical substances from nearby fields or environment; thus, in this organic farming we use groundwater, which has less mineral and organic micro-pollutants.

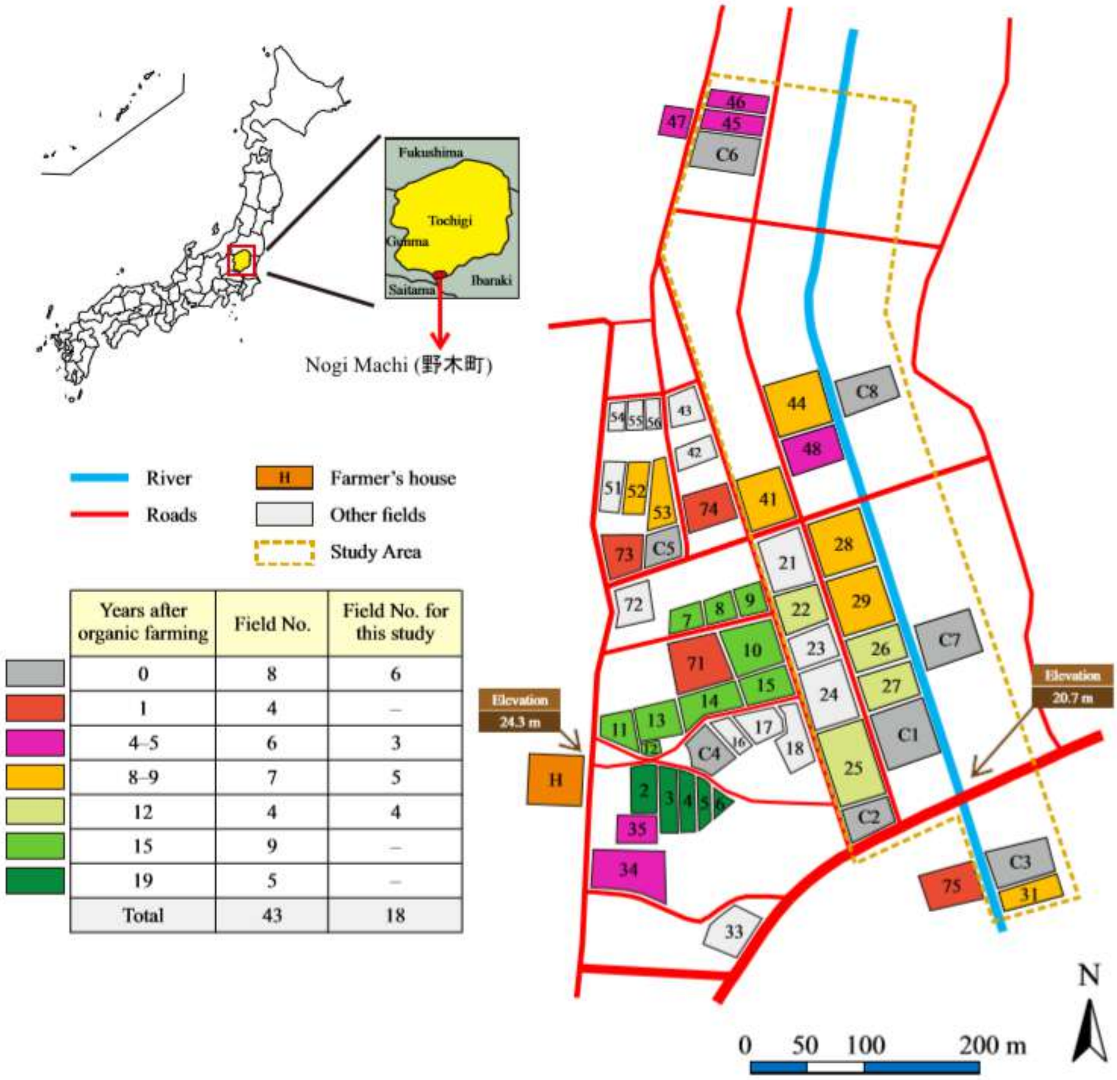
8. Recommendations before implementing the practice

It is recommended to consider the potential of local organic matter such as integrating the agriculture and animal husbandry, utilization of livestock manure, weeds, legume, crop residue, etc., which is possible to be applied from the field and environment.

9. Potential barriers for adoption

Table 61. Potential barriers to adoption

Barrier	YES/NO	
Cultural	Yes	Farmers are reluctant to adopt organic farming because of their assumption that the application of agrochemicals agents will be more productive than organic farming (Hsieh, 2005).
Economic	Yes	Organic farming practices are often regarded as uneconomic or expensive to implement besides the potential rice yield loss (Hokazono and Hayashi, 2012; Ingram *et al.*, 2014).



Years after organic farming	Field No.	Field No. for this study
0	8	6
1	4	–
4–5	6	3
8–9	7	5
12	4	4
15	9	–
19	5	–
Total	43	18

Figure 5. The map showing the site of fields with inserted map of Japan and Tochigi Prefecture to indicate research position. The fields were managed conventionally until organic rice farming was begun in 1992, gradually expanding to the more than 70 fields until 2013 under organic management. In October 2013, total of 43 fields were investigated, but we selected 18 fields located along the riverside with the original soil properties similar to those of the conventional fields for this study

Photos

Photo 34. Foxtail (top) and milk vetch (bottom) growth in the field incorporated before plowing

References

Bouwman, L.A. & Arts, W.B.M. 2000. Effects of soil compaction on the relationships between nematodes, grass production and soil physical properties. *Applied Soil Ecology*, 14(3): 213–222. https://doi.org/10.1016/S0929-1393(00)00055-X

Brussaard, L., Kuyper, T.W., Didden, W.A.M., Goede, R.G.M. de & Bloem, J. 2004. Biological soil quality from biomass to biodiversity - importance and resilience to management stress and disturbance. *In* P. Schjønning, S. Elmholt & B.T. Christensen, eds. *Managing soil quality: challenges in modern agriculture*, pp. 139–161. Wallingford, CABI. (also available at http://www.cabi.org/cabebooks/ebook/20033208661).

Cheng, W., Kimani, S.M., Kanno, T., Tang, S., Oo, A.Z., Tawaraya, K., Sudo, S., Sasaki, Y. & Yoshida, N. 2018. Forage rice varieties Fukuhibiki and Tachisuzuka emit larger CH_4 than edible rice Haenuki. *Soil Science and Plant Nutrition*, 64(1): 77–83. https://doi.org/10.1080/00380768.2017.1378569

Cheng, W., Takei, M., Sato, C., Kautsar, V., Sato, S., Tawaraya, K. & Yasuda, H. 2015. Combined use of Azolla and loach suppressed paddy weeds and increased organic rice yield: second season results. *Journal of Wetlands Environmental Management*, 3(1): 1–13.

Coyle, W.T. 1981. *Japan's Rice Policy*. Foreign Agricultural Economic Report No. 164. Washington, DC. USA.

Efferson, J.N. 1985. Rice quality in world markets. *Rice grain quality and marketing*, pp. 1–13. Manila, Philippines, International Rice Research Institute.

FAO. 2014. *World reference base for soil resources 2014: international soil classification system for naming soils and creating legends for soil maps.* Rome, FAO.

Hokazono, S. & Hayashi, K. 2012. Variability in environmental impacts during conversion from conventional to organic farming: a comparison among three rice production systems in Japan. *Journal of Cleaner Production*, 28: 101–112. https://doi.org/10.1016/j.jclepro.2011.12.005

Hokazono, S., Hayashi, K. & Sato, M. 2009. Potentialities of organic and sustainable rice production in Japan from a life cycle perspective. *Agronomy Research*, 7(1): 257–262.

Hsieh, S.-C. 2005. Organic farming for sustainable agriculture in Asia with special reference to Taiwan experience. *Research Institute of Tropical Agriculture and International Cooperation, National Pingtung University of Science and Technology, Pingtung, Taiwan 65*: 30.

Ingram, J., Mills, J., Frelih-Larsen, A., Davis, M., Merante, P., Ringrose, S., Molnar, A., Sánchez, B., Ghaley, B.B. & Karaczun, Z. 2014. Managing Soil Organic Carbon: A Farm Perspective. *EuroChoices*, 13(2): 12–19. https://doi.org/10.1111/1746-692X.12057

IPCC. 2006. Consistent Representation of Lands. *2006 IPCC Guidelines for National Greenhouse Gas Inventories Volume 4: Agriculture, Forestry and Other Land Use*, p. Kanagawa, Japan, IGES (Institute for Global Environmental Strategies).

JMA. 2020. Oyama (value per year). In: *Japan Meteorological Agency* [online]. [Cited 3 June 2020]. https://www.data.jma.go.jp/obd/stats/etrn/index.php

Kautsar, V., Cheng, W., Tawaraya, K., Yamada, S., Toriyama, K. & Kobayashi, K. 2020. Carbon and nitrogen stocks and their mineralization potentials are higher under organic than conventional farming practices in Japanese Andosols. *Soil Science and Plant Nutrition*, 66(1): 144–151. https://doi.org/10.1080/00380768.2019.1705739

Kobayashi, Y., Suzuki, S., Watanabe, N., Yoshizawa, T., Ueki, Y., Suzuki, T. & Kaneda, S. 2008. Effect of long-term organic matter application to double-cropping paddy fields on rice [*Oryza sativa*] and barley [*Hordeum vulgare*] yields in Tochigi prefecture [Japan]. *Bulletin of the Tochigi Prefectural Agricultural Experiment Station (Japan)*.

Koolen, A.J. 1987. Deformation and compaction of elemental soil volumes and effects on mechanical soil properties. *Soil and Tillage Research*, 10(1): 5–19. https://doi.org/10.1016/0167-1987(87)90003-1

Sakuraoka, R., Toriyama, K., Kobayashi, K., Yamada, S., Kamioka, H. & Mori, S. 2018. Incorporation of fallow weed increases phosphorus availability in a farmer's organic rice fields on allophanic Andosol in eastern Japan. *Soil Science and Plant Nutrition*, 64(3): 300–305. https://doi.org/10.1080/00380768.2018.1473006

Seufert, V., Ramankutty, N. & Foley, J.A. 2012. Comparing the yields of organic and conventional agriculture. *Nature*, 485(7397): 229–232. https://doi.org/10.1038/nature11069

Surekha, K., Jhansilakshmi, V., Somasekhar, N., Latha, P.C., Kumar, R.M., Rani, N.S., Rao, K.V. & Viraktamath, B.C. 2010. Status of Organic Farming and Research Experiences in Rice. *Journal of Rice Research*, 3(1): 23–35.

Tanaka, A., Toriyama, K. & Kobayashi, K. 2011. Less yield reduction induced by high temperature in a paddy field under organic fertilizer management in Tochigi prefecture. *Journal of Agricultural Meteorology*, 67(4): 249–258. https://doi.org/10.2480/agrmet.67.4.7

Tanaka, A., Toriyama, K. & Kobayashi, K. 2012. Nitrogen supply via internal nutrient cycling of residues and weeds in lowland rice farming. *Field Crops Research*, 137: 251–260. https://doi.org/10.1016/j.fcr.2012.09.005

Unger, P.W. & Jones, O.R. 1998. Long-term tillage and cropping systems affect bulk density and penetration resistance of soil cropped to dryland wheat and grain sorghum. *Soil and Tillage Research*, 45(1–2): 39–57. https://doi.org/10.1016/S0167-1987(97)00068-8

Won, C.H., Choi, Y.H., Shin, M.H., Lim, K.J. & Choi, J.D. 2012. Effects of rice straw mats on runoff and sediment discharge in a laboratory rainfall simulation. *Geoderma*, 189–190: 164–169. https://doi.org/10.1016/j.geoderma.2012.06.017

15. Conservation tillage to tackle smog issue and improve carbon sequestration in rice-wheat cropping system in Pakistan

Waqar Ahmad[1,3], Munir Zia[2,4], Khalid Mahmood[5]

[1]Focal Person Asian Soil Partnership, Pakistan

[2]R&D Department, Fauji Fertilizer Company Ltd., Rawalpindi, Pakistan

[3] School of Agriculture and Food Sciences, The University of Queensland, Australia

[4]School of Biosciences, University of Nottingham, United Kingdom of Great Britain and Northern Ireland

[5]North Wyke Farm, Okehampton, Devon, Rothamsted Research, United Kingdom of Great Britain and Northern Ireland

1. Related practices

Conservation tillage, Conservation agriculture, Rice paddy management

2. Description of the case study

In the rice-wheat system used across Punjab, Pakistan, 80 percent of paddy fields are combine-harvested, which leaves large amounts of crop residues in the field. In absence of any appropriate technology, burning of these residues during the humid months of October, and November aggravates the smog issue in both Pakistani and Indian Punjab. Since rice is harvested late in the belt every year, farmers are forced to burn rice stubbles and residues in an effort to clear fields for sowing of wheat crop. The rice straw burning results in significant losses into the atmosphere (for example up to 80 percent of N (Dobermann and Witt, 2000 and Dobermann and Fairhurst, 2002), 25 percent loss of P, 21 percent loss of potassium (K) (Ponnamperuma, 1984), and 4–60 percent loss of sulphur (S) (Lefroy, Chaitep and Blair, 1994) have been reported. Besides this, population and enzymatic activities of microorganisms in the topsoil layer are also affected negatively by the burning of residue (Kumar *et al.,* 2019). It has been estimated that discontinuation of prescribed burning, at the existing soil organic matter level of ~0.65 percent (0.38 percent total organic carbon) could potentially mitigate the emission ~ 8.58 Gigagram (Gg) of CO_2, 7.06 Gg of CH_4, 0.41 Gg of N_2O, and 0.08 Gg of particulate matter per annum (Authors' own calculation based on magnitude of rice crop's residue burning in 10 districts of rice-belt, Punjab, Pakistan). Moreover, an increase of up to 30 percent in the current levels of C stock (10.2 tC per ha) of the selected area is possible by changing this management practice (calculated based on the SOC analysis of ~5000 soil samples). Shyamsundar *et al.* (2019) in a review estimated 80 percent less GHG emission (i.e. 933 kg CO_2eq/ha from conservation tillage vs 4757 CO_2eq kg per ha via farmer practice (burning followed by disc harrow)). To address the issue, various R&D organizations introduced conservation tillage practices (called happy seeder and super seeder) in the belt to help timely planting of wheat alongside chopping of the leaves, stalks/stubbles – all in one operation in combine-harvested, un-ploughed fields, without burning rice residue. In addition, as per similar practice(s) data from neighboring India, farmers can secure about 4 percent more grains, 20 percent savings in irrigation and therefore 20 percent higher profits. Pakistan Agriculture Research Center (PARC) and Punjab Agriculture department with the help of International Maize and Wheat Improvement Center (CIMMYT) and the International Centre for Agricultural Research in Dry Areas (ICARDA) are working with the private industry to manufacture the seeders. Based on the success story, USAID also supported its subsidized availability to rice farmers.

3. Context of the case study

The rice-wheat system of Punjab, Pakistan (~1.7 million hectares) falls under semi-arid climate (10-16-inch rainfall/annum) and contributes approximately 56 percent of the national total rice production. Due to a shortage of labour, about 80 percent of paddy fields are combine-harvested. The mechanical harvesting results into large amounts of leave stalks that are about 30 cm tall. Moreover, the harvested stalk is normally spread across the field after extraction of the paddy grain. Even the manual harvest leaves some stalks above the field surface although of a smaller height. Burning such stalks is the quickest and cheapest to adequately prepare the field for wheat drilling. However, this intentional burning during the winter season (October, and November) aggravates the smog issue in Pakistani Punjab and in the bordering Indian Punjab regions, which results in transportation disruptions, school closures, and health impacts. The practice also burns the organic matter, essential crop nutrients as well as beneficial fauna present in the surface layer. This is a significant add-on to the GHG emission inventory.

To address the issue, various R&D organizations introduced the concept of conservation-tillage seeders (Happy Seeder; Super Seeder) in the belt to help plant wheat (and/or apply fertilizer) alongside chopping of the leave stalks/stubbles – all in one operation in combine-harvested, un-ploughed fields, without burning rice residue. Asia has only 2.2 percent area under conservation tillage, and various programs in the region (including Pakistan) have been launched from time to time to test and promote innovative agricultural practices including reduced, zero, and conservation tillage. The conservation-tillage seeders gained popularity especially with the advancement in mechanical harvest of paddy crop in the region. The environment friendly technology will prove a boon to the farming community and the government could help farmers in making provision of this tool for improving soil health and environment for sustainable agriculture.

4. Possibility of scaling up

The conservation tillage practice requires farmers to understand the long-term benefits in terms of soil organic carbon (SOC) enhancement for scaling-up. Due to the serious trans-boundary concerns about smog, Government of Punjab Environmental Protection Department has recently imposed a ban on burning of crop residue under section 144 of Code of Criminal Procedure (The Punjab Environmental Protection Act 1997, Pakistan Penal Code). This has authorized all the regional civil administrators to impose hefty fines and imprisonment in case of violation of the Act. However, while realizing that punitive measures alone will not help to adopt this good practice, the Government of Punjab, Pakistan has also announced 80 percent subsidy to growers who want to purchase the Happy Seeder for their farms. The Pakistan Agriculture Research Council with the help of CIMMYT and ICARDA are already working with farm equipment industry in the private sector to scale-up the manufacturing of a locally modified, cost-effective and viable models of the seeders. Super Seeder, although under continuous improvement, is more suited in case of the stalk leftover after mechanical harvest whereas the Happy Seeder is more suited after manual harvest of the paddy. During the year 2016-17, USAID and ICARDA have assisted the local manufacturers to manufacture such seeders on subsidized rates. Since the Government of Punjab (Pakistan) has now banned the burning of rice crop residues/stubbles, and therefore farmers have no option but to go after such innovative technology rather than sacrifice in grain yield through late sowing of wheat crop. FAO, Pakistan also supports this initiative to make the crop production system resilient (FAO, 2020). Besides the benefits, many contradictions regarding the impact of different tillage practices also exist in the literature mainly because of the short-term studies (Cusser *et al.,* 2020). In Pakistan, the disagreement is only limited to no-till practice on heavy clayey soils. No objection is found on using CA-based practices, especially zero till, for medium to heavy-textured soils. No-tillage is the extreme form of conservation tillage and has been largely reported to increase the SOC while mitigating the GHG emission from the agricultural soils. The increases in SOC are strongly associated to the addition of crop residues. Subject to financial assistance from donors, the afore-mentioned organizations in Pakistan are eager to join hands with the private sector to mitigate the impacts of prescribed and wild burning in the belt. Government of Punjab, Pakistan, in accordance with the Federal Government Package for Agriculture, is providing PK Rs. 97500 (~587 USD) as a subsidy to farmers for purchase of the seeders. Following calculated estimates of Shyamsundar *et al.* (2019), scaling-up adoption in the initial stages to ~50 percent of the rice-wheat cropped area (0.85 million ha) across Pakistan would require approx. 6 634 seeders.

5. Impact on soil organic carbon stocks

The improvement of SOC stocks presented in Table 62 was calculated as follows:

The accumulation of SOC and net accrual is a slow process. There are many drivers to enhance its accrual process, not limited to: agro-climatic conditions, associated land management practices, land use (crop in rotation), application of chemical and organic sources of nutrients, quantity and quality of SOC being added to the fields, and C/N (Ahmad *et al.,* 2014). We believe that the accrual of SOC may initiate within minimum two crop cycles (1 year). We assumed that an increase ~30 percent may be due to the incorporation of rice straw (Ali and Nabi, 2016; Hung *et al.,* 2020; Li *et al.,* 2009), and 15-30 percent protection of SOC will be associated by adopting conservation-tillage practice which involves no burning of the existing SOC (based on various studies, literature reviewed and cited in, for this case study, and authors' own observation). The addition of SOC will also be triggered by the saving of about 26 liter per hectare less diesel on tillage operations, and 4 percent of more grains compared to late sown wheat (Shyamsundar *et al.,* 2019). The additional C released at rhizosphere level and assimilated as plant biomass will further add to the net C assimilation (Ali *et al.,* 2013).

Bulk density in puddled soils (rice-belt) is 1.8 t/m^3 (Raza *et al.,* 2005). Due to decades old puddling in the paddy fields, this higher bulk density of 1.8 t/m^3 is common in rice belt of Pakistan. The corresponding value for this case study was taken into account considering the increased bulk density of one of the fields in the Research Area of Ayub Agricultural Research Institute, Punjab. Moreover, soil properties including bulk density in paddy fields vary with the cultivation history – age of the paddy field. Soil macro-pores are strongly influenced by the variable ages of paddy rice fields and regulate the soil water flow rate and flow paths in the paddy sown fields (Yi *et al.,* 2020).

The calculation of SOC stocks was made following the FAO-LEAP methodology (FAO, 2019).

Table 62. Estimated evolution of SOC stocks after application of sustainable soil management practices in the rice belt of Punjab, Pakistan

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration	Depth (cm)	More information	Reference
Rice belt of Punjab, Pakistan (10 districts)	Semi-arid	Mainly silt loam to clay loam	10.24	3-5, it will potentially be 13-15 after the adoption of conservation tillage practice (>30% increase)	The accrual of SOC may initiate within minimum two crop cycles (1 year)	0-15	30% increase will be due to the incorporation of rice straw, and 15-30% protection of SOC will be associated by adopting conservation-tillage practice which involves no burning of the existing SOC	Unpublished

6. Other benefits of the practice

6.1. Benefits for soil properties

The conservation tillage (i.e., Happy seeder) practice helps save irrigation water by 20 percent through better conservation of soil moisture and adds mulch layer over the soil that discourages evaporation through capillary action in addition to less weed(s) growth due to restricted sunlight. In absence of any prescribed burning, there would also be no more damage to soil native fauna. A three-year trial in India concluded that Happy Seeder-based conservation-tillage treatment had infiltration rate of 0.37 cm/hr, which was significantly higher than that of conventional tillage practice where infiltration rate was observed as 0.24 cm/hr (Gathala *et al.*, 2011).

6.2 Minimization of threats to soil functions

Table 63. Soil threats

Soil threats	
Soil erosion	The adoption of this conservation tillage practice has been known to potentially decelerate soil loss and maintain soil fertility and health.
Nutrient imbalance and cycles	This practice has been reported to sustain soil fertility.
Soil biodiversity loss	The practice conserves soil biodiversity.
Soil compaction	The practice decreases soil compaction therefore enhances steady state infiltration rate of applied irrigation.
Soil water management	The practice conserves soil moisture therefore 20 percent saving in irrigation water.

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

At time of soil preparation, about 26 liter per hectare less diesel is consumed on tillage operations (Shyamsundar *et al.*, 2019). In addition, farmers are able to secure about 4 percent of more grains compared to late sown wheat (Shyamsundar *et al.*, 2019). Before introduction of mechanical harvest of the rice crop, rice residues were used as a fuel but nowadays farmers see no interest in their collection from the field (for fuel purpose).

6.4 Mitigation of and adaptation to climate change

The emissions reduction was estimated assuming the following:

1. For this case study, we used the emission factors (EFs) that mimic/simulate the prevailing conditions in Pakistan. Also, the rest of the factors being considered by the neighboring countries, especially India were taken into account. It is to note that the smog issue in Pakistani Punjab because of rice residues burning is as intense as in the bordering Indian Punjab region:
    - 1 460 gCO_2 per kg dry matter (Gupta *et al.,* 2004);
        - gCH_4 per kg dry rice straw (sourced from the AP-42 database developed for rice straw, U.S. Environmental Protection Agency, 1992);
    - 0.07 g N_2O per kg rice straw; and
    - 13 g particulate matter per kg dry rice straw (Andreae and Merlet, 2001).
2. Straw generated from the production of 1 ton of rice grains ˜ 1.4 ton (IPCC, 2006)

3. Combustion factor of 0.80 (field observation as prescribed burning is restricted to 80 percent of farmers' field)

Gupta *et al.* (2004) concluded that burning 1 ton of straw (like in this case study) releases 3 kg particulate matter, 60 kg CO, 1460 kg CO_2, 199 kg ash, and 2 kg SO_2 in the air. Moreover, significant amount of the non-plant-available nitrogen present in surface soil also is lost to the environment due to the burning practice at farmers' fields. Giardina, Sanford, Jr. and Døckersmith (2000) also reported that due to the slash burning, about 150 kg per hectare of non-plant-available N in 0-5 cm top soil was transformed by heat, of which 82 kg per hectare supplied the increase in mineral N and 68 kg N per hectare were lost from the soil.

Adoption of this practice would improve carbon sequestration in soil and it has the potential to mitigate the emission ~ 8.58 Gg of CO_2, 7.06 Gg of CH_4, 0.41 Gg of N_2O, and 0.08 Gg of particulate matter, per annum from the 10 districts of rice-belt Punjab, Pakistan.

6.5 Socio-economic benefits

Social benefits will accrue if government provides incentives to rice-wheat farmers in Punjab for the adoption of this technology. In terms of the cost savings and yield gains mentioned earlier, the areas with less intensified agriculture would conceivably gain more from adoption of this technology than the highly intensified agricultural areas, thereby potentially reducing regional inequality (Laxmi, Erenstein and Gupta, 2007). Beyond the farm level, Happy seeder technology opens a new service industry—be it for machinery manufacturers or custom hiring services (Dixon *et al.,* 2007).

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

No tradeoffs recorded.

7.2 Conflict with other practice(s)

The new approach (Happy Seeder; Super Seeder technology) contradicts farmers' traditional approaches (Business as usual practices) but is widely appreciated for being environmentally-friendly because of 80 percent less GHG emissions.

8. Recommendations before implementing the practice

First priority of the small-scale landholders should be manual harvest of paddy crop which would ensure minimum leftover (stalks) in the field. In case of labor shortage and/or large landholding, combine harvesters be opted. Under such circumstances, farmers should keep in touch with their local extension worker so that they can purchase conservation tillage equipment on subsidized price once subsidy is announced. This would not only help them acquire the equipment at lower cost buy also help so them wheat crop, timely. In case of combine harvested paddy, farmers should go for the Super seeder instead of happy seeder since the latter is better suited for manually harvested fields. Farmer education through extension workers is also a continuous requirement so that farmers receive an update in case a better version of the equipment is introduced in the market. Higher levels of adoption at farmers' end will ensure reduction in air pollution, and improvements in soil health, primarily through improvements in soil nutrient levels and soil organic matter. For this, policy makers should also be lobbied to continue subsidy over such a beneficial technology.

9. Potential barriers for adoption

Table 64. Potential barriers to adoption

Barrier	YES/NO	
Economic	Yes	Farmers have no choice except to burn the rice residues due to the very short window available for wheat sowing. Obviously, they would prefer to burn rather than delayed sowing since the latter would have economic implications, especially in terms of lower yield therefore, a penalty on farm profitability.
Institutional	Yes	The most prominent factor appears to be the lack of suitable seeders, and institutional capacity making them available for the use of small to medium sized land-holding farmers.
Knowledge	No	Farmers have superficial knowledge over the effect of burning on the loss of carbon, other macro and micronutrients, loss of biodiversity, and soil health.

Photos

Photo 35. Close up of wheat field with conservation tillage (Happy Seeder) from rice belt

Photo 36. Wheat grown with Happy Seeder. No requirement for irrigation even after 40 days of sowing thus 20 percent savings in irrigation water

Photo 37. Locally manufactured Happy Seeder. The machine is compact, light-weight and tractor mounted. It consists of two separate units, a straw management unit and a sowing unit. The Happy Seeder can handle the paddy straw and do the sowing job without any tillage. It consists of straw cutting and chopping unit and a sowing drill combined in one machine. It sows the seeds of next crop in one operational pass of the field, while retaining the rice reside as surface mulch

Photo 38. Farmer using "Super Seeder" to sow wheat immediately after mechanical harvest (using combine harvester) of paddy. At time of land preparation, about 26 liter per hectare less diesel is consumed on tillage operations. In addition, farmers can secure about 4 percent more grains compared to late sown wheat. Before introduction of mechanical harvest of the rice crop, residues were used as a fuel but now farmers seldom care about their collection from the field for fuel purpose

Photo 39. Aerial view to reflect open crop field burning of rice residues at the farm gate in Punjab, Pakistan. The main motivation for burning rice crop residues has been the timely planting of wheat. With the introduction of the Happy Seeder; and Super Seeder technology, the issue of delayed sowing seems to be resolved with many additional benefits in terms of carbon sequestration, less environmental pollution, and soil biodiversity

References

Ahmad, W., Singh, B., Dijkstra, F.A., Dalal, R.C. & Geelan-Small, P. 2014. Temperature sensitivity and carbon release in an acidic soil amended with lime and mulch. *Geoderma,* 214–215: 168–176. https://doi.org/10.1016/j.geoderma.2013.09.014

Ali, M.K., Ahmad, W., Malhi, S.S., Atta, B.M., Zia, M.H. & Ghafoor, A. 2013. Potential of carbon dioxide biosequestration of saline-sodic soils during amelioration under rice-wheat land use. *Communications in Soil Science and Plant Analysis,* 44: 2625–2635. https://doi.org/10.1080/00103624.2013.811522

Ali, I. & Nabi, G. 2016. Soil carbon and nitrogen mineralization dynamics following incorporation and surface application of rice and wheat residues. *Soil and Environment,* 35(2): 207-2015. https://doi.org/10.1080/00103624.2013.811522

Andreae, M.O. & Merlet, P. 2001. Emission to trace gases and aerosols from biomass burning. *Global Biogeochemical Cycles*, 15: 955-966. https://doi.org/10.5194/acp-19-8523-2019

Cusser, S., Bahlai, C., Swinton, S.M., Robertson, G.P. & Haddad, N.M. 2020. Long-term research avoids spurious and misleading trends in sustainability attributes of no-till. *Global Change Biology*, 26: 3715-3725. https://doi.org/10.1111/gcb.15080

Dixon, J., Hellin, J., Erenstein, O. & Kosina, P. 2007. U-impact pathway for diagnosis and impact assessment of crop improvement. *Journal of Agricultural Science*, 145: 195–206. https://doi.org/10.1017/S0021859607007046

Dobermann, A. & Witt, C. 2000. The potential impact of crop intensification on carbon and nitrogen cycling in intensive rice systems. *International Rice Research Institute,* Los Baños, Philippines, pp. 1-25

Dobermann, A. & Fairhurst, T.H. 2002. Rice Straw Management. *Better Crops International*, 16: 1–11. May, Special Supplement.

EPA. 1992. *Safeguarding the Future: Credible Science, Credible Decisions*. EPA/600/9-91/050. Expert Panel on the Role of Science at EPA, U.S. Environmental Protection Agency, Washington, DC. March 1992.

FAO. 2019. *Measuring and modelling soil carbon stocks and stock changes in livestock production systems: Guidelines for assessment (Version 1)*. Livestock Environmental Assessment and Performance (LEAP) Partnership. Rome, FAO. 170 pp. Licence: CC BY-NC-SA 3.0 IGO.

FAO. 2020. *Remote sensing for space-time mapping of smog in Punjab and identification of the underlying causes using geographic information system (R-SMOG)*. Islamabad, FAO. (also available at http://www.fao.org/documents/card/en/c/ca6989en).

Gathala, M. K., Vivak, K., Virender, K., Saharawat, Y. S., Blackwell, J. & Ladha, J. K. 2011. *Happy seeder technology: a solution for residue management for the sustainability of the rice-wheat system of the Indo-Gangetic Plains*. Presented at 5th World Congress of Conservation Agriculture Incorporating 3rd farming Systems Design Conference, Sept 2011, Brisbane, Australia, www.wcca2011.org

Giardina, C.P, Sanford, Jr. R.L. & Døckersmith, I.C. 2000. Changes in soil phosphorus and nitrogen during slash-and-burn clearing of a Dry Tropical Forest. *Soil Science Society of America Journal*, 64: 399-405. https://doi.org/10.2136/sssaj2000.641399x

Gupta, P.K., Sahai, S., Singh, N., Dixit, C.K., Singh, D.P., Sharma, C. & Garg, S.C. 2004. Residue burning in rice-wheat cropping system: Causes and implications. *Current Science*, 87(12): 1713–1715.

Hung, N.V, Maguyon-Detras, M. C., Migo, M. V., Quilloy, R., Balingbing, C., Chivenge, P., & Gummert, M. 2020. Rice straw overview: availability, properties, and management practices. *In* M. Gummert, N.V. Hung, P. Chivenge, B. Douthwaite (Eds.) *Sustainable rice straw management*. Springer Open. ISBN: 978-3-030-32372-1.

IPCC. 2006. *IPCC Guidelines for National Greenhouse Gas Inventories 2006, Prepared by the National Greenhouse Gas Inventories Programme*. Intergovernmental Panel on Climate Change, Core Writing Team, H.S, Eggleston, L. Buendia, K. Miwa, T. Ngara and K. Tanabe (Eds.). Institute for Global Environmental Strategies, Japan.

Kumar, A., Kushwaha, K.K., Singh, S., Shivay, Y.S., Meena, M.C. & Nain, L. 2019. Effect of paddy straw burning on soil microbial dynamics in sandy loam soil of Indo-Gangetic plains. *Environmental Technology and Innovation,* 16: 100469. https://doi.org/10.1016/j.eti.2019.100469

Laxmi, V., Erenstein, O. & Gupta, R.K. 2007. Impact of Zero Tillage in India's Rice-Wheat Systems. Mexico, D.F.: CIMMYT. New Delhi, India. ISBN 978-970-648-159-7. (also available at: https://repository.cimmyt.org/xmlui/bitstream/handle/10883/1064/90319.pdf)

Lefroy, R.D.B., Chaitep, W. & Blair, G.J. 1994. Release of sulfur from rice residues under flooded and non-flooded soil conditions. *Australian Journal of Agricultural Research,* 45: 657-667. https://doi.org/10.1071/AR9940657

Li, C.X., Huang, S., Peng, X.X., Huang, Q.R. & Zhang, W.J. 2009. Differences in soil organic carbon fractions between paddy field and upland field in red soil region of south China. *Journal of Agro-Environment Science*, 28: 606–611. https://doi.org/10.1007/s11368-013-0789-9

Mahajan, A. & Gupta, R.D., eds. 2009. Soil-Related Constraints in the Rice and Wheat Production. *Integrated Nutrient Management (INM) in a Sustainable Rice–Wheat Cropping System*, pp. 169–183. Dordrecht, Springer Netherlands. https://doi.org/10.1007/978-1-4020-9875-8_11

Ponnamperuma, F.N. 1984. Straw as a source of nitrogen for wetland rice. *In* Banta, S., Mandoza, C.V. (Eds) Organic Matter and Rice, IRRI, Los Banos, Philippines. pp. 117-136

Raza, W., Yousaf, S., Niaz, A., Rasheed, M.K. & Hussain, I. 2005. Sub-soil compaction effects on soil properties, nutrient uptake and yield of maize fodder (*Zea mays* L.). *Pakistan Journal of Botany*, 37(4): 933-940.

Shyamsundar, P., Springer, N.P., Tallis, H., Polasky, S., Jat, M. L., Sidhu, H. S., Krishnapriya, P. P., Skiba, N., Ginn, W., Ahuja, V., Cummins, J., Datta, I. Dholakia, H. H., Dixon, J., Gerard, B., Gupta, R., Hellmann, J., Jadhav, A., Jat, H. S., Keil, A., Ladha, J. K., Lopez-Ridaura, S., Nandrajog, S. P., Paul, S., Ritter, A., Sharma, P. C., Singh, R., Singh, D. & Somanathan, R. 2019. Fields on fire: Alternatives to crop residue burning in India. *Science*, 365 : 536-538. https://doi.org/10.1126/science.aaw4085

Yi, J., Qiu, W., Hu, W., Zhang, H., Liu, M., Zhang, D. & Jiang, Y. 2020. Effects of cultivation history in paddy rice on vertical water flows and related soil properties. *Soil and Tillage Research*, 200: 104613. https://doi.org/10.1016/j.still.2020.104613

16. Water regimes in rainfed rice-paddies in Indonesia and Thailand

Fahmuddin Agus

Indonesian Soil Research Institute, Bogor, Indonesia

1. Related practices

Alternate wetting and drying of paddy rice management.

2. Description of the case study

The conventional practice of rice cultivation in most parts of Asia is by continuous flooding during the crop season. However, during the dry season rice, or under rainfed rice system, water may not be sufficient to support the conventional flooded system. Alternate wetting and drying (AWD) may provide the answer for water management without sacrificing the yield (Chu *et al.*, 2015; Sander, Samson and Buresh, 2014).

Two cases are presented here. First, Indonesian case with conventional 5 cm continuous flooding (CF_I), and irrigating to 5 cm flooding, followed by drying until the water table reached -15 cm from surface at which time the land was flooded again to 5 cm (AWD_I) (Setyanto *et al.*, 2018). Second, a Thailand case, where the plots were flooded with 1–2 cm water during 1–14 day after broadcasting (DAB), 5 cm water during 15–105 DAB, and irrigation was stopped two weeks before harvest, at 119 DAB (CF_T). For the AWD in Thailand (AWD_T) the plots were flooded with 1–2 cm water from the first until 25 DAB, and then irrigation was stopped. The plot was re-irrigated to a depth of about 5 cm once the water level dropped to -15 cm from the soil surface (Maneepitak *et al.*, 2019). Methane (CH_4) and nitrous oxide (N_2O) fluxes were monitored from permanent chambers connected to a gas chromatography. Both studies focused on the impacts of the AWD practice on carbon stocks, GHG emissions and yield, and other soil properties have not been considered in the frame of this case-study.

3. Context of the case study

The study in Indonesia was conducted in Pati District, Central Java (6°46′39.7” S and 111°11′53.0” E) during six consecutive rice growing seasons from 2013 to 2016, three crops in the wet season (WS) from

November to March and three crops in the dry season (DS) from March to July. The location has a mean annual rainfall of about 1500 mm, and mean air temperature from 24 to 40 °C. The soil is silt loam, Aeric Endoaquepts (Soil Survey Staff, 2014). The site in Thailand was located at the Ayutthaya (14°21′54.79″N, 100°36′19.71″E, 2m above mean sea level). This region has a tropical savanna climate characterized by warm temperature throughout the year, and two distinct wet (May–October) and dry (November–April) seasons. The mean annual temperature is around 27 °C and the annual rainfall ranges from 1 000 to 1 400 mm. The soil is strongly acidic (very-fine, mixed, active, acid isohyperthermic, Vertic Endoaquepts).

4. Possibility of scaling up

Paddy rice is very common in Asia and in parts of South Asia, and also important in parts of Africa and Latin America. The AWD system can potentially be scaled up in these areas, especially those under rainfed system and in areas with relatively low (<1500 mm) annual rainfall.

5. Impact on soil organic carbon stocks

Table 65. Evolution of carbon stocks in the two locations of Indonesia and Thailand under AWD

Location	Climate zone (annual rainfall, mm)	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Duration	More information	Reference
Pati, Indonesia	707–1768†, Tropical	Silt loam, Aeric Endoaquepts	15.9‡	0.14§	3 years (6 cropping seasons)	10.7 tCO_2-e of CH_4 emission reduction	Setyanto *et al.*, (2018), data recalculated
Ayutthaya, Thailand	1000–1400, Tropical	Very fine, mixed, active, acid isohyperthermic, Vertic Endoaquepts	28.5‡	0.06§	1 year (two cropping seasons)	2.5 tCO_2-e of CH_4 emission reduction	Maneepitak *et al.* (2019), data recalculated

†Juana station, data of 2011-2015 (pusdataru.jatengprov.go.id, downloaded March 30, 2020)

‡Estimated C stock for 0-30 cm depth, assuming the same C contents of 0-15 cm depth of 5.5 (Indonesia) and 9.5 g/kg (Thailand), with that of 15-30 cm depth, and soil bulk density of 1 Mg/m^3 in both cases.

§In terms of methane emission reduction from alternate wetting and drying. Although drying of submerged soil may cause a loss of soil organic carbon in the form of carbon dioxide (CO_2), but this aspect was not studied in the two studies referred here.

6. Other benefits of the practice

6.1 Minimization of threats to soil functions

Table 66. Soil threats

Soil threats	
Soil water management	The alternate wetting and drying increases water use efficiency (Chu *et al.*, 2015; Maneepitak *et al.*, 2019; Setyanto *et al.*, 2018), implying the possibility of a larger planting area .

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

In the Thailand case, rice grain yield significantly higher under the AWD both during the wet and the dry seasons. In the wet season, the yield for CF and AWD treatments were 3.71 ± 0.18 t/ha and 4.28 ± 0.16 t/ha, respectively, in the dry the yield was 3.91 ± 0.06 t/ha and 4.20 ± 0.06 t/ha, respectively (Maneepitak *et al.*, 2019).

6.3 Mitigation of and adaptation to climate change

Significant methane emission reduction from the implementation of AWD (Table 65).

6.4 Socio-economic benefits

Under water shortage condition, water saved from AWD practice could be used for irrigating larger areas in the nearby, and hence improve the socio-economic performance for communities, such as done in a similar study from Nepal (Howell, Shrestha and Dodd, 2015).

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

No tradeoffs identified.

7.2 Increases in greenhouse gas emissions

In both cases of the Indonesian and Thailand case studies, there were very small and insignificant (<0.2 t CO_2-e ha/yr) increase in N_2O emissions under the AWD (Maneepitak *et al.*, 2019; Setyanto *et al.*, 2018).

7.3 Negative impact on production

In the Indonesian case the grain yield was not significantly affected by the AWD treatment, where the wet season yield was 6.87 t/ha for both CF and AWD treatments while the dry season yield was 5.12 t/ha for CF and 5.20 t/ha for AWD treatment (Setyanto *et al.*, 2018).

8. Recommendations before implementing the practice

This approach is necessary for areas with insufficient water supplies to optimize rice production on larger areas. Weather prediction, for instance by using weather prediction system (e.g. cropping calendar; katam.litbang.pertanian.go.id) may be needed for assessing suitable areas for the practice. Demonstration plots may be needed too to convince farmers of the superiority.

9. Potential barriers for adoption

The information provided in this table comes from a study led in Nepal in 2015 (Howell, Shrestha and Dodd, 2015), however, these observations are also relevant and adaptable to the Thailand and Indonesian case (author's opinion).

Table 67. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	A case in Nepal mentioned that farmers felt that it's more difficult to weed because of more compacted soil under AWD (Howell, Shrestha and Dodd, 2015)
Cultural	Yes	Tendency among farmers to flood the field as much as possible (Howell, Shrestha and Dodd, 2015).

Barrier	YES/NO	
Institutional	Yes	Lack of demonstration plots to convince farmers of the superiority of AWD (Howell, Shrestha and Dodd, 2015)
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

Photo 40. Flooded (left) and dried (right) conditions of rice paddy in Jakenan, Central Java, Indonesia

References

Chu, G., Wang, Z., Zhang, H., Liu, L., Yang, J. & Zhang, J. 2015. Alternate wetting and moderate drying increases rice yield and reduces methane emission in paddy field with wheat straw residue incorporation. *Food and Energy Security*, 4(3): 238–254. https://doi.org/10.1002/FES3.66

Howell, K.R., Shrestha, P. & Dodd, I.C. 2015. Alternate wetting and drying irrigation maintained rice yields despite half the irrigation volume, but is currently unlikely to be adopted by smallholder lowland rice farmers in Nepal. *Food and Energy Security*, 4(2): 144–157. https://doi.org/10.1002/fes3.58

Maneepitak, S., Ullah, H., Datta, A., Shrestha, R.P., Shrestha, S. & Kachenchart, B. 2019. Effects of water and rice straw management practices on water savings and greenhouse gas emissions from a double-rice paddy field in the Central Plain of Thailand. *European Journal of Agronomy*, 107(April): 18–29. https://doi.org/10.1016/j.eja.2019.04.002

Sander, B.O., Samson, M. & Buresh, R.J. 2014. Methane and nitrous oxide emissions from flooded rice fields as affected by water and straw management between rice crops. *Geoderma*, 235–236: 355–362. https://doi.org/10.1016/j.geoderma.2014.07.020

Setyanto, P., Pramono, A., Adriany, T.A., Susilawati, H.L., Tokida, T., Padre, A.T. & Minamikawa, K. 2018. Alternate wetting and drying reduces methane emission from a rice paddy in Central Java, Indonesia without yield loss. *Soil Science and Plant Nutrition*, 64(1): 23–30. https://doi.org/10.1080/00380768.2017.1409600

Soil Survey Staff. 2014. *Keys to Soil Taxonomy, 12th ed.* United States Department of Agriculture, Natural Resources Conservation Service, USA. 328 pp.

17. Mangrove restoration in abandoned ponds in Bali, Indonesia

Frida Sidik[1], Virni Budi Arifanti[2]

[1]Institute for Marine Research and Observation, Ministry of Marine Affairs and Fisheries, Indonesia

[2]Center for Research and Development of Social, Economy, Policy and Climate Change, Ministry of Environment and Forestry, Indonesia

1. Practice(s) used

Mangrove restoration

2. Description of the case study

Many abandoned aquaculture ponds have been re-converted to mangrove forest in an attempt to restore mangrove cover and function. Where mangrove restoration takes place in appropriate sites and with appropriate approaches, it can result in relative high rates of mangrove growth and gains in sediment accretion, especially in minerogenic mangrove setting (Balke and Friess, 2016; Primavera *et al.*, 2019). Mangroves are not dry-land forests, but a coastal habitat living in the intertidal zone that is strongly influenced by two environmental factors: hydrology and sediment supply (Ellison, 2009; Woodroffe, 1992). Restoration approaches by primarily restoring the hydrological and sedimentary regimes can lead to SOC accumulation through enhancing sediment inputs and biomass production (Cameron *et al.*, 2019; Matsui *et al.*, 2010; Sidik, Adame and Lovelock, 2019). The case study identified the recovery of C function after 10 years of restoration presented by plant growth and sediment accumulation, which can be used as a proxy for the success of C sequestration strategies in restoration site. The restoration took place in abandoned ponds in the estuary that were previously mangrove forests (Proisy *et al.*, 2017). In early 2000s, mangrove plantation was initiated in a number of ponds, mostly using *Rhizophora sp* seedlings. Natural colonization of non-planted mangroves (*Avicennia sp* and *Sonneratia alba*) was identified after many pond walls and gates were noticeably damaged. Removal of these hydrological barriers has facilitated tidal re-instatement and introduced sediment accumulation that subsequently promotes mangrove growth and increases SOC pools. A periodical measurement of surface elevation was undertaken using fixed rod surface elevation table (RSET) on the permanent plots. With a reference datum, RSET is used for assessing soil surface change, which is corresponded to soil volume, to determine the average soil gain/loss per year (Howard *et al.*, 2014). In parallel with surface elevation measurement, we assessed net primary production by monitoring mangrove growth, root production and litterfalls. We found that mangrove trees also contributed to soil C sequestration as mangrove materials are of the main source of SOC in growing mangrove forest. This restoration and monitoring approach can provide a more successful management strategy for post aquaculture practice that leads to 'blue' C benefits.

3. Context of the case study

The site was located within the catchment area of the Perancak estuary, Bali (8° 23′ 40″ S, 114° 37′ 39″ E), which comprise of a mix of aquaculture ponds, paddy fields and mangrove forests. The estuary is characterised by sedimentary limestones and alluvial platforms that attributes to minerogenic mangrove setting. The climate of the region is dominated by an annual monsoon, with an annual temperature of 29– 32 °C and annual rainfall between 1 500 and 2 500 mm.

4. Possibility of scaling up

The case study provides understanding of the important roles of hydrology in achieving mangrove restoration goals, especially in the site where hydrology is previously restricted. As walls and gates are the main barrier of mangrove establishment in our sites (Photo 41), coastal structures such as dams, breakwaters, and dykes can also act with similar consequences in other cases. Thus, the approach can be adapted to other sites where mangrove growth is sediment-dependent (minerogenic setting) but the sediment supply is lacking due to water input blockage or prolonged water inundation caused by the man-made structures.

Photo 41. Breaching of abandoned pond walls allows reinstatement of hydrological regimes

5. Impact on soil organic carbon stocks

The study of Sidik, Adame and Lovelock (2019) showed that after 10 years of restoration, the recovery of C function was demonstrated by the improvement of surface SOC content and SOC accumulation. Surface SOC content, collected from the top 5 cm soil layer, has increased from 0.50 percent (floor of active ponds) to 2.21 percent (restored mangroves), which is close to those in the nearby intact forests (2.32 percent). Hydrological recovery promoted sediment supply and mangrove development in restoration site that resulted in increase of soil surface elevation of 0.9 cm/year, which is equivalent to 1.5 MgC/ha/year.   The rate was close to nearby intact forests that accumulated soil C with average rate of 2.2 MgC/ha/year. Additional sediment samples were also taken from restoration site and intact forests to assess carbon stock to a depth 80 cm using PVC pipe. The estimate of C stored in the sediment (<100 cm) in restored mangroves was much higher (167 MgC/ha) than the intact forests (127 Mg C/ha), which may possibly due to high soil C content of some ponds than those of intact forests.

6. Other benefits of the practice

6.1. Benefits for soil properties

We assessed the soil characteristics, including soil texture, bulk density, and porewater salinity. Soil texture was dominated by silt loam and silt clay loam with relatively uniform bulk density (0.74 – 0.86 g/cm^3) throughout the sites and sediment depth profile (Ardana, 2019; Sidik, 2014). Porewater salinity ranged from 21 to 37 ppt and was slightly high during the dry season (Ardana, 2019).

6.2 Minimization of threats to soil functions

Table 68. Soil threats

Soil threats	
Soil erosion	Mangrove roots trap sediments and facilitate sediment accumulation, enabling mangroves to maintain or increase soil surface elevation, which are 0.9 cm/year (restored mangroves) and 1.1 cm/year (intact forests) (Sidik, 2014)
Soil water management	There is positive correlation between rainfall and root growth, leading to increase in soil volume and soil surface elevation (Sidik, 2014).

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

Natural recruitment, triggered by hydrological reinstatement, has created a mosaic of species mixture in restoration site due to the natural dispersal of seedlings/propagules (Photo 42). This can result in an increase of species richness that leads to higher biodiversity and ecosystem productivity (Bosire *et al.*, 2008).

Photo 42. A mixture of planted and non-planted mangrove in restoration sites around Perancak River, Bali

6.4 Mitigation of and adaptation to climate change

This strategy can support the climate incentives for restoration of non-productive ponds. In terms of climate change mitigation, C emissions by mineralization of soil carbon and mangrove loss due to pond construction (see section 7.2) can be compensated through restoration. With the mean of soil elevation change rates of 0.9 cm/year, the 10 years of restoration caused a gain of 9 cm of soil with C removal to the system. Our value may be lower than the other restoration sites, which can mostly be attributed to variation in soil properties and geomorphological processes (Rovai *et al.*, 2018; Twilley, Rovai and Riul, 2018).

Surface and subsurface soil accumulation most likely contribute to Increase in soil surface elevation, which is the key of mangroves to keep up with sea level rise (Cahoon and Guntenspergen, 2010).

6.5 Socio-economic benefits

Mangrove species richness provides a variety of mangrove products that are benefiting local incomes. Natural regeneration of non-planted mangroves can generate diverse mangrove products come from multispecies, while planted mangroves tend to be monospecies (*Rhizphopora* sp.), for example food sources from *Nypa frutica, Bruguiera gymnorrhiza, Sonneratia caseolaris, Acanthus ilicifolius,* and dyes from *Ceriops decandra, Bruguiera gymnorrhiza*.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 69. Soil threats

Soil threats	
Soil compaction	Soil subsidence was detected in some monitoring plots, which might be caused by soil compaction (Sidik, Neil and Lovelock, 2016; Whelan *et al.*, 2005).

7.2 Increases in greenhouse gas emissions

A study in aquaculture ponds in Kalimantan, Indonesia suggested that losing one hectare of an intact mangrove forest converted to aquaculture ponds is equivalent to - 1925 tCO_2eq/ha/year with SOC stock in the 0–300 cm soil profile decreased by about 55 percent (Arifanti *et al.*, 2019). In active ponds, the estimated emissions from soils vary from 18.8 (mineral soil) to 25.6 (organic soil) Mg CO_2 eq/ha/year (Cameron *et al.*, 2019; Sidik and Lovelock, 2013).

7.3 Conflict with other practice(s)

Economic incentives to boost aquaculture production may result in re-conversion of restored mangroves to aquaculture ponds. Silvofishery has been proposed as a *win-win* solution to address this issue, combining aquaculture practice with mangrove plantation inside the ponds. However, silvofishery only has some potential for small C sequestration through plant growth and has no reported values for SOC (van Oudenhoven *et al.*, 2014) since this practice keeps the dykes maintained and restrict water exchange. Alternatively, the present approach can be proposed in combination with aquaculture practice in the frame of carbon incentives where a minimum of 50 percent mangrove cover should be set at an aquaculture farm (McEwin and McNally, 2014).

8. Recommendations before implementing the practice

Successful restoration requires appropriate soil surface elevation (Oh, Friess and Brown, 2017), and thus measurement of soil surface elevation and tides is necessary, especially if this technique will be combined with plantation. Both attribute to inundation hydroperiod that influences mangrove colonization establishment and early development (Krauss *et al.*, 2008).

9. Potential barriers for adoption

Table 70. Potential barriers to adoption

Barrier	YES/NO	
Cultural / Social	Yes	The key process of this proposed approach is tidal re-instatement by breaking the walls or gates. Landowners may refuse to break the walls and prefer to maintain it for specific purposes (e.g. silvo-fishery, ecotourism, land protection).
Economic	Yes	Mangrove plantation can generate income for local people from seedlings/propagules sales and man-hour workers (Walton *et al.*, 2006). The community may reject the proposed approach, although it can be cost-effective, as it requires less or no seedlings/propagules and less manpower for restoration effort.
Institutional	Yes	Plantation may result in rapid mangrove development and is more preferable than proposed approach, especially when the government or coastal managers have desire for a rapid fix.
Legal (Right to soil)	Yes	Because we still face with land tenure conflicts, i.e. local people who operate the ponds illegally do not always want to plant mangroves in their ponds (sylvofishery).
Knowledge	Yes	The adoption of this approach may take time as mangrove restoration has been known as effort in 'plantation' instead of 'bringing mangrove to the previous state'. Hydrology and soil knowledge is neglected in plantation effort.

References

Ardana, B.A. 2019. Pemetaan distribusi jenis tanah dan salinitas tanah terhadap nilai dominasi spesies mangrove di Desa Perancak dan Desa Budeng Bali. Universitas Brawijaya

Arifanti, V.B., Kauffman, J.B., Hadriyanto, D., Murdiyarso, D. & Diana, R. 2019. Carbon dynamics and land use carbon footprints in mangrove-converted aquaculture : The case of the Mahakam Delta, Indonesia. Forest Ecology and Management, 432: 17–29. https://doi.org/10.1016/j.foreco.2018.08.047

Balke, T. & Friess, D.A. 2016. Geomorphic knowledge for mangrove restoration: A pan-tropical categorization. Earth Surface Processes and Landforms, 41: 231–239. https://doi.org/10.1002/esp.3841

Bosire, J.O., Dahdouh-Guebas, F., Walton, M., Crona, B.I., Lewis III, R.R., Field, C., Kairo, J.G. & Koedam, N. 2008. Functionality of restored mangroves: A review. Aquatic Botany, 89(2): 251–259. https://doi.org/10.1016/j.aquabot.2008.03.010

Cahoon, D.R. & Guntenspergen, G.R. 2010. Climate Change, Sea-Level Rise, and Coastal Wetlands. National Wetlands Newsletter, 32: 8–12.

Cameron, C., Hutley, L.B., Friess, D.A. & Brown, B. 2019. High greenhouse gas emissions mitigation benefits from mangrove rehabilitation in Sulawesi, Indonesia. Ecosystem Services, 40: 101035. https://doi.org/10.1016/j.ecoser.2019.101035

Ellison, J.C. 2009. Geomorphology and sedimentology of mangroves. In G.M.E. Perillo, E. Wolanski, D.R. Cahoon & M.M. Brinson, eds. Coastal wetlands an integrated ecosystem approach, pp. 565–591. Elsevier.

Howard, J., Hoyt, S., Isensee, K., Pidgeon, E. & Telszewski, M. 2014. Coastal blue carbon. Methods for assessing carbon stocks and emissions factors in mangroves, tidal salt marshes, and seagrass meadows. Arlington, Virginia, Conservation International, Intergovernmental Oceanographic Commission of UNESCO, International Union for Conservation of Nature.

Krauss, K.W., Lovelock, C.E., McKee, K.L., López-Hoffman, L., Ewe, S.M.. & Sousa, W.P. 2008. Environmental drivers in mangrove establishment and early development: A review. Aquatic Botany, 89(2): 105–127. https://doi.org/10.1016/j.aquabot.2007.12.014

Matsui, N., Suekuni, J., Nogami, M., Havanond, S. & Salikul, P. 2010. Mangrove rehabilitation dynamics and soil organic carbon changes as a result of full hydraulic restoration and re-grading of a previously intensively managed shrimp pond. Wetlands Ecology and Management, 18(2): 233–242. https://doi.org/10.1007/s11273-009-9162-6

McEwin, A. & McNally, R. 2014. Organic shrimp certification and carbon financing: An assessment for the mangroves and markets project in Ca Mau Province, Vietnam. 81 pp.

Oh, R.R.Y., Friess, D.A. & Brown, B. 2017. The role of surface elevation in the rehabilitation of abandoned aquaculture ponds to mangrove forests, Sulawesi, Indonesia. Ecological Engineering, 100: 325–334. https://doi.org/10.1016/j.ecoleng.2016.12.021

van Oudenhoven, A.P.E., Siahainenia, A.J., Sualia, I., Tonneijck, F.H., van der Ploeg, S. & de Groot, R.S. 2014. Effects of different management regimes on mangrove ecosystem services in Java, Indonesia. Ocean and Coastal Management, 116: 75. https://doi.org/10.1016/j.ocecoaman.2015.08.003

Primavera, J.H., Friess, D.A., Lavieren, H. Van & Lee, S.Y. 2019. The Mangrove Ecosystem. World Seas: An Environmental Evaluation, pp. 1–34. Elsevier.

Proisy, C., Viennois, G., Sidik, F., Andayani, A., Enright, J.A., Guitet, S., Gusmawati, N., Lemonnier, H., Muthusankar, G., Olagoke, A., Prosperi, J., Rahmania, R., Ricout, A. & Soulard, B. 2017. Monitoring mangrove forests after aquaculture abandonment using time series of very high spatial resolution satellite images: A case study from the Perancak estuary, Bali, Indonesia. Marine Pollution Bulletin. https://doi.org/10.1016/j.marpolbul.2017.05.056

Rovai, A.S., Twilley, R.R., Castañeda-moya, E., Riul, P., Cifuentes-jara, M., Manrow-villalobos, M., Horta, P.A., Simonassi, J.C., Fonseca, A.L. & Pagliosa, P.R. 2018. Global controls on carbon storage in mangrove soils. Nature Climate Change, 8(6): 534–538. https://doi.org/10.1038/s41558-018-0162-5

Sidik, F. 2014. Mangrove forest responses to environmental change in Indonesia. The University of Queensland

Sidik, F., Adame, M.F. & Lovelock, C.E. 2019. Carbon sequestration and fluxes of restored mangroves in abandoned aquaculture ponds. Journal of the Indian Ocean Region: 1–16. https://doi.org/10.1080/19480881.2019.1605659

Sidik, F. & Lovelock, C.E. 2013. CO2 Efflux from shrimp ponds in Indonesia. PloS one, 8(6). https://doi.org/10.1371/journal.pone.0066329

Sidik, F., Neil, D. & Lovelock, C.E. 2016. Effect of high sedimentation rates on surface sediment dynamics and mangrove growth in the Porong River, Indonesia. Marine Pollution Bulletin, 107: 355–363. https://doi.org/10.1016/j.marpolbul.2016.02.048

Twilley, R.R., Rovai, A.S. & Riul, P. 2018. Coastal morphology explains global blue carbon distributions. Frontiers in Ecology and the Environment, 16(9): 1–6. https://doi.org/10.1002/fee.1937

Walton, M.E.M., Samonte-tan, G.P.B., Primavera, J.H., Edwards-jones, G. & Vay, L.L.E. 2006. Are mangroves worth replanting ? The direct economic benefits of a community-based reforestation project. Environmental Conservation. https://doi.org/10.1017/S0376892906003341

Whelan, K.R.T., Smith, T.J., Cahoon, D.R., Lynch, J.C., Anderson, G.H. & Geological, U.S. 2005. Groundwater control of mangrove surface elevation : shrink and swell Varies with soil depth. Estuaries, 28(6): 833–843.

Woodroffe, C.D. 1992. Mangrove sediments and geomorphology. In A.I. Robertson & D.M. Alongi, eds. Tropical mangrove ecosystems, p. 329. Washington DC, American Geophysical Union.

18. Management of common reed (*Phragmites australis*) in Mediterranean wetlands, Spain

Jose Navarro Pedreño[1], Francisco Rubio Villena[2]

[1]Department of Agrochemistry and Environment, University Miguel Hernández of Elche, Elche, Spain

[2]L'Unió de Llauradors I Ramaders, Elche, Spain

1. Related practices; hot-spot

Wetland conservation, Organic matter additions, Mulching wetlands; Wetlands

2. Description of the case study

The presence of genus *Phragmites*, common reed, is very frequent in Mediterranean wetlands. In this context, the management of this species found close to irrigation and drainage channels (Photo 43) and in general, in abandoned agricultural fields, is quite complex due to the large root system and the capacity to regrowth. Common reed can be a very competitive weed in arable farming of moist lands. Stock management and crop production is sometimes difficult due to the presence of this plant. One of the strategies applied to eliminate this plant is the use of fire, by burning them when they are in a dry state or applying some petrol to facilitate the combustion (Photo 44). However, the smoke produced leads to greenhouse gases (GHG) emissions and becomes into an environmental problem, affecting insects and other fauna as well. This hampers soil organic carbon (SOC) sequestration or maintenance, as the soil carbon stock is rapidly transformed mainly to CO_2 emitted into the atmosphere.

Another strategy traditionally used is the application of herbicides to control this plant. However, the use of pesticides in wetland areas is not recommended, as water is very sensitive to the presence of pollutants, possibly leading to an extensive pollution (diffuse pollution).

An alternative that can help control the carbon emissions and increase SOC is the use of common reed residues without burning them. An example of environmentally friendly management to avoid this problem is that which is practiced in the wetland cultivation area of "Carrizales de Elche (Els Carrisars d'Elx)", in the South East of Spain. Common reed is cut (Photo 45), when there is no damage to the avifauna, and applied as mulching, animal feeding or animal litter. Here this plant is important to maintain biodiversity because many small birds have their breeding place there and are necessary to control insects and other pests. Therefore, it is important to understand what is the best period to cut and control the common reed (Photo 46).

3. Context of the case study

The area of "Carrizales de Elche (Els Carrisars d'Elx)" is in the middle of two Ramsar sites, included in the Special Protection Areas of the European Natura 2000 Network and recognized as a national Natural Parks by the Valencian Government (regional Spanish authority). This area has an important agriculture based most of them in traditional and local products like pomegranate and melon of Carrizales.

The Natural Parks include brackish and saline pools, halophytic and submerged vegetation, and emergent reedbeds, and an extensive complex of active salt pans ("*salinas*") and seasonal saline pools, bordered by dunes and beaches. This area is important for various species of breeding waterbirds and supports a great biodiversity including avifauna, invertebrates, reptiles and fish.

The wetland cultivation area of "Carrizales de Elche", in the South East of Spain, is in the middle of the two Natural Parks as the Figure 6 shows. The soils are originally Fluvisols (although other soil groups are presented as Solonchacks and Gleysols), most of them cultivated along years, sometimes more than a century. In fact, the intensive agriculture has transformed the soils and most of them can be considered as Anthrosols (IUSS Working Group WRB, 2015). The area is under semiarid climate, warm temperate dry, and its importance, from the environmental point of view, is the connection of both Ramsar sites.

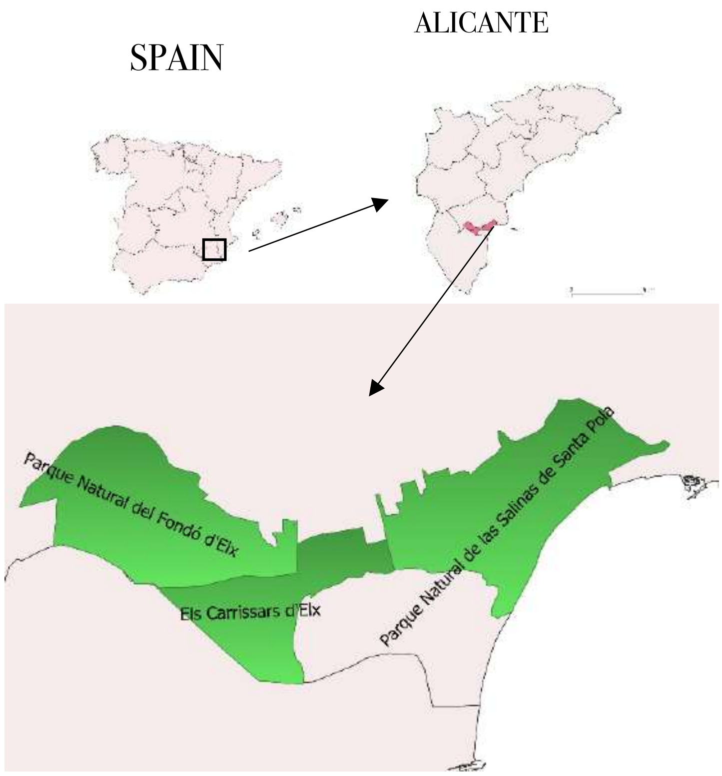


Figure 6. Location of the area of study of "Els Carrisars d'Elx"

4. Possibility of scaling up

This strategy can be used in many other Mediterranean wetlands and the case study would be a good example to be implemented in many countries where the genus *Phragmites,* wetlands and agricultural areas are in collusion.

5. Impact on soil organic carbon stocks

The mineralization and humification of common reed is determined by the ratio C/N. The ratio of carbon to nitrogen observed was high in the area, similar to that found by Obolewski, Strzelczak and Kiepas-Kokot (2007) (over 32:1). This means that it takes time for the mineralization of the plant residue and lead to an accumulation of organic matter on the topsoil with slow transformation.

The Soil Organic Matter (SOM) baseline is very variable in the area, from 0.3 to more than 4 percent of in the topsoil, depending on the proximity of wetlands and the agricultural practices. It is not easy to determine the increment of SOM derived for the use of common reed residues as mulching. However, it is important in the first step, to reduce the emissions of gases avoiding the traditional practice of burning the common reed.

It can be estimated that the SOC baseline in the topsoil would be between 0.2 to 2 percent of SOC, which means around 7 to 70 tC/ha in the agricultural areas. Applying mulching with crushed common reed of 3 cm on the topsoil, with a bulk density of the waste around 0.1 g/cm^3, the organic carbon added on the topsoil would be about 15 tC/ha. This organic matter is not easily degraded and persists for long periods covering the soil.

The increment of SOM along time depends on the location inside the area, the type of soil and the agricultural practice. Moreover, after mulching period, ploughing (15-20 cm) can help to accelerate the incorporation of the organic waste to the topsoil (Photo 47).

These estimations have been obtained in a regional survey done in the mentioned area, situated in the

Southeast of Spain, Valencian Community, under a warm temperate dry climate and in a long term cultivated soils, Anthrosols.

Plants residues from crops of these areas can also be used following the same purpose, and under adequate management, can help to increase the organic matter content of soils.

6. Other benefits of the practice

6.1. Benefits for soil properties

The benefits derived from this practice, mulching with plant residues from common reed, are greater than only avoiding GHG's emissions due to the combustion of organic wastes. Cutting and adding hydrophytes to the soils should be important to reduce the emissions avoiding the practice of burning these plants during dry season and maintain the ecosystems services associated to them. On the other hand, this management practice helps to maintain soil moisture in the dry season, saving irrigation water between 20-40 percent (Rico, Navarro-Pedreño and Gómez, 2016). After the cultivation period, the incorporation of organic matter is favoured by ploughing, preparing the soil for the next period of cultivation.

A better effect would be obtained if water tables were stabilized under the soil surface (drainage) with low fluctuations. Under an excess of moisture, the decomposition of the plant wastes would be reduced and SOC accumulated as litter. This effect is similar to that observed in natural wetlands where thick peat layers formed from roots and rhizomes of *Phragmites australis* can be found.

6.2 Minimization of threats to soil functions

Table 71. Soil threats

Soil threats	
Soil erosion	The mulch has positive effects protecting soil to rain erosion as soil is covered (reducing drop impact).
Nutrient imbalance and cycles	The slow organic matter degradation facilitates the liberation of nutrients. However, more research is needed to clarify the dynamics of soil nitrogen.
Soil salinization and alkalinization	Avoiding burning reduces the presence of salts and oxides that can increase the salinization originated by the combustion process of the common reed.
Soil biodiversity loss	The indirect effect of controlling the common reed in a period without disturbance of avifauna nesting period, and also avoiding burning the common reed, soil biota is favoured.
Soil water management	Mulching reduces water evaporation during summer.

6.3 Mitigation of and adaptation to climate change

Reduction of the GHG emissions associated to the traditional practices of burning the common reed.

7. Potential barriers for adoption

Table 72. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	The use of machinery in wetland areas is always difficult due to the own structure of the areas.
Economic	Yes	Manpower and the design of appropriate machinery need an initial investment
Knowledge	Yes	It is important to learn when and how to do this practice to avoid environmental problems to fauna.

Photos

These photographs illustrated the differences between the traditional burning practice and the use of common reed cut for soil protection, including the protection of the irrigation channels (Photo 48).

Photo 43. Common reed in the area of "Carrizales de Elche" around a drainage channel

Photo 44. Traditional burning practice of common reed. Sometimes, this practice is favored by using petrol to facilitate the combustion

Photo 45. Machinery to cut the common reed. There is a similar machine that it is on a boat to cut from the drainage channels the plant

Photo 46. Common reed at the end of summer period and until winter

Photo 47. Ploughing after a period of cultivation favoring the incorporation of organic wastes to the topsoil

Photo 48. Common reed cut and use to protect the border of a drainage channel

References

Obolewski, K., Strzelczak, A. & Kiepas-Kokot, A. 2007. Chemical composition of reed Phragmites Australis (Cav.,) trin. ex steud. Versus density and structure of periphyton in various aquatic ecosystems. *Journal of Elementology*, 12(1): 63–79.

IUSS Working Group WRB 2015. *World Reference Base for Soil Resources 2014, update 2015 - International soil classification system for naming soils and creating legends for soil maps*. World Soil Resources Reports No. 106. FAO, Rome.

Rico-Hernández, J.R., Navarro-Pedreño, J. & Gómez, I. 2016. Evaluation of plant waste used as mulch on soil moisture retention. *Spanish Journal of Soil Science*, 6(2): 133-144. https://doi.org/10.3232/SJSS.2016.V6.N2.05

19. Preserving soil organic carbon in prairie wetlands of Central North America

Sheel Bansal, Brian A. Tangen

U.S. Geological Survey, Northern Prairie Wildlife Research Center, Jamestown, North Dakota, United States of America

1. Related practices and hot-spots

Wetland conservation (Avoiding drainage of wetlands), wetland restoration; Wetlands.

2. Description of the case study

Wetlands of the Prairie Pothole Region (PPR) in the Great Plains of central North America are numerous, densely distributed, and have highly productive plant and animal communities (Photo 49). When in a natural, unaltered condition, these wetlands store relatively large amounts of organic carbon in their soils (Photo 50). Human alterations, such as extensive drainage and land-use conversion for agriculture (Figure 7), have been linked with the loss of soil organic carbon (SOC) and associated emission of carbon dioxide (CO_2), as well as impacts to other ecosystem services provided by these wetlands, such as wildlife and waterfowl habitat, plant biodiversity, flood mitigation, groundwater recharge, nutrient removal and retention, and recreation (Gleason *et al.*, 2011). It has been estimated that more than half of the wetlands of the PPR have been lost due to drainage and other disturbances, with losses approaching 90 percent in some areas (Dahl, 2014; Serran *et al.*, 2018). The goal of this case study was to identify land-management strategies that are consistent with maintaining and increasing SOC stocks of PPR wetlands.

Two overarching strategies generally are promoted to preserve and enhance SOC stocks of PPR wetlands: avoided drainage and restoration. Avoided drainage (i.e. conservation) involves protecting natural, unaltered wetlands from the impacts of human activities for the purpose of retaining natural wetland hydrologic regimes and associated functions and services such as carbon storage and sequestration. Restoration involves reestablishing natural hydrology (rewetting) with the purpose of enhancing wetland functions and services that were previously lost due to human activities. Preservation and restoration of wetland habitat typically are coupled with preservation and restoration of the adjacent grasslands that additionally provide SOC sequestration potential and other ecosystem services. Avoided drainage provides immediate and long-lasting benefits of SOC stock preservation, while restoration is slower and often requires many years to replenish predrained SOC stocks. Both strategies are associated with higher methane (CH_4) emissions but lower CO_2 and nitrous oxide (N_2O) emissions.

3. Context of the case study

The PPR, an area encompassing approximately 820 000 km^2 of the United States (US) and Canada, is the largest wetland complex in North America and the 10th largest in the world (Dahl, 2014). Nearly 10 million glacially-formed, depressional wetlands are distributed throughout the landscape of the PPR. The region was historically a grassland prairie, but is now dominated by croplands, including soybean (*Glycine max*), wheat (*Triticum aestivum*), and corn (*Zea mays*). This case study examined over 500 wetlands throughout the US portion of the PPR (Figure 8). The PPR encompasses warm-temperate-dry, warm-temperate-moist, and cool-temperate-dry climate zones characterized by warm summers and cool winters. Mean annual temperature and precipitation are approximately 4 °C (39 °F) and 470 mm (18.5 inches), respectively. The fertile soils of the PPR are typified by thick, dark, surface soils and generally are categorized as Aquolls, Udolls, or Ustolls within the overall classification of Mollisols (USDA, 2020).

4. Possibility of scaling up

The strategy of avoided drainage is relevant to wetlands around the globe, especially those in regions characterized by intensive agricultural activity. Similarly, the strategy of wetland restoration is relevant to areas where human activities have resulted in wetland drainage and disturbance. The findings from this case study are most relevant to depressional, upland-embedded (referred to as 'geographically isolated' or 'non-floodplain') wetlands such as those found in the PPR, vernal pools, playa wetlands, karst wetlands, pocosin wetlands, Carolina bays, and kettle ponds that are widely distributed around the world (Tiner, 2003).

5. Impact on soil organic carbon stocks

Results of this case study from the PPR show that average SOC stocks in the upper 15 cm (~6 inches) of natural (i.e. intact, undrained) vs. drained wetlands are 78 and 56 t SOC/ha (tonnes SOC per hectare), respectively (Tangen and Bansal, 2019, 2020). Therefore, the strategy of avoided drainage represents a benefit of 22 t SOC/ha. This estimate assumes that SOC stocks of natural, non-drained wetlands are relatively stable with time in the absence of artificial disturbances. In addition to avoid loss of SOC, natural wetlands store additional carbon in wetland vegetation live biomass. The results of this case study also suggest that restoration of wetlands could result in the sequestration of approximately 0.8 to 3.0 t SOC/ha/yr (Euliss *et al.*, 2006; Badiou *et al.*, 2011; Tangen and Bansal, 2020). The process of SOC sequestration following wetland restoration, however, is highly variable and it can take many years to reach natural levels of SOC (Euliss *et al.*, 2006; Badiou *et al.*, 2011; Tangen and Bansal, 2020).

6. Other benefits of the practice

6.1. Benefits to soil properties

It has been well established that wetland drainage and human activities result in increased soil compaction and bulk density in wetlands (von Haden, Yang and DeLucia, 2020; Tangen and Bansal, 2020). Therefore, avoided drainage and reduced disturbance to wetlands would preserve natural, physical soil structure. Sustaining natural vegetation conditions in the adjacent uplands also would help avert excessive sedimentation and eutrophication, maintain natural infiltration and runoff characteristics, and limit inputs of agricultural chemicals into wetlands (Gleason and Euliss, 1998; van der Kamp, Hayashi and Gallén, 2003; McMurry, Belden and Smith, 2016; Baulch *et al.*, 2019; Bansal *et al.*, 2019). Restoration likely would have similar benefits, but it is not clear to what degree or on what timescale soil structural properties would recover. Natural or restored wetlands are also important as water storage sites to help mitigate flooding during periods of high water, and for providing habitat for wildlife and fish; most notably, natural and restored PPR wetlands provide breeding, brood-rearing, and migratory stopover habitat for a large proportion of North America's waterfowl (Batt *et al.*, 1989).

6.2 Minimization of threats to soil functions

Minimizing soil treats. The effects of wetland management have ancillary effects on soils described in the list below.

Table 73. Soil threats

Soil threats	
Soil erosion	Avoided drainage and preservation of natural wetland habitats (including adjacent uplands) prevents excessive soil erosion and sediment deposition (Gleason and Euliss, 1998).
Nutrient imbalance and cycles)	Avoided drainage and restoration of wetlands preserves/restores natural nutrient cycling (Marton, Creed and Lewis, 2015).
Soil contamination / pollution	Wetland drainage and human disturbance in the adjacent uplands can result in inputs of agricultural chemicals and nutrients (e.g., nitrogen and phosphorus). Therefore, avoided drainage and restoration of wetland habitats (including the adjacent uplands) would avert these impacts.
Soil biodiversity loss	Avoided drainage or restoration helps maintain natural wetland animal, plant, and microbial communities (Gleason *et al.* 2011).

Soil threats	
Soil compaction	Avoided drainage and preservation of natural wetland habitats can prevent soil compaction.
Soil water management	Natural and restored wetlands store excess water during periods of high water and flooding (Gleason *et al.*, 2011).

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

Avoided drainage and preservation of natural wetland habitats, along with restoration, can provide land for livestock grazing, haying practices, or biomass for biofuel production (Photo 51). Indigenous communities also use wetland vegetation such as cattail (*Typha*) for food and fuels (Morton, 1975; Mitich, 2000; Bansal *et al.*, 2019). However, these practices do reduce the amount of land available for conventional crop production.

6.4 Mitigation of and adaptation to climate change

Wetland drainage and land-use conversion for human activities can result in increased emissions of CO_2, the largest contributor to human-caused climate warming. Drainage for agriculture also increases emissions of N_2O, a potent greenhouse gas (GHG) (approximately 300 CO_2 equivalents over a 100-year time scale) associated with land-management practices such as application of agricultural fertilizer (Tangen, Finocchiaro and Gleason, 2015). Therefore, avoided drainage and preservation of natural wetland habitats, along with restoration, would avert CO_2 and N_2O emissions.

6.5 Socio-economic benefits

Preservation and restoration of wetland habitats can provide socio-economic benefits associated with wildlife habitat and outdoor recreation (Gascoigne *et al.*, 2011). Additionally, environmental markets can provide monetary compensation for ecosystem services such as SOC sequestration. Preservation of wetland habitats also has the potential to reduce costs associated with downstream flooding and lake/river eutrophication (Marton *et al.*, 2015).

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

All of the 10 soil threats are alleviated through avoided drainage or restoration of wetlands.

7.2 Increases in greenhouse gas emissions

Greenhouse gas fluxes and SOC sequestration rates of PPR wetlands are highly variable across landscapes and time. Therefore, the net GHG balance of PPR wetlands also fluctuates spatially and temporally. Consequently, determination of the long-term net GHG balance associated with the practices of avoided drainage and restoration is challenging. However, generalizations based on previous research suggest that human disturbances to PPR wetlands, such as drainage, can result in increased emissions of CO_2 and N_2O (Tangen, Finocchiaro and Gleason, 2015). Methane emissions, on the other hand, have the potential to be reduced when wetlands are drained. Despite the uncertainty, numerous studies have concluded that PPR wetlands sequester SOC (inputs exceed losses) and are net sinks of atmospheric carbon, even when accounting for increased CH_4 emissions (Euliss *et al.*, 2006; Badiou *et al.*, 2011).

7.3 Negative impact on production

Avoided drainage and restoration of wetlands reduce the amount of land available for conventional crop production.

8. Potential barriers for adoption

Potential barriers for adoption. Wetland conservation and restoration practices face challenges described in Table 74.

Table 74. Potential barriers to adoption

Barrier	YES/NO	
Economic	Yes	Wetlands on farms reduce the amount of crop production.
Legal (Right to soil)	Yes	Upland-embedded (geographically isolated) wetlands receive fewer legal protections compared to other wetland types (Marton, Creed and Lewis, 2015).

Photos

Photo 49. Wetlands in the Prairie Pothole Region of the Great Plains of central North America are numerous, densely distributed, and have highly productive plant and animal communities

Photo 50. Wetlands of the Prairie Pothole Region store large amounts of soil organic carbon

Photo 51. Biomass harvest of wetland vegetation for biofuels

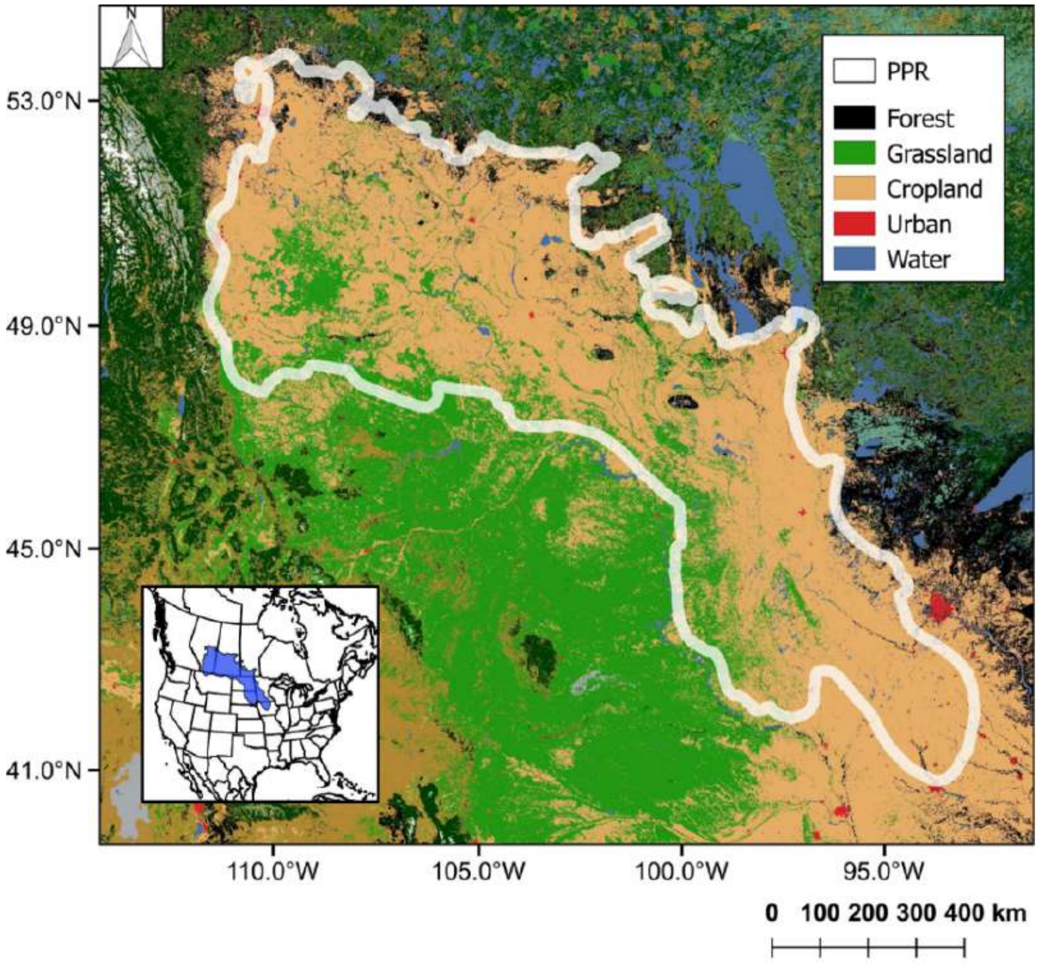


Figure 7. Croplands dominate the Prairie Pothole Region (shaded area of inset) in the United States of America and Canada, leading to extensive wetland drainage or conversion for agriculture. The legend identifies the primary land cover classes within the Prairie Pothole Region (land cover data from the Commission for Environmental Cooperation; http://www.cec.org/)

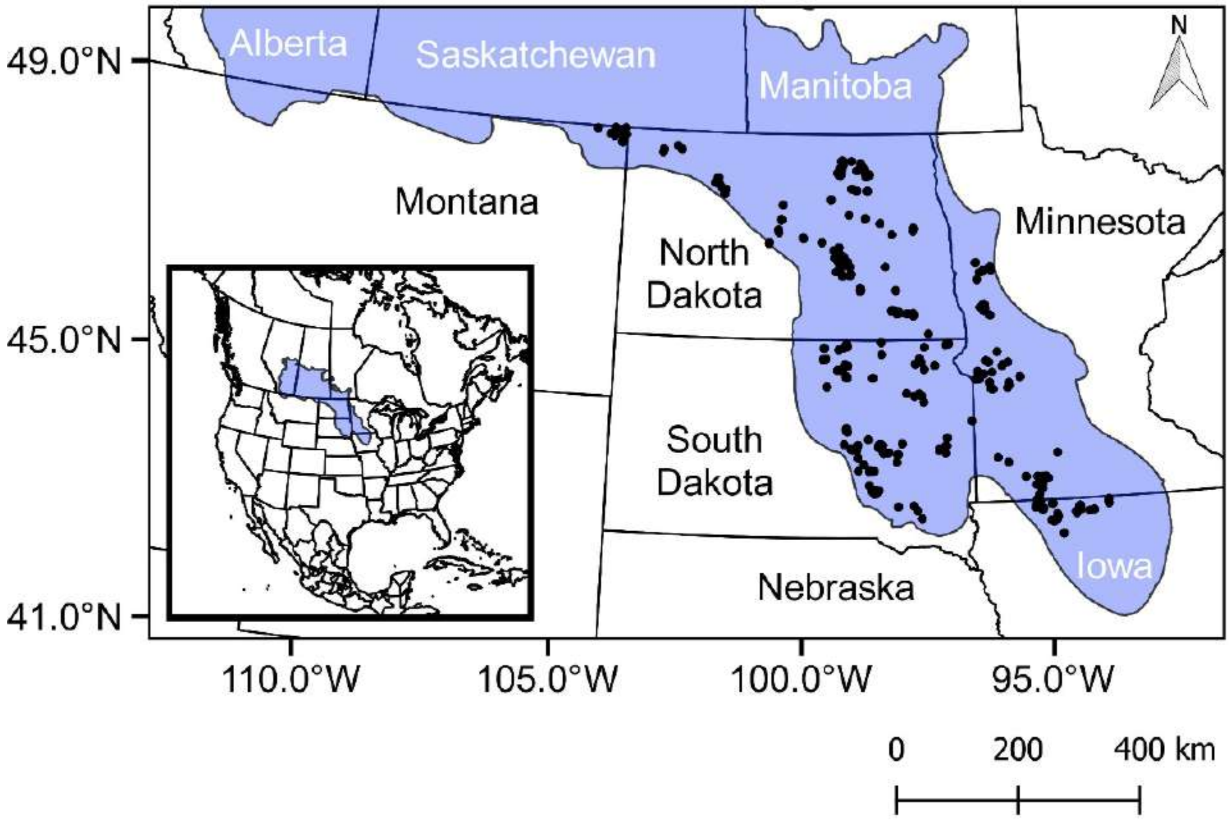


Figure 8. Prairie Pothole Region of North America (shaded area, inset) and locations of study wetlands (dots) for the case study on wetland soil organic carbon stocks

References

Badiou, P., McDougal, R., Pennock, D. & Clark, B. 2011. Greenhouse gas emissions and carbon sequestration potential in restored wetlands of the Canadian prairie pothole region. *Wetlands Ecology and Management*, 19(3): 237–256. https://doi.org/10.1007/s11273-011-9214-6

Bansal, S., Lishawa, S.C., Newman, S., Tangen, B.A., Wilcox, D., Albert, D., Anteau, M.J., Chimney, M.J., Cressey, R.L., DeKeyser, E., Elgersma, K.J., Finkelstein, S.A., Freeland, J., Grosshans, R., Klug, P.E., Larkin, D.J., Lawrence, B.A., Linz, G., Marburger, J., Noe, G., Otto, C., Reo, N., Richards, J., Richardson, C., Rodgers, L., Schrank, A.J., Svedarsky, D., Travis, S., Tuchman, N. & Windham-Myers, L. 2019. Typha (Cattail) Invasion in North American Wetlands: Biology, Regional Problems, Impacts, Ecosystem Services, and Management. *Wetlands*, 39(4): 645–684. https://doi.org/10.1007/s13157-019-01174-7

Batt, B.D.J., Anderson, M.G., Anderson, C.D. & Caswell, F.D. 1989. The use of prairie potholes by North American ducks. In*:* van der Valk, A.G., ed. *Northern Prairie Wetlands.* pp. 204–227. Ames, Iowa State University Press.

Baulch, H.M., Elliott, J.A., Cordeiro, M.R.C., Flaten, D.N., Lobb, D.A. & Wilson, H.F. 2019. Soil and water management: opportunities to mitigate nutrient losses to surface waters in the Northern Great Plains. *Environmental Reviews*, 27(4): 447–477. https://doi.org/10.1139/er-2018-0101

Dahl, T.E. 2014. *Status and trends of prairie wetlands in the United States 1997 to 2009.* U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C., USA. (also available at: https://www.fws.gov/wetlands/documents/Status-and-Trends-of-Prairie-Wetlands-in-the-United-States-1997-to-2009.pdf)

Euliss, N.H., Gleason, R.A., Olness, A., McDougal, R.L., Murkin, H.R., Robarts, R.D., Bourbonniere, R.A. & Warner, B.G. 2006. North American prairie wetlands are important non forested land-based carbon storage sites. *Science of The Total Environment*, 361(1–3): 179–188. https://doi.org/10.1016/j.scitotenv.2005.06.007

Gascoigne, W.R., Hoag, D., Koontz, L., Tangen, B.A., Shaffer, T.L., & Gleason, R.A. 2011. Valuing ecosystem and economic services across land-use scenarios in the Prairie Pothole Region of the Dakotas, USA. *Ecological Economics,* 70(10): 1715–1725. https://doi.org/10.1016/j.ecolecon.2011.04.010

Gleason, R.A. & Euliss, N.H., Jr. 1998. Sedimentation of prairie wetlands. *Great Plains Research,* 8(Spring 1998): 97–112.

Gleason, R.A., Euliss, N.H., Jr., Tangen, B.A., Laubhan, M.K. & Browne, B.A. 2011. USDA conservation program and practice effects on wetland ecosystem services in the Prairie Pothole Region. *Ecological Applications,* 21(Suppl 1): S65–S81. https://doi.org/10.1890/09-0216.1

Marton, J.M., Creed, I.F. & Lewis, D.B., *et al.* 2015. Geographically isolated wetlands are important biogeochemical reactors on the landscape. *Bioscience,* 65(4): 408–418. https://doi.org/10.1093/biosci/biv009

McMurry, S.T., Belden, J.B. & Smith, L.M., *et al.* 2016. Land use effects on pesticides in sediments of prairie pothole wetlands in North and South Dakota. *Science of the Total Environment,* 565(682–689.

Mitich, L.M. 2000. Common Cattail, *Typha latifolia* L. *Weed Technology,* 14(2): 446–450. https://doi.org/10.1614/0890-037X(2000)014[0446:CCTLL]2.0.CO;2

Morton, J.F. 1975. Cattails (*Typha* spp.) – Weed Problem or Potential Crop? *Economic Botany,* 29(1): 7–29.

Serran, J.N., Creed, I.F., Ameli, A.A. & Aldred, D.A. 2018. Estimating rates of wetland loss using power-law functions. *Wetlands*, 38(1): 109–120. https://doi.org/10.1007/s13157-017-0960-y

Tangen, B.A. & Bansal, S. 2019. Soil properties and greenhouse gas fluxes of Prairie Pothole Region wetlands: a comprehensive data release. U.S. Geological Survey data release. (also available at: https://www.sciencebase.gov/catalog/item/59a86e39e4b0421949a84627)

Tangen, B.A. & Bansal, S. 2020. Soil organic carbon stocks and sequestration rates of inland, freshwater wetlands: Sources of variability and uncertainty. *Science of The Total Environment*: 141444. https://doi.org/10.1016/j.scitotenv.2020.141444

Tangen, B.A., Finocchiaro, R.G. & Gleason, R.A. 2015. Effects of land use on greenhouse gas fluxes and soil properties of wetland catchments in the Prairie Pothole Region of North America. *Science of the Total Environment,* 533: 391–409. https://doi.org/10.1016/j.scitotenv.2015.06.148

Tiner, R.W. 2003. Geographically isolated wetlands of the United States. *Wetlands*, 23(3): 494–516. https://doi.org/10.1672/0277-5212(2003)023[0494:GIWOTU]2.0.CO;2

U.S. Department of Agriculture, natural Resources Conservation Service (USDA). 2020. Mollisols Map [online]. [Cited 25 June 2020]. www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/class/maps/?cid=nrcs142p2_053604

van der Kamp, G., Hayashi, M. & Gallén, D. 2003. Comparing the hydrology of grassed and cultivated catchments in the semi-arid Canadian prairies. *Hydrological Processes,* 17(3): 559–575. https://doi.org/10.1002/hyp.1157

von Haden, A.C., Yang, W.H. & DeLucia, E.H. 2020. Soils' dirty little secret: Depth-based comparisons can be inadequate for quantifying changes in soil organic carbon and other mineral soil properties. *Global Change Biology,* 26(7): 3759–3770. https://doi.org/10.1111/gcb.15124

20. Biomass from reeds as a substitute for peat in energy production in Lida region, Grodno oblast, Belarus

Wendelin Wichtmann[1], Felix Beer[2], Laura Villegas[3]

[1,2] University of Greifswald, partner in the Greifswald Mire Centre, Germany

[3]Food and Agriculture Organization (FAO), Rome, Italy

1. Related practices and hot-spot

Paludiculture; Restoration of peatlands; Peatland

2. Description of the case study

The case study is located in excavated fen peatlands in the Lida region in Belarus. The degraded peatlands cover several hundred hectares and were rewetted after being taken out of use. After rewetting, large reedbeds developed by natural succession. The sustainable harvesting of biomass from these rewetted peatlands (paludiculture) is being tested at several sites in cooperation with local stakeholders (nature reserves, peat factories, collective farms, local energy suppliers). Different types of adapted harvesting machinery and methods are being tested and different uses of the harvested reeds explored. Amongst them are energy production and roof thatching.

3. Context of the case study

More than half of Belarus' total peatland area – nearly 2 million hectares – has been drained for agriculture, forestry and peat extraction. Besides this area, 122 200 hectares are cutover peatlands that have been abandoned after peat excavation, and 36 800 hectares are still being utilized. During the last few years, about 50 000 ha of drained peatlands have been rewetted, and another 500 000 hectares are potentially available for hydrological restoration. The peatlands are mainly influenced by groundwater and river water, and have peat depths of 30–100 cm. After rewetting the water table is regulated between +45 to -15 cm with respect to the surface (Wichtmann *et al.,* 2016).

4. Possibility of scaling up

This practice can be adopted on all peatland sites that have been degraded by agriculture or peat exploitation. In Belarus, the peat extraction sites alone count for more than 299 000 hectares. Biofuels could be produced for the local market or for export to Western Europe. There is currently no particular interest in alternative renewable energy resources in Belarus, as briquettes made of peat or wood are still traditionally used for fuel in remote areas and there is an oversupply of nuclear energy throughout the country, which makes any alternatives unattractive and expensive.

5. Impact on soil organic carbon stocks

There are no peat soil carbon sequestration studies known for rewetted peatland sites, and hence there can only be estimations (Table 75). In general, rewetting of formerly drained organic soils not only reduces GHG emissions, but also creates a favorable environment for re-establishment of peat forming conditions and reactivating the C sink function, which is characteristic of well-functioning natural mires. In the case of reed, it is estimated that only 10–20 percent of its net primary production contributes to peat formation and maintenance, and therefore an estimated of 80–90 percent is available to harvest without compromising the peatland integrity (Wichtmann and Joosten, 2007).

Table 75. Estimation of the additional C storage after implementation of paludiculture on the Lida site

Location	Context	Additional C storage potential (tC/ha/year)	More information	Reference
Lida, Belarus	Wetland energy project at Succow Foundation, funded by EUAid	approx.+/-0-5	Succow Stiftung (2014)	Wichtmann *et al.* (2012a) Wichtmann *et al.*(2012b) Kundas *et al.* (2016) Rodzkin, Kundas and Wichtmann (2017)

6. Other benefits of the practice

6.1. Benefits for soil properties

Biomass harvesting from rewetted peatlands contributes to the export of large amounts of nutrients from the peatland site (Wichtmann *et al.,* 2014; Geurts *et al.,* 2020), which indirectly provides cleaner discharge waters and makes these sites more attractive to biodiversity adapted to natural peatlands (Tanneberger and Kucbacka, 2018). For more benefits of rewetting and paludiculture on soil properties, see Factsheets n° 12 and 13 (Volume 5 of this manual).

6.2 Minimization of threats to soil functions

Table 76. Soil threats

Soil threats	
Soil erosion	Peat is protected by high water tables which stops further decomposition of SOC, vegetation establishment prevents soil erosion.
Nutrient imbalance and cycles	As peat mineralization is stopped by rewetting, no more nutrients are mobilized. Free nitrate (NO3) is denitrified to N2, phosphate dissolution might be a problem due to bad quality of rewetting waters.
Soil biodiversity loss	If applied on former peat extraction sites, biodiversity adapted to natural near peatland conditions is increased.
Soil water management	Water is retained in the landscape with increased buffering capacities having a positive impact on mitigation of droughts and floods.

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

The production of briquettes and pellets from reed on rewetted peatlands represents an opportunity to increase the production of renewable energies on the long-term, compared to the finite character of drainage-based peat extraction for energy. In the near future, new resources from paludiculture can be used and new production lines can be developed with increasingly positive climate effects that depend on the kind of utilization of the biomass harvested from Wetland Buffer Zones (WBZs). Production of pellets and briquettes, as well as biogas from reed can represent an alternative to replace some fossil fuels. The effectivity for substitution of fossil fuels by direct burning of biomass from wet peatlands is higher than for biogas (Wichtmann *et al.*, 2019). Finally, the utilization of biomass for construction and insulation materials takes the C out of the system for several decades, which is highly appreciated in times, when the reduction of greenhouse gases is difficult.

6.4 Mitigation of and adaptation to climate change

Restoring organic soils with natural reedbeds provide a substantial GHG emissions reduction, especially of CO_2 and nitrous oxide (N_2O). By rewetting drained sites, emission reduction potential is estimated at 17 tons Carbon per hectare per year – turning formerly emitting, drained peatlands into climate neutral wet and sustainably-used peatlands. In Belarus, the paludiculture can potentially reduce 3.1–9.4 Mt CO_2eq. per year, representing 13–40 percent of the country's total emission from the agricultural sector (Günther *et al.*, 2015).

An often-underestimated climate effect of functioning wetlands is the local cooling by increased evapotranspiration (Wahren *et al.*, 2016). Furthermore, halting degradation and mineralization of the SOC, prevents further degradation connected with decreasing productivity (Zeitz, 2016), which boosts climate change adaptive capacity.

6.5 Socio-economic benefits

The regional production of biomass briquettes and pellets instead of peat briquettes is opening up new income opportunities in rural areas. Peat factories in Belarus are encouraged to re-orient their operations towards activities that use renewable biomass sustainably and replace land management practices associated with degradation.

As peat utilization for energy purposes is a finite, non-renewable resource (once peat is finished, no new concessions are given) the utilization of biomass from reeds is a sustainable solution for energy demand. The whole production and trading chain from peat-based economy can be overtaken by the reed-based economy.

6.6 Other benefits of the practice

Paludiculture on drained and degraded peatland sites that have been rewetted can improve habitats for threatened peatland species as the peat deposits, which are remaining after peat excavation, are protected due to high water tables.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Reedbed utilization competes with the abandonment and sole use of the rewetted peatlands for the development and stabilization of site adapted biodiversity. This means that, before implementation on a large scale, it is necessary to do some knowledge based spatial planning to designate areas for either paludiculture or nature development.

7.2 Increases in greenhouse gas emissions

Greenhouse gases from peatlands are dependent on water table conditions. Carbon dioxide (CO_2) and nitrous oxide (N_2O) are associated with lower water tables, and methane (CH_4) with flooded conditions – when medium water level is higher than 5 cm below ground level. However, analysis show that CH_4 emissions from rewetted peatlands do not differ significantly from undrained ones (Hiraishi *et al.*, 2013; Tanneberger *et al.*, 2020) and are smaller (in CO_2 equivalents) than CO_2 emissions from drained peat (Jablonska *et al.,* 2020). Thus, CH_4 emissions are not as important as CO_2 emissions in the long-term climate effects, as CH_4 is degraded rather fast in the atmosphere (Günther *et al.*, 2020). GHG balance also depends on the present vegetation.

7.3 Conflict with other practice(s)

Conflicts with forestry utilization or drainage-based agricultural use might occur.

7.4 Negative impact on production

As rewetted soils are rather soft, the production of biomass is strongly dependent on the availability of site adapted machinery for harvesting and transporting of biomass.

8. Potential barriers for adoption

In Belarus, harvesting of reeds is still a challenge. Major investments are necessary for special harvesting machinery that can move on soft, wet or inundated soils; this limits harvesting activities for short time periods with deep frost, which makes the wet peatlands accessible.

Photo

Photo 52. New designed harvester for cattail and common reed

References

Geurts, J.J.M., Oehmke, C., Lambertini, C., Eller, F., Sorrell, B.K., Mandiola, S.R., Grootjans, A.P., Brix, H., Wichtmann, W., Lamers, L.P.M. & Fritz, C. 2020. Nutrient removal potential and biomass production by Phragmites australis and Typha latifolia on European rewetted peat and mineral soils. Science of The Total Environment, 747: 141102. https://doi.org/10.1016/j.scitotenv.2020.141102

Günther, A., Huth, V., Jurasinski, G. & Glatzel, S. 2015. The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen. GCB Bioenergy, 7(5): 1092–1106. https://doi.org/10.1111/gcbb.12214

Günther, A., Barthelmes, A., Huth, V., Joosten, H., Jurasinski, G., Koebsch, F. & Couwenberg, J. 2020. Prompt rewetting of drained peatlands reduces climate warming despite methane emissions. Nature Communications, 11(1): 1644. https://doi.org/10.1038/s41467-020-15499-z

Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Jamsranjav, B., Fukuda, M. & Troxler, T. 2013. Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. IPCC: Geneva, Switzerland, 2014; p. 354.

Kundas, S.P., Wichtmann, W., Rodzkin, A.I., Sivograkov, O.V. & Zalesky, I.P. 2016. Utilization of wetland biomass in energy purposes. In Proceedings of the international scientific conference "Problems of rational use of natural resources and sustainable development of Polesie". Minsk 14 – 17.9.2016, Vol. 2, pp. 265 – 26.

Jabłonska, E., Wisniewska, M., Marcinkowski, P., Grygoruk, M., Walton, C.R., Zak, D., Hoffmann, C.C., Larsen, S.E., Trepel. M. & Kotowski, W. 2020: Catchment-Scale Analysis Reveals High Cost-Effectiveness of Wetland Buffer Zones as a Remedy to Non-Point Nutrient Pollution in North-Eastern Poland. Water, 12(3): 629. https://doi.org/10.3390/w12030629

Rodzkin, A., Kundas, S. & Wichtmann, W. 2017. Life cycle assessment of biomass production from drained wetlands areas for composite briquettes fabrication. Energy Procedia, 128: 261-267. https://doi.org/10.1016/j.egypro.2017.09.069

Succow Stiftung. 2014. Interreg DESIRE. In: Succow-stiftung [online]. Greifswald, Germany. [Cited 18 August 2021]. https://www.succow-stiftung.de/en/peatlands-climate/interreg-desire/

Tanneberger, F., Appulo, L., Ewert, S., Lakner, S., O'Briolchain, n., Peters, J. & Wichtmann, W. 2020 (accepted). The Power of Nature-based Solutions: How Peatlands can help us to achieve Key EU Sustainability Objectives. Advanced Sustainability Systems, 2000146, 10 p.

Tanneberger, F. & Kubacka, J. 2018. The Aquatic Warbler Conservation Handbook. Brandenburg State Office for Environment (LfU), Potsdam. P. 260.

Wahren, A., Brust, K., Dittrich, I. & Edom, F. 2016. Local climate and hydrology. In Wichtmann et. al. Paludiculture – productive use of wet peatlands. Climate protection, biodiversity, regional economic benefits. Schweizerbart Science Publishers, pp 102 – 105.

Wichtmann, W. & Joosten, H. 2007. Paludiculture: peat formation and renewable resources from rewetted peatlands. IMCG Newsletter 2007/3: 24-28.

Wichtmann, W., Haberl, A. & Tanneberger, F. 2012a. Production of biomass in wet peatlands (paludiculture). The EU-AID project `Wetland energy` in Belarus – solutions for the substitution of fossil fuels (peat briquettes) by biomass from wet peatlands. Bio-energy forum Rostock, Schriftenreihe Umweltingenieurwesen. Band 32, pp 85– 96.

Wichtmann, W., Sivagrakau, A., Haberl, A., Tanovitskaya, N., Rakovich, V. & Rodzkin, A. 2012b. Using biomass as a substitute for peat – examples for wet peatland management (paludiculture) in Belarus. In Proceedings of the IPS congress. Helsinki, June 2012, 5 p.

Wichtmann, W., Oehmke, C., Bärisch, S., Deschan, F., Malashevich, V. & Tanneberger, F. 2014. Characteristics of Biomass from wet fens in Belarus and their potential to substitute peat briquettes as a fuel. Mires & Peat, 13(6): 1-10. (also available at: http://www.mires-and-peat.net/pages/volumes/map13/map1306.php)

Wichtmann, W. Kapitsa, V., Tanneberger, F.G. & Tanovitskaya, N. 2016. Belarus – Biomass from rewetted peatlands as a substitute for peat and promoting biodiversity. pp 205 – 206. In Wichtmann *et al.* Paludiculture, productive use of wet peatlands. Scheizerbart Science publisher.

Wichtmann, W., Bork, L., Dahms, T., Körner, N., Kabengele, G.R., Oehmke, C. Wenzel, M. & Barz, M. 2019. Das Projekt Bonamoor. Biomasseproduktion und Optimierung auf nassen Moorstandorten und deren thermische Verwertung. Proceedings zum 13. Rostocker Bioenergieforum. Schriftenreihe Umweltingenieurwesen. Band 87. Universität Rostock. S. 135 – 145.

Zeitz, J. 2016. Impact of drainage on productivity. In W. Wichtmann, C. Schröder & H. Joosten, eds. Paludiculture – productive use of wet peatlands, pp. 9–12. Schweizerbart Science Publishers. Stuttgart, Germany.

21. Sphagnum farming for replacing peat as horticultural growing media, Lower Saxony, Germany

Greta Gaudig, Matthias Krebs, Sabine Wichmann

University of Greifswald, partner in the Greifswald Mire Centre, Germany

1. Related practices and hot-spot

Paludiculture; Restoration of peatlands; Peatland

2. Description of the case study

Sphagnum (peat moss) biomass provides a renewable alternative to fossil peat in horticulture. However, so far it has only been collected in the wild. When *Sphagnum* as a new agricultural crop is cultivated on rewetted bogs, the high and stable water tables greatly reduce greenhouse gas (GHG) emissions and stop the subsidence of the formerly drained peat soil. Sphagnum farming combines long-term land productivity with climate change mitigation and sustainable employment in rural areas. It also provides habitats for rare bog species and preserves the land's paleo-environmental archives.

After initial successful small-scale tests, in 2011 the first 4 hectare commercial-size pilot trial was established. The heavily degraded topsoil of a drained agricultural bog grassland was removed, a water management system installed and *Sphagnum* mosses were spread as founder material with a manure spreader mounted on a modified snow groomer. After one and a half years a well-growing *Sphagnum* lawn had been established. In 2016, the first mechanical harvest, using an excavator with a 14-metre-long arm and mowing bucket, provided *Sphagnum* shoots for extension of the Sphagnum farming trial to 14 ha (Figure 9). The results demonstrate the feasibility of large-scale Sphagnum farming. Methods and machines are now being developed to optimise and scale up cultivation.

For more information see Gaudig *et al.* (2018), Wichmann *et al.* (2020) or www.sphagnumfarming.com .

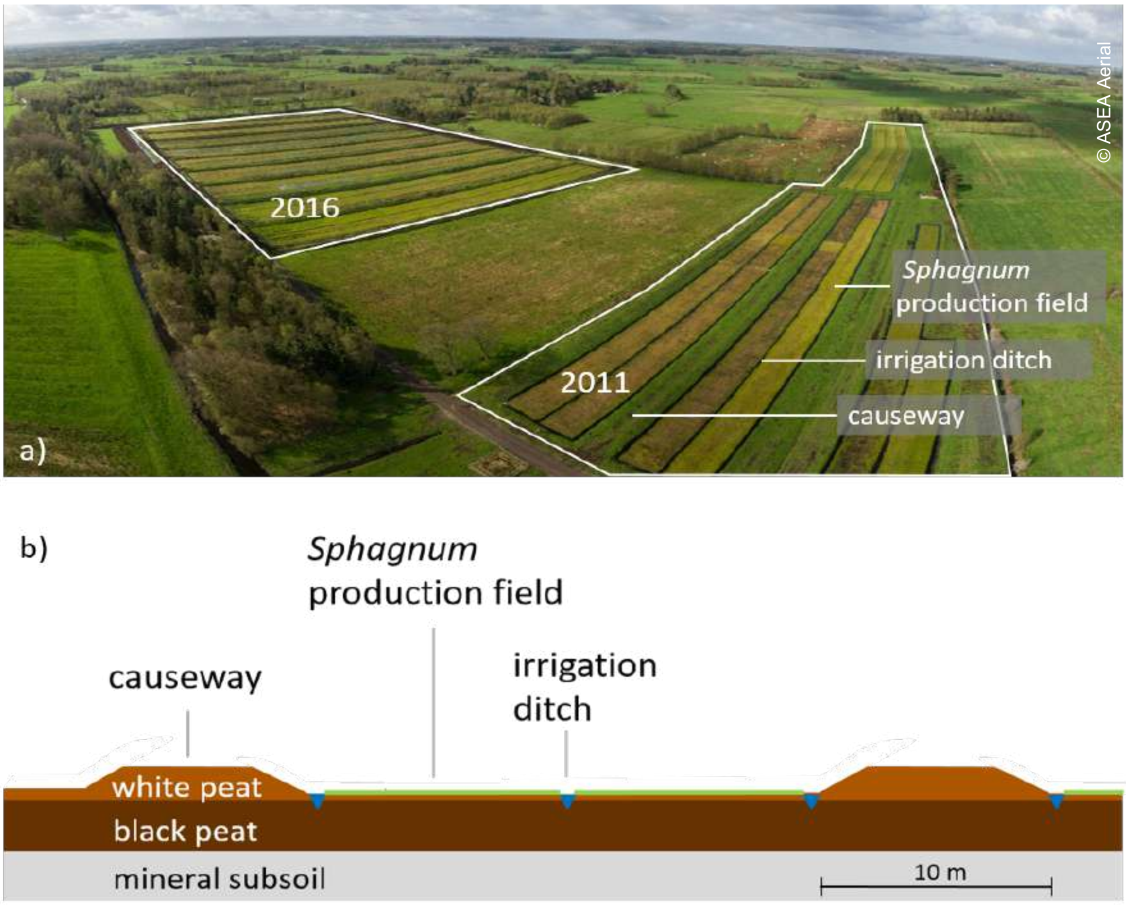


Figure 9. Sphagnum farming site in the peatland Hankhauser Moor. a) Aerial view of the pilot site with the five-year-old section established in 2011, the extension section established in 2016 and the surrounding drained grassland and b) schematic cross-section

3. Context of the case study

The case study site (53° 15.80′ N, 08° 16.05′ E) is situated in the peatland Hankhauser Moor (near Rastede, Lower Saxony, Germany), a former raised bog with a 2–2.5-metre-thick peat layer. The bog has been intensively drained for over six decades to allow agricultural use as grassland. The site was rewetted and transformed into paludiculture in 2011 and expanded in 2016 to cultivate *Sphagnum* (peat moss) as a crop and harvest *Sphagnum* biomass for its usage as raw material in growing media. For highest yields, a permanently high water table (0 to -10 cm in relation to peat moss surface) has to be ensured, mostly by active water management (Brust *et al.*, 2018).

4. Possibility of scaling up

The developed production method can be transferred to other regions, as long as sufficient water supply and water quality can be ensured. This case study has the potential for large-scale implementation.

5. Impact on soil organic carbon stocks

After rewetting, the emissions at the study site, which were caused by its former use as a drained bog grassland (ca. 23 tonnes CO_2eq per ha per year), were nearly attenuated (Günther *et al.*, 2017) and the residual peat body of ca. 2 m preserved. When removing nearly the entire "above ground" *Sphagnum* biomass by harvest, there is nearly no C sequestration. The climate change mitigation benefit results from preserving the peat layer as an enormous long-term carbon store, both on site and off site by contributing to phasing out peat extraction.

6. Other benefits of the practice

6.1. Benefits for soil properties

In comparison to drainage-based peatland use, *Sphagnum* farming preserves the existing peat layer and prevents soil subsidence and loss of productive land. Peat mineralisation and thus CO_2 emissions are stopped and evapotranspiration has a cooling effect. Additionally, nutrients that had accumulated in the peat soil during former drainage-based agriculture are declining after rewetting (Vroom *et al.,* 2020), as *Sphagnum* biomass can sequester high loads of nutrients (N, P, K), preventing off-site eutrophication and downstream pollution (*ibid.*, Temmink *et al.*, 2017).

6.2 Minimization of threats to soil functions

Table 77. Soil threats

Soil threats	
Soil erosion	Wet conditions stop peat mineralization and prevent peat soil subsidence and loss of productive land (Wichtmann, Schröder and Joosten, 2016).
Nutrient imbalance and cycles	Fertilization is not necessary since peat mosses in natural bogs are growing under nutrient poor conditions, but nutrient sources in Sphagnum farming sites are atmospheric deposition and irrigation water. With balanced stoichiometry (NPK) peat mosses grow well even under nutrient rich conditions, in particular high N loads, and sequester nutrients in their biomass (Temmink *et al.*, 2017; Gaudig, Krebs and Joosten, 2020).
Soil salinization and alkalinization	Salinization and alkalinization are rare processes in bogs. Sphagnum mosses acidify their environment (Clymo and Hayward, 1982) and prevent any kind of alkalinization.

Soil threats	
Soil contamination / pollution	Nutrients accumulation from previous practices decline after rewetting due to leaching and by removing the nutrient binding Sphagnum biomass, nutrients will be extracted (Vroom *et al.*, 2020). No pesticides are used for Sphagnum farming.
Soil acidification	Bogs are characterized by acidic site conditions due to their ecology (Rydin and Jeglum, 2013).
Soil biodiversity loss	There is no information on soil biodiversity.
Soil sealing	Sphagnum farming is agricultural use without soil sealing (Gaudig *et al.*, 2018).
Soil compaction	Because of the high water table close to the bog surface, degradation of the peat and its compaction is stopped. Adapted machinery with low soil pressure may also lower the compaction (Gaudig *et al.*, 2018).
Soil water management	For highest yields an active water management (irrigation system with ditches, pumps, outlets) is necessary to keep the water permanently just below the peat moss surface (Brust *et al.*, 2018; Gaudig *et al.*, 2020). The peat soil is completely water saturated after successful rewetting (Brust *et al.*, 2018).

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

Sphagnum farming does not provide food, but raw material to produce horticultural growing media which are partly used for growing vegetables, fruits and herbs and can support food security.

6.4 Mitigation of and adaptation to climate change

Conventional (drainage-based) bog grasslands emit ca. 23 t CO_2eq/ha/year (UBA, 2018). In contrast, *Sphagnum* farming sites emit only a tenth of that – ca. 2.5 t CO_2eq/ha/year – in the establishment phase of the *Sphagnum* lawn including all elements of the production system (Günther *et al.*, 2017). The potential for further reduction is high – e.g. by minimizing topsoil removal during site preparation and the proportion of ditches, since most greenhouse gases are emitted as methane (CH_4) from the ditches. Additionally, *Sphagnum* farming has a cooling effect on local climate. Further GHG emissions are reduced when using the produced renewable *Sphagnum* biomass for the substitution of fossil peat as growing media.

6.5 Socio-economic benefits

Sphagnum farming can help to maintain and provide employment in rural areas, especially in agriculture, substrate industry and horticulture. A commercial-scale implementation, an increasing market demand for renewables, and setting climate targets for the agricultural and horticultural sectors will accelerate the development of *Sphagnum* farming as a profitable alternative to drainage-based peatland agriculture and peat extraction (Wichmann *et al.*, 2020).

6.6 Other benefits of the practice

Sphagnum farming has also benefitted biodiversity conservation, since the sites represent a valuable surrogate habitat for rare, bog-typical species (Muster *et al.*, 2015; Gaudig and Krebs, 2016).

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Not known.

7.2 Increases in greenhouse gas emissions

Ditches emit methane and causeways emit CO_2. Nitrous oxide (N_2O) is negligible in a Sphagnum farming site.

7.3 Conflict with other practice(s)

The case study site was formerly used as bog grassland for dairy farming, considered as a typical cultural landscape and supported by agricultural payments. As long as Common Agricultural Policy (CAP) funding for drainage-based peatland utilization is continued and incentives for peatland rewetting and paludiculture are missing, the large-scale implementation of *Sphagnum* farming will be difficult (Greifswald Mire Centre *et al.*, 2020).

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

Water is the most important production factor for successful *Sphagnum* farming. When water supply is deficient, *Sphagnum* establishment and growth might be hampered. High nutrient inputs – by atmospheric deposition, irrigation water or peat soils – can affect target *Sphagnum* species and stimulate growth of other mosses and vascular plants as competitors, or the appearance of pathogenic fungi. Vascular plants can be controlled by regular mowing (Gaudig *et al.* 2017; Wichmann *et al.* 2020) and pre-treatment of the water could improve its quality for irrigating a *Sphagnum* farming site. For the large-scale implementation of *Sphagnum* farming founder material is essential to install new sites. There is no big market for it yet, but technologies for mass propagation are already developed (Beike *et al.*, 2014; Caporn *et al.*, 2018).

8. Recommendations before implementing the practice

Before implementing *Sphagnum* farming, the site conditions – peat thickness and composition, surface structure, drainage system, availability of water and its quality, hydrological situation – should be assessed carefully. Additionally, suitable and available *Sphagnum* species need to be selected. For further information – also for practical implementation of Sphagnum farming – see Gaudig *et al.* (2018).

9. Potential barriers for adoption

Besides the adjustment of the political framework more research is needed for the large-scale implementation of *Sphagnum* farming to reach technological maturity and to reduce costs. Fields of research are e.g. highly productive *Sphagnum* taxa, production of founder material, optimisation of site conditions, production and processing, development of machinery. Additionally, implementation in growing media of new, *Sphagnum* biomass-based substrates, adaption of the plant cultivation and an increase in acceptance by general public and practitioners are important (

Table 78).

Table 78. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	• Degraded peat with low water permeability: water transport to the mosses would be hampered, especially in dry seasons; • Nutrient-rich topsoil and irrigation water: competition with vascular plants; • Alkaline irrigation water is toxic for peat mosses; and • Availability of founder material is currently limited
Cultural	Yes	• *Sphagnum* farming / paludiculture is a paradigm shift from traditional drainage-based to wet agriculture in peatlands: possible resistance from the local community.
Economic	Yes	• High investment and management costs; and • Low peat price represents a tough competition on the market
Legal (Right to soil)	Yes	• *Sphagnum* is not listed as a crop: no agricultural subsidies exist; • Transformation of grassland into permanent cultures is limited: compensatory measures are necessary; • There are no economic incentives, e.g. for installation of a new *Sphagnum* farming site; and • Incentives for ecosystem services such as mitigating GHG emissions are missing.
Knowledge	Yes	• Research into highly productive *Sphagnum* taxa and optimization of site conditions are needed to enlarge the yields; and • Production and processing techniques need to be further researched.
Natural resource	Yes	• *Sphagnum* spp. are protected species: production of founder material
Other	Yes	• Development of adapted machinery; and • Implementation of in growing media

References

Beike, A.K., Spagnuolo, V., Lüth, V., Steinhart, F., Ramos-Gómez, J., Krebs, M., Adamo, P., Rey-Asensio, A.I., Angel Fernández, J., Giordano, S., Decker, E.L. & Reski, R. 2015. Clonal in vitro propagation of peat mosses (*Sphagnum* L.) as novel green resources for basic and applied research. *Plant Cell, Tissue and Organ Culture (PCTOC)*, 120(3): 1037–1049. https://doi.org/10.1007/s11240-014-0658-2

Brust, K., Krebs, M., Wahren, A., Gaudig, G. & Joosten, H. 2018. The water balance of a Sphagnum farming site in north-west Germany. *Mires Peat,* 10: 1–12. https://doi.org/10.19189/MaP.2017.OMB.301

Caporn, S.J.M., Rosenburgh, A.E., Keightley, A.T., Hinde, S.L., Riggs, J., Buckler, M. & Wright, N.A. 2018. *Sphagnum* restoration on degraded blanket and raised bogs in the UK using micropropagated source material: a review of progress. *Mires and Peat*, 20(9): 1–17. https://doi.org/10.19189/MaP.2017.OMB.306

Clymo, R.S. & Hayward, P.M. 1982. The ecology of *Sphagnum*. *In* Smith, A.J.E. (Ed.) *Bryophyte Ecology*, Chapman and Hall, London, 229–289.

Gaudig, G. & Krebs, M. 2016. Torfmooskulturen als Ersatzlebensraum – Nachhaltige Moornutzung trägt zum Artenschutz bei (*Sphagnum* cultures as surrogate habitat – sustainable peatland utilisation supports species conservation). *Biologie in unserer Zeit*, 46(4): 251–257 (in German).

Gaudig, G., Krebs, M. & Joosten, H. 2017. Sphagnum farming on cut-over bog in NW Germany: Long-term studies on *Sphagnum* growth. *Mires and Peat,* 20(4): 1–19. https://doi.org/10.19189/MaP.2016.OMB.238

Gaudig, G., Krebs, M., Prager, A., Wichmann, S., Barney, M., Caporn, S.J.M., Emmel, M., Fritz, C., Graf, M., Grobe, A., Gutierrez Pacheco, S., Hogue-Hugron, S., Holzträger, S., Irrgang, S., Kämäräinen, A., Karofeld, E., Koch, G., Koebbing, J.F., Kumar, S., Matchutadze, I., Oberpaur, C., Oestmann, J., Raabe, P., Rammes, D., Rochefort, L., Schmilewksi, G., Sendžikaitė, J., Smolders, A., St-Hilaire, B., van de Riet, B., Wright, B., Wright, N., Zoch, L. & Joosten, H. 2018. Sphagnum farming from species selection to the production of growing media: a review. *Mires and Peat,* 20(13): 1–30. https://doi.org/10.19189/MaP.2018.OMB.340

Gaudig, G., Krebs, M. & Joosten, H. 2020. Sphagnum growth under N saturation: interactive effects of water level and P or K fertilization. *Plant Biology*, 22(3): 394–403. https://doi.org/10.1111/plb.13092

Greifswald Mire Centre, National University of Ireland, Galway & Wetlands International. 2020. *Position paper on peatlands in the EU's Common Agriculture Policy (CAP) after 2020*.

Günther, A., Jurasinski, G., Albrecht, K., Gaudig, G., Krebs, M. & Glatzel, S. 2017. Greenhouse gas balance of an establishing *Sphagnum* culture on a former bog grassland in Germany. *Mires and Peat*, 20(2): 1–16. https://doi.org/10.19189/MaP.2015.OMB.210

Muster, C., Gaudig, G., Krebs, M. & Joosten, H. 2015. Sphagnum farming: the promised land for peat bog species? *Biodiversity and Conservation*, 24(8): 1989–2009. https://doi.org/10.1007/s10531-015-0922-8

Rydin, H. & Jeglum, J. 2013. *The biology of peatlands*. 2nd edition, Oxford University Press, 398 pp.

Temmink, R.J.M., Fritz, C., van Dijk, G., Hensgens, G., Lamers, L.P.M., Krebs, M., Gaudig, G. & Joosten, H. 2017. Sphagnum farming in a eutrophic world: The importance of optimal nutrient stoichiometry. *Ecological Engineering*, 98: 196–205. https://doi.org/10.1016/j.ecoleng.2016.10.069

UBA (Umweltbundesamt). 2018. Submission under the United Nations Framework Convention on Climate Change and the Kyoto Protocol 2018 - National Inventory Report for the German Greenhouse Gas Inventory 1990–2016. 949 p.

Vroom, R.J.E., Temmink, R.J.M., van Dijk, G., Joosten, H., Lamers, L.P.M., Smolders, A.J.P., Krebs, M., Gaudig, G. & Fritz, C. 2020. Nutrient dynamics of Sphagnum farming on rewetted bog grassland in NW Germany. *Science of The Total Environment*, 726: 138470. https://doi.org/10.1016/j.scitotenv.2020.138470

Wichmann, S., Krebs, M., Kumar, S. & Gaudig, G. 2020. Paludiculture on former bog grassland: Profitability of Sphagnum farming in North West Germany. *Mires and Peat*, 26(8): 1–18. https://doi.org/10.19189/MaP.2019.SNPG.StA.1768

Wichtmann, W., Schröder, C. & Joosten, H. (eds.) 2016. *Paludiculture, Productive Use of Wet Peatlands: Climate Protection, Biodiversity, Regional Economic Benefits*. Schweizerbart Science Publishers, Stuttgart, 272 pp.

Urban soils

Case Study ID	Region	Title	Practice 1	Practice 2	Practice 3	Duration
22	Europe	Carbon storage in soils built from waste for tree plantation in Angers, France	Urban trees			3
23	Europe	Urban agriculture on rooftops in Paris, France - the T4P research project (Pilot Project of Parisian Productive Rooftops)	Urban agriculture			5
24	Europe	Organic amendments for soils rehabilitation of open-pit mines in Spain	Organic amendment			6 to 8
25	Europe	Urban Forestry effects on soil carbon in Leicester, United Kingdom of Great Britain and Northern Ireland	Urban forestry			20 to 100
26	North America	Urban Agriculture in Tacoma, Washington, United States of America	Urban agriculture			2
27	North America	Soil Organic Carbon in forested and non-forested urban plots in the Chicagoland Region, United States of America	Urban forestry			Various

Case Study ID	Region	Title	Practice 1	Practice 2	Practice 3	Duration
28	North America	Compost application to restore post-disturbance soil health in Montgomery County, Virginia, United States of America	Gardens, parks and lawns			4
29	North America	Management of ornamental lawns and athletic fields in California, United States of America	Gardens, parks and lawns			2 to 33
30	North America	Water and residues management on a golf course, Nebraska, United States of America	Gardens, parks and lawns	Mulching	Adapted irrigation	4
31	North America	Maintenance of Marshlands in Urban Tidal Wetlands in New York City, United States of America	Gardens, parks and lawns	Mulching	Adapted irrigation	4

22. Carbon storage in soils built from waste for tree plantation in Angers, France

Patrice Cannavo, Laure Vidal-Beaudet

EPHOR, l'institut Agro, IRSTV, Angers, France

1. Related practices and hot-spot

Urban infrastructures, Sewage sludge additions, Compost additions; Technosols

2. Description of the case study

Soils in urban areas are generally not very fertile, which limits plant growth (Morel, Schwartz and Florentin, 2005). In many cases, trees are installed in planting pits with limited volume, which constrains root development (Lindsey and Bassuk, 1991). These (artificial) soils are subject to compaction due to human activity, which reduces water and air circulation. In addition, the management of chemical fertilization is problematic because nutrient inputs are technically difficult after tree planting. Thus the physico-chemical properties of urban soils can limit tree growth.

This project aimed to create fertile soils by using waste and by-products generated by human activity instead of natural soil from arable land, which, in France has become a limited resource. At the same time, the humans are waste producers and in 2012, waste production in France represented 344 million tonnes, of which only 64 percent is recycled (ADEME, 2014). This case study was carried out within the framework of the SITERRE project (2011-16, ADEME funding).

In order to preserve agricultural land, it is necessary to better recycle the waste produced by mankind and to build soils that can mimic the functions of natural soil as a support material for planting. Within the framework of the project, 11 waste types (5 organic and 6 mineral) were selected from those present in the European waste catalogue (European commission N° 94/3/EEC, 1993). As the basic element of the soil is the aggregate, it was necessary to mix these wastes in order to promote soil structuring. Mineral and organic wastes were therefore mixed in order to select the best ones to build fertile Technosols for the planting of street trees. In this study, we limited ourselves to mixtures of three materials. We have studied the ability of trees to grow in these mixtures.

3. Context of the case study

The study took place at Agrocampus Ouest (47°28′ N, 00°36′ W), Angers (France) between 2013 and 2016. The climate is warm temperate dry, with an average annual temperature of 12 °C and an average annual rainfall of 650 mm. The experimental set-up is detailed in Cannavo, Vidal-Beaudet and Grosbellet (2018).

In 2013, 9 lysimeters of 0.490 m^3 each were built to imitate a planting pit (reduced to a scale of 1/3) and these lysimeters were all planted with Norway maple (*Acer platanoïdes*). Each of them was made of two materials: a growing material (GM) favouring root recovery and common to all the lysimeters, and a structural material (SM) ensuring resistance to soil compaction and good infiltration. GM was a mixture of brick and green waste compost occupying a volume of 25 L in the centre of each lysimeter (cylindrical shape 36 cm diameter, 25 cm depth), while SM occupied the rest of the lysimeter (475 L). Three SMs of earth-stone mixtures containing about 70 percent stones (calibre 40-80 mm) by volume, were studied: (1) the control mixture used routinely by the city of Angers for 30 years (noted SM-CT), (2) a mixture of excavated soil, track ballast, and sewage plant sludge (noted SM-TB/SS) and (3) a mixture of acidic excavated soil, building demolition waste, and green waste (noted SM-DR/GW). All material properties are presented in Table 79. The study included three replicates for each structural material. The initial dry bulk densities were 0.49, 1.71, 1.68 and 1.38 g/cm^3 for GM, SM-CT, SM-TB/SS and SM-DR/GW, respectively. Each year, a repetition of each treatment was destructively sampled (2014, 2015 and 2016). Root and aboveground biomasses and architecture of the trees were measured. Soils were analysed (Total organic carbon, total N, water pH, CEC, CaO, K_2O, Na_2O, MgO, and Olsen P).

Table 79. Origin and composition of the materials used in this study (from Yilmaz *et al.* 2018), and soil granulometry (Cannavo, Vidal-Beaudet and Grosbellet, 2018)

Treatments	Fine mineral material	Coarse mineral material		Organic material		Soil granulometry (g/kg)		
	Weight ratio	Origin	Weight ratio	Origin	Weight ratio	Clay	Silt	Sand
GM	0	Brick manufacturing	0.58	Sewage sludge + green waste compost	0.42	118	399	482
SM-CT	0.27*	Chalcedony	0.70	Leaf litter	0.03	167	396	437
SM-TB/SS	0.20	Track Ballast	0.76	Sewage Sludge	0.04	111	383	507
SM-DR/GW	0.22	Demolition Rubble	0.75	Green Waste	0.03	107	338	555

GM: growing material, SM: structural material, CT: control, TB/SS: track ballast/sewage sludge, DR/GW: demolition rubble, green waste; * SM-CT was made of arable soil (A layer), and the other two SMs were made of subsoil.

4. Possibility of scaling up

The practice can be applied in different pedoclimatic contexts all over the world. For the SITERRE program, the selected wastes are present all over France, and a bibliographical study has been carried out to verify that these wastes have very few recovery routes. Tree-planting pits are present in all cities and particularly in the hyper-centre where the contribution of impervious soil surface is the highest. The study was carried out on a small scale (1:3) over a period of 3 years and can be very easily extrapolated to the longer term and in larger pits. The practice will soon be conducted on a larger scale, thanks to pilot sites installed in different French cities (SITERRE II project, 2020-23).

5. Impact on soil organic carbon stocks

Table 80. Evolution of SOC stocks in the 3-year study

Location	Climate zone	Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha/yr)	Depth (cm)	More information	Reference
Angers, France	Temperate	Technosol GM + SM-CT	7.9	50.2	57	We suppose that the baseline C stock corresponds to that of a sealed soil, which will be replaced by a Technosol	Cannavo, Vidal-Beaudet and Grosbellet (2018)
		GM + SM-TB/SS	0	52.2			
		GM + SM-DR/GW	0	35.6			

GM: Growing Material, SM: Structural Material, CT: Control, TB/SS: Track Ballast/Sewage sludge, DR/GW: Demolition Rubble, Green Waste

6. Other benefits of the practice

6.1. Benefits for soil properties

During the 3 years of experimentation, the physico-chemical properties of the soils were monitored. Concerning the physical properties, the soil water reservoir is lower than the standard expected for an urban soil (ie, 0.1 mm water cm^{-1} soil; Huot *et al.*, 2017), due to a large number of stones (70 percent by volume). Nevertheless, lysimeters on a larger scale (2 m^3) can have a water autonomy of 25 to 30 days in warm temperate dry climate (Cannavo, Vidal-Beaudet and Grosbellet, (2018)). These Technosols benefit from good aeration conditions (macroporosity greater than 0.2 v/v).

Concerning the chemical properties, macro and microelement analyses show that the mixed wastes has neither excess nor deficiency in these elements.

6.2 Minimization of threats to soil functions

Table 81. Soil threats

Soil threats	
Nutrient imbalance and cycles	The organic matter in these Technosols guarantees a satisfactory bioavailability of the elements and correct plant nutrition.
Soil salinization and alkalinization	Salinity tends to decrease along years thanks to salt leaching.
Soil biodiversity loss	We were able to observe the presence of earthworms and anthills after 1 year of experimentation, demonstrating the capacity of these Technosols to host macrofauna. Similarly, an important development of weeds was observed.
Soil sealing	The main function of these Technosols is to support vegetation, thus making it possible to fight against soil sealing.
Soil compaction	The structural material contains 70 percent of stone, ensuring a resilience to mechanical stresses applied to the soil and preventing compaction.

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

The main service provided is well-being and the living environment, as these Technosols will be used as a support for ornamental plants. The production of food plants is not possible as wastes are reused.

6.4 Mitigation of and adaptation to climate change

This case study focused on the ability of soils to produce ornamental species such as trees. Street trees play an essential role in the cooling of cities during hot weather. They store carbon in their biomass and also reduce energy consumption for air-conditioning system.

GHG measurements were not performed. Nevertheless, tree planting pits in cities are intended to contribute to urban greening thus introducing plants that contribute to carbon sequestration. Moreover, these soils were constructed by adding organic waste, thus enhancing the carbon sequestration capacity.

6.5 Socio-economic benefits

There are two socio-economic benefits of this case study:

- The improvement of the living environment of the inhabitants and the feeling of well-being, linked to the greening of cities. In addition, urban greening contributes to the fight against the urban heat island phenomenon and the associated health risks.
- The attractiveness of cities through greening is enhanced. Many labels exist in France, including the "flowery towns and villages" label. At the level of districts and buildings, green spaces contribute to embellishing these structures.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 82. Soil threats

Soil threats	
Soil salinization and alkalinization	The electrical conductivity of soils has shown a risk of low to moderate salinity, especially in the mixture from demolition waste and green waste.
Soil contamination / pollution	The pure materials were analysed for pollutants (metallic trace elements, PAH and PCB). High levels of Pb, Zn, Cu, Cd and Cr were found in sewage sludge and track ballast). There is no PCB contamination, only the track ballast has high levels of certain PAHs (pyrene, fluoranthene). However, once mixed, the levels remained below the current standards.
Soil acidification	Acidification is possible if organic amendment is not supplied
Soil compaction	Possibility of GHG emissions due to anoxic conditions more favorable in case of compacted soil.
Soil water management	The soil water reservoir is reduced because of the stones. Irrigation is generally necessary for 3 years after planting. Then the trees are self-sustaining.

7.2 Increases in greenhouse gas emissions

It is difficult to quantify possible emissions. It is likely that CO_2 and N_2O emissions will occur during the first year of tree installation. We have indeed carried out other experiments in the past in soils receiving between 20 and 40 percent of organic matter by volume. The organic matter content decreases significantly in the first year and stabilizes in the following years (Cannavo, Vidal-Beaudet and Grosbellet, 2018); Cannavo *et al.,* 2018). This biodegradation of organic matter has been accompanied by CO_2 emissions, while N_2O emissions are possible if anoxic conditions prevail. This remains very likely if soils suffer from compaction.

8. Recommendations before implementing the practice

It is important to analyse the physico-chemical properties of pure materials before producing mixtures. In particular, attention must be paid to the pH of the materials in order to guarantee a mixture whose pH is adequate to the needs of the plants (i.e. between 6.5 and 7.5). Thus, the choice of materials and their mixing proportions must be considered in order to guarantee agronomic properties favourable to plant growth. This study was realised in reduced-size pits (0.5-m^3). In cities, the volume of street tree pits filled with structural material is on average 6–10 times higher. With such a volume, we estimated that the soil's nutrient reserve autonomy would be at least 20 years, and 3–4 weeks for water supply (Cannavo *et al.*, 2018). Finally, the handling of waste is subject to regulations depending on each country.

9. Potential barriers for adoption

Table 83. Potential barriers to adoption

Barrier	YES/NO	
Cultural / Social	Yes	The public may be reluctant to know that they are near soils constructed from waste. Communication to citizens is important.
Economic	No	Waste must be locally present to reduce transport costs.
Legal (Right to soil)	Yes	Waste is subject to strict regulations; its use for soil construction must comply with the legislation in force.
Knowledge	Yes	Longer-term trials need to be established through pilot sites within cities to verify health and environmental safety and to confirm that plant growth is possible.

Photos

Photo 53. Experimental set up at AGROCAMPUS OUEST Angers (France), one year after planting. The 9 lysimeters are on the ground to facilitate their dismantling and sampling. Bales of straw were placed around them to insulate the lysimeters thermally and reproduce

Photo 54. Tree root development 3 years after planting (2016) in a Technosol composed of excavated earth, track ballast, and sewage sludge, AGROCAMPUS OUEST

References

ADEME, 2014. *Déchets Edition 2014, Chiffres clés*. Ademe Editions. (also available at: https://www.ademe.fr/sites/default/files/assets/documents/chiffres-cles-dechets-edition-2014-8147.pdf)

Cambou, A. 2018. *Evaluation du stock et de la stabilité du carbone organique dans les sols urbains*. Agrocampus Ouest, France (PhD dissertation). https://tel.archives-ouvertes.fr/tel-02088714/document

Cannavo, P., Vidal-Beaudet, L. & Grosbellet, C. 2014. Prediction of long-term sustainability of constructed urban soil: impact of high amounts of organic matter on soil physical properties and water transfer. *Soil Use and Management*, 30(2): 272–284. https://doi.org/10.1111/sum.12112

Cannavo, P., Guénon, R., Galopin, G. & Vidal-Beaudet, L. 2018. Technosols made with various urban wastes showed contrasted performance for tree development during a 3-year experiment. *Environmental Earth Sciences*, 77(18): 650. https://doi.org/10.1007/s12665-018-7848-x

Lindsey, P. & Bassuk, N., 1991. Specifying Soil Volumes to Meet the Water Needs of Mature Urban Street Trees and Trees in Containers. *Journal of Arboriculture*, 17: 141-149.

Morel, J.L., Schwartz, C. & Florentin, L., 2005. Urban soils. *In Encyclopedia of Soils in the Environment*. p.202-208.

Huot, H., Séré, G., Vidal-Beaudet, L., Leguédois, S., Schwartz, C., Watteau, F., Morel, J.L. 2017. Pedogenic processes in soils of urban, industrial, traffic, mining and military areas. *In* Levin, M.J., Kim, K.-H.J., Morel, J.L., Burghardt, W., Charzynski, P., Shaw, R.K. (Eds.) *Soils within Cities. Global approaches to their sustainable management – composition, properties, and functions of soils of the urban environment*. Edited on behalf of IUSS Working Group SUITMA, Catena-Schweizerbart, Stuttgart. (also available at https://hal.archives-ouvertes.fr/hal-01595163).

23. Urban agriculture on rooftops in Paris, France - the T4P research project (*Pilot Project of Parisian Productive Rooftops*)

Baptiste J-P. Grard[1], Nastaran Manouchehri[2], Sophie Joimel[1], Christine Aubry[3], Claire Chenu[1]

[1]Université Paris-Saclay, INRAE, AgroParisTech, UMR ECOSYS, Thiverval-Grignon, France

[2]Université Paris-Saclay, INRAE, AgroParisTech, UMR SayFood, Paris, France

[3]Université Paris-Saclay, INRAE, AgroParisTech, UMR SAD-APT, Université Paris-Saclay, Paris, France

1. Related practices and hot-spot

Green roof, urban agriculture, compost additions, crop rotations, adequate irrigation practices; Technosols

2. Description of the case study

In the last decade, urban agriculture has been a growing topic for urban stakeholders worldwide (Specht *et al.*, 2013). Considering the negative effects of urban growth and development (soil sealing, habitat fragmentation, pollution etc. ; Artmann and Sartison, 2018), urban agriculture is perceived as one way to counteract some of them (for instance through water retention, food production and biodiversity). The scarcity of available space in cities leads to consider underused urban spaces such as rooftops, which implies a soil construction through the use of different materials. In this case, the constructed soil can be referred to as Isolatic Technosol. Constructed Technosols for productive rooftops are aimed to be used as growing media for several years and have to meet specific technical requirements (e.g. related to the load capacity of the roof) in addition to the expected function of supporting plant growth.

Since 2012, the Pilot Project of Parisian Productive Rooftops (T4P) aims to study productive systems, i.e. producing edible biomass, on rooftops, based on the only use of urban wastes as Technosol component. Different experimental trials address the potential for food production, ecosystem services delivery and the temporal evolution of the Technosols. The use of local materials, such as urban organic waste to build these soils, offers multiple advantages:

1. it avoids the consumption of non-renewable resources such as peat or the transport of rural soils to cities;
2. it avoids the costs incurred and the harmful greenhouse gases generated by the transport and treatment of organic waste;
3. contribution to circular economy with an associated benefit gained from the nutrients contained in organic wastes, thereby reducing the consumption use of mineral fertilizers; and
4. the materials is usually lightweight. Potential disadvantages include substrate shrinkage, nutrient loss through storm water runoff, and carbon dioxide emissions through substrate mineralization.

The study of constructed Technosol and the evaluation of their potential for ecosystem services delivery involved a series of experimental trials (4 since 2012) using wooden boxes (<1 m^2) with different Technosol composition and vegetable species. Five urban wastes were used as Technosol components: green waste and biowaste compost, spent mushroom substrate, crushed wood from tree pruning and crushed tiles and bricks.

3. Context of the case study

Since 2012, the research project T4P (AgroParisTech, 2020) takes place on the rooftop of AgroParisTech (French Technical University of Agronomy; coordinates: 48°50′24.4" N, 2°20′54.5" E). The experimental roof is under a typical Marine West Coast Climate (Cfb–Köppen climate classification) in a housing neighborhood where this roof is the highest in a perimeter of 200 m.

Wooden containers, classically utilized as a backyard composters, were used as experimental units. These are easy to handle and allow to distribute at best the additional weight on the roof.

The constructed Technosols were inspired by an original American gardening practice now popularized in France: *lasagna* beds (Collaert, 2010). The idea is to mimic a natural soil by superimposing layers of "brown" and "green" organic matter. These terms relate to the decomposability of the organic matter. Typical constituent of a brown layer is crushed wood having a low rate of mineralization and providing an input of carbon with a high C/N ratio. The green layer can be made of green waste compost with a high mineralization rate and a low C/N ratio. In our case, every year, at the beginning of the cropping season, we added an additional layer of "green" organic matter. This ensured an input of organic matter providing nutrients by mineralization to compensate for that used by the previous crop and ensured a sufficient volume for root anchorage.

Three types of urban organic by-products were used as parent materials (i.e. material used to build the Technosol) of the Technosols:

- Green waste compost, from urban public parks and green spaces provided by a local company (®BioYvelinesServices).
- Crushed wood, made of pruning urban trees provided by the same company.
- Spent mushroom substrate, based on coffee grounds collected in Paris. Used coffee grounds from Paris cafés are utilized to produce the mushroom Pleurotus Ostreatus (®La boîte à champignons).
- Woodchips (0–40mm) served as a mulch in each wooden container (a layer of 3cm), in order to minimize substrate evaporation and weeds.

Each plot was organized identically (Figure 10): it was filled at the bottom with 5 cm of expanded clay pebbles used as a water reserve and surrounded by an EPDM (ethylene–propylene–diene monomer) membrane. On the top of this we placed 30 cm of growing substrate surrounded by a "geotextile" through which the roots could grow (called "filter layer" in the Figure 11).

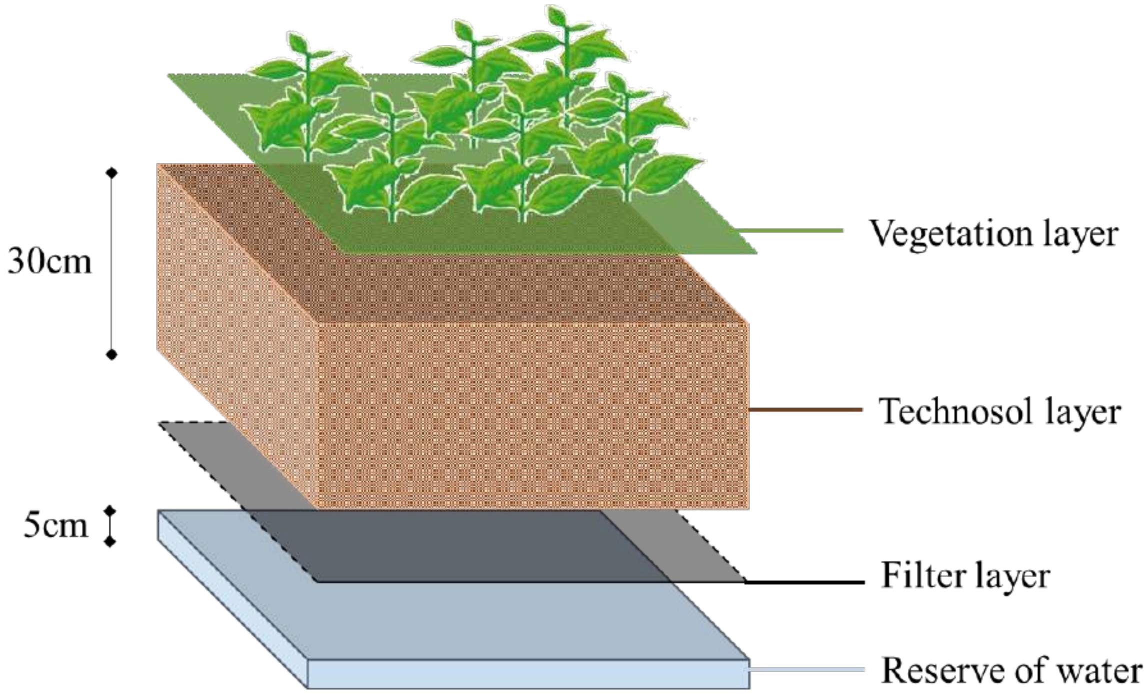


Figure 10. Vertical organization of an experimental plot

The experimental system presented here consisted of 9 wooden boxes of 0.64m^2 (80*80cm) each, with 0.5 m between them. In order to test the effects of Technosols nature and organization on food production and pro ecosystem services, three different modalities (Figure 11) were set up in March 2012 (experiment n°1), each with three replicates:

- Lasagna (**L**): a 15 cm layer of green waste compost (called the upper layer – Table 84) above a 15 cm layer of crushed wood (called the lower layer – Table 84).
- Lasagna with residues of spent mushroom substrate (**L-R**): a 12.5 cm layer of green waste compost, over a 5 cm layer of spent mushroom substrate and a 12.5 cm layer of crushed wood.
- A mix (**M**): 30 cm of green waste compost and crushed wood mixture (50/50 v/v).

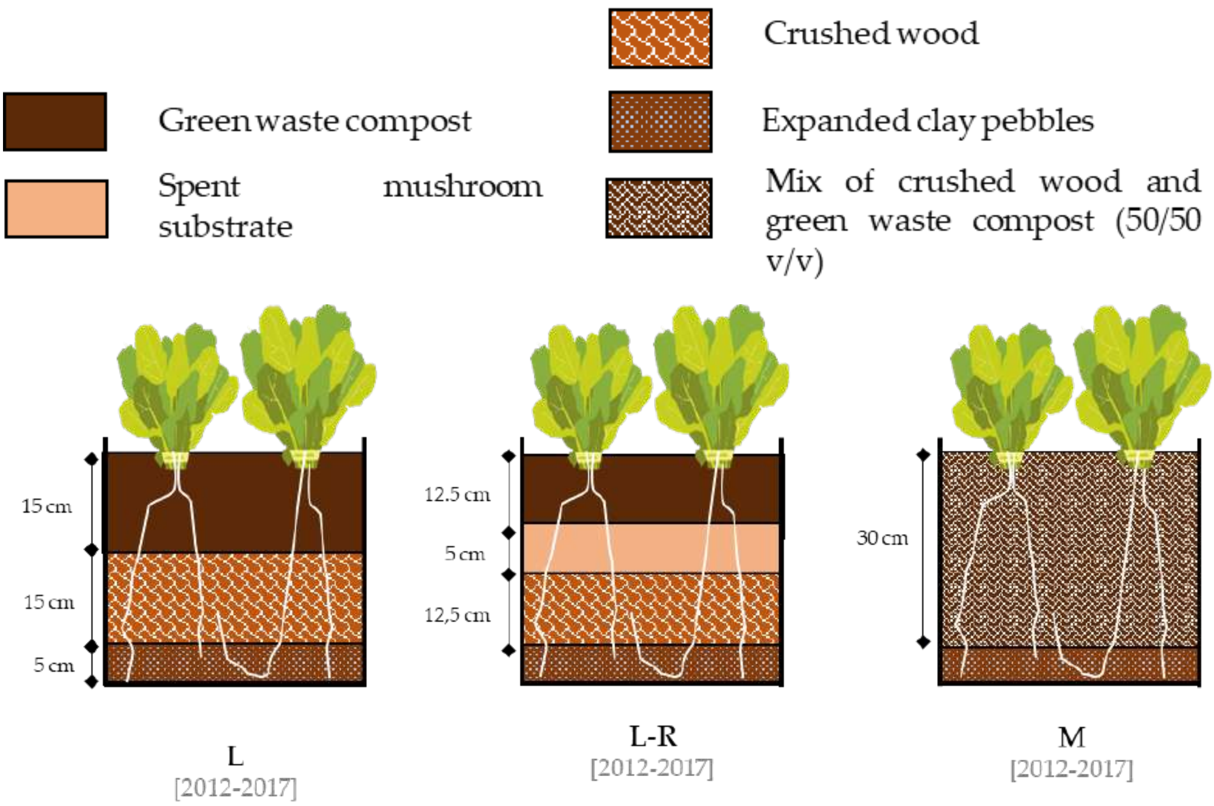


Figure 11. Scheme of the different experimental modalities

Crop rotation

Diversified crop rotations were set to maximize the available space both vertically and horizontally and associate crops whenever possible. More than 10 vegetable species were grown in the experimental units: tomatoes, basil, lettuce, radish etc. The same rotation/crops was applied between replicates and within an experimental unit.

Irrigation

The experimental plots were both rainfed and irrigated with tap water. Tap water was distributed thanks to a drip irrigation system to minimize the loss due to evaporation. Water consumption was measured with water meters.

Farm inputs

No mineral fertilizers were used, organic fertilization being ensured by the organic material added at the beginning of each cropping season (March or April) when the containers were re-filled to their initial height of 30 cm, to compensate for substrate compaction and biodegradation. As an illustration of the input here is the description for the modality L – the input is the green waste compost: 1st year = 78l/plot; 2nd year = 63l/plot ; 3rd year = 46l/plot; 4th year = 23l/plot and 5th year = 56l/plot.

4. Possibility of scaling up

The experiment has been conducted under an Oceanic temperate climate but can be replicated under other climatic conditions. Urban waste quality depends on source and process and may vary in other locations thus affecting the results; nevertheless, scaling up is possible and is encouraged, using local organic resources.

The main goal of the project is not to give a "ready-to-use recipe" of Technosol based on urban wastes but to inspire and help practitioners and other researchers that want to design, experiment and use this type of growing system. As an example two companies were created in France inspired by the T4P project. Both firms (Topager (https://topager.com) and Cultures en ville (Cultures en Ville, 2021)) do install productive rooftops with Technosols that are mainly or exclusively based on urban waste. Yet, the T4P experiment has only tested a limited combination of Technosol layouts and materials and many further possibilities remain, based on urban waste local availability and quality.

5. Impact on soil organic carbon stocks

To determine the organic content and stock of the Technosol, two types of measurements were achieved:

- Technosol height was recorded at the beginning of each growing season using a simple system of level difference between the upper and the lower point of the box and the Technosol surface. Ten measurements were recorded along the diagonal of each plot using a graduated stick.

- Parent materials were analyzed at different times as well as the Technosols. The upper (10/15cm) and lower horizon (10/15cm) of Technosols were sampled after five years using soil core samplers. A quantity of 500 g of soil samples were used for agronomic and pollutant analyses and were dried at 40 °C and crushed to pass at 2 mm sieve prior to analysis. Organic carbon content was measured by the soil laboratory of INRA by dry combustion (by heating at 1 000 °C with O2– [NF ISO 10694]). On the field, bulk density was measured according to NF EN 13041.

Organic carbon stock was then calculated using the dry bulk density, the organic C content and the volume of Technosol.

Table 84. Evolution of SOC stocks in the 5-year study of urban farming on a rooftop in Paris, France

Grard et al. (2020).

Climate is Marine West coast according to the Köppen classification and the Technosol is classified as Isolatic Technosol, according to the World Reference Base, WRB (IUSS Working Group WRB, 2015).

Data from parent material: green waste compost (bulk density = 0.2 ± 0.02g.cm^{-3} ; organic carbon = 230 ± 22.6 g/kg) ; spent mushroom substrate (bulk density = 0.1 ± 0.02 g/cm^3 ; organic carbon = 415 ± 42.9 g/kg) and crushed wood (bulk density = 0.1 ± 0.04 g/cm^3 ; organic carbon = 454.3 ± 5.7 g/kg).

Treatment	Height	Baseline organic C stock (tCorg/ha)	Bulk density – 5th year (g/cm^3)	Organic carbon – 5th year (g/kg)	Estimated organic C stock after 5 years (tCorg/ha)	Duration(Years)	Depth (cm)	More information	Reference
L	Upper layer	175	0.4 ± 0.01	184.3 ± 21.5	178	5	~0–30	Baseline of organic C corresponds here to the stock of organic C when the Technosol is installed on the roof.	Grard *et al.* (2020)
L	Lower layer	175	0.4 ± 0.04	213.7 ± 24	178	5	~0–30	Baseline of organic C corresponds here to the stock of organic C when the Technosol is installed on the roof.	Grard *et al.* (2020)
L-R	Upper layer	197	0.3 ± 0.1	200.3 ± 17.3	164	5	~0–30	Baseline of organic C corresponds here to the stock of organic C when the Technosol is installed on the roof.	Grard *et al.* (2020)
L-R	Lower layer	197	0.3 ± 0.03	242.7 ± 11	164	5	~0–30	Baseline of organic C corresponds here to the stock of organic C when the Technosol is installed on the roof.	Grard *et al.* (2020)
M	Upper layer	175	0.2 ± 0.04	260.7 ± 10.1	153	5	~0–30	Baseline of organic C corresponds here to the stock of organic C when the Technosol is installed on the roof.	Grard *et al.* (2020)
M	Lower layer	175	0.2 ± 0.1	238 ± 28.6	153	5	~0–30	Baseline of organic C corresponds here to the stock of organic C when the Technosol is installed on the roof.	Grard *et al.* (2020)

After five years, we observe that bulk density of the Technosols was higher than that of the parent materials, showing that the observed subsidence was associated with Technosol compaction. This evolution coupled with biodegradation occurring in the Technosol results over time to a steadily decreasing of Corg stocks (between +2 percent to -15 percent depending on the Technosol composition; Table 84 and/or Grard *et al.*, 2020). The yearly input of parental material (as described above) explain for a part this evolution resulting in a yearly (but decreasing) input of organic matter.

6. Other benefits of the practice

6.1. Benefits for soil properties

Technosols physico-chemical properties were monitored all along the 5 years of experiment. The thickness of the Technosols decreased strongly in the first year, with an average subsidence (gradual lowering of the surface elevation) of 39 percent for all treatments. The Technosols thickness remained between 8 percent to 21 percent of the initial height in the following years. Bulk density after five years was higher than that of the parent materials, showing that the observed subsidence was associated with Technosol compaction. The increase in bulk density resulted in a slight decrease of total porosity that remained between 30 percent to 40 percent of the total volume. Overall, the Technosols were very fertile, exhibiting a large porosity, much larger than in mineral soils used for market gardening, with a neutral to slightly alkaline pH (Grard *et al.*, 2020). Available contents of P and K were much higher than the P and K contents above which Department of Environment Food and Rural Affaires (DEFRA) Fertilizer manual (Artmann and Sartison, 2018) recommend not to fertilize vegetable crops. Regarding nitrogen, the contents of both total N and mineral N were very high in the Technosols compared to mineral soils of urban vegetable gardens (Joimel *et al.*, 2016).

In the lasagna bed system, the lower layers showed the strongest evolution with increases in C and N contents, CEC and nutrients over time (Grard *et al.*, 2020). This can be explained by biodegradation, i.e. a fast decomposition of the crushed wood as well as by transfer of upper layers particles or solutes downwards, by gravity, by leaching and/or by the earthworm's activity. Illuviation and/or lixiviation are hence taking place in the profile of the constructed Technosols. As for other Technosols, similar pedogenesis processes to those occurring in natural soils were observed (Séré *et al.*, 2010), that is., biodegradation and illuviation although at a higher rate than in natural soils. In the study, the high content of organic matter, available water (irrigation) and available oxygen (large porosity of the Technosol) may explain the high intensity of processes such as biodegradation. Mesofauna were found to be abundant and diverse, which also explains the high rates of biodegradation (Joimel *et al.*, 2018).

6.2 Minimization of threats to soil functions

Table 85. Soil threats

Soil threats	
Nutrient imbalance and cycles	High fertility of the Technosol which delivered enough nutrients to allow for edible biomass production.
Soil contamination / pollution	All urban wastes used respected existing norms regarding contaminants (NF U 44-551). Over the 5 years of the experiment the trace metal content of Technosols did not increase except for Zn (Grard *et al.*, 2015, 2020).
Soil biodiversity loss	The Technosols hosted larger and more diverse communities of mesofauna (and more specifically of Collembola) than agricultural soils in the same region (Joimel *et al.*, 2018).
Soil sealing	Technosols are developed on sealed soil. They do not mitigate the sealing, but some of its effects, allowing infiltration and retention of rainwater and hosting biodiversity.
Soil compaction	The use of coarse material such as crushed wood ensures a low dry bulk density (0.2–0.4 g/cm3). The measurements made in this system showed that soil compaction was not an issue.
Soil water management	Technosols exhibited large available water content in (18–31 gwater/gdrysoil), explained by their high organic matter content.

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

A relatively high productivity per square meter in all treatments was observed (11.5 ± 5.3 kg/m^2 of biomass on average for all treatments during 5 years ; Grard *et al.*, 2020) as compared to average yields in community gardening (1.2–2.6 kg/m^2), in horticulture in northern countries (2.5–3.3 kg/m^2) and in professional and intensive gardening (5.4–7.1 kg/m^2; Weidner, Yang and Hamm, 2019). The measured yields were close to those of other rooftop studies with Technosols made of organic materials (e.g. Orsini *et al.*, 2014). Nevertheless, caution should be taken when comparing yields of different vegetables and in different area of the world due to climate differences, different water content of vegetables, different vegetables varieties, different Technosol composition as well as differences in the surface area considered. In rooftop farming, a minimum surface area needs to be dedicated to other uses than food production, e.g. pathways. Here, yields were expressed per productive surface area, while Orsini *et al.*, (2014) consider that only 65 percent of the roof surface could be dedicated to food production. In this specific study case, if this percentage of 65 percent is used, the average yield of L-R would, during the five growing seasons, decrease from 14.7 kg/m^2 to 9.6 kg/m^2 at roof level. Caution should therefore be taken to clearly define the surface area considered. In addition, other criteria could be taken into account when comparing food production in different urban agriculture systems, such as the size of the vegetables, marketable yield or nutritional quality. Such parameters should be considered in future studies on urban agriculture.

Regarding food production and Technosols, while the nature of the parent materials had little effect on yields, layered Technosols (L and L-R modalities) allowed for significantly higher yields than homogeneous ones, in which compost and crushed wood had been mixed (M modality ; Grard *et al.*, 2020). The results do not allow to explain clearly this effect, but two hypothesis can be made, that are not exclusive: (i) a lack of nitrogen when compost and crushed wood are mixed; (ii) different availabilities of air and water in the Technosol due to the initial layering of parent material that evolve across time.

6.4 Mitigation of and adaptation to climate change

GHGs (except for carbon) were not directly measured in this experiment. Nevertheless, it should be noted that the use (and valorization) of urban organic wastes can avoid their burning and transport allowing GHG emission savings meanwhile contributing to large organic C stocks in Technosols. Using life cycle assessment we demonstrated the lower impact of using urban organic waste as component of Technosol rather than potting soil as classically used (Dorr *et al.*, 2017).

6.5 Socio-economic benefits

Socio-economic benefits were not directly measured from the experiment as it was not a direct goal of the research project. Nevertheless, thanks to the project we acknowledge different (direct or incidental) socio-economic benefits:

- The creation of two firms from Topager (2012) and Cultures en ville (2021) that are nowadays working at the installation of productive rooftop with similar growing system than those developed in our experiment (trying to re-use urban waste as Technosol component). Both firms were created in link with the project and by people involved in it.

- The project has also tried to raise public awareness regarding the topic of urban farming. In average each year, around 1000 people are visiting our research experimental rooftop. Mainly students but also journalists and general public interested by the topic of urban agriculture are coming. This public interest for the topic could also be illustrated through the different demand that we face for grey literature publication to raise the awareness of the public on the topic: TEDx Talks (2019), a media article (Baptiste and Mathieu, 2018), etc.

- The idea to re-use waste from cities to design fertile Technosol in more circular and local economy has a great economic development potential regarding the volume of natural and nonrenewable material currently used in cities to design green space (Vidal-Beaudet, 2018). This was illustrated by the development of the two firms and by the interest of many public entities for the experiment.

6.6 Other benefits of the practice

Cities face many challenges regarding water management. Productive rooftops are an opportunity to store rainwater and delay their flowing in the sewage system. Over two years, we quantified the rainwater retention capacity of the Technosols to be 74 to 81 percent of incident rainwater (Grard *et al.*, 2018). Specific attention should be given to precise management of irrigation, not to hamper this storage capacity (Harada *et al.*, 2018).

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 86. Soil threats

Soil threats	
Nutrient imbalance and cycles	This growing system is based on the recycling of organic wastes and the nutrients they contain. Attention should be given to biodegradation, so that enough nutrients are available for plant growth without leaching of excess nutrients.
Soil contamination / pollution	The trace metal content of Technosols did not significantly increase during the 5 years of experimentation, except for Zinc which increased by 15 percent of its initial value and may be explained by atmospheric fallout.
Soil compaction	Soil compaction is not an issue in such organic wastes made Technosols.
Soil water management	Irrigation should be managed carefully to satisfy plants needs while allowing the Technosols to be able to store rainwater.

7.2 Increases in greenhouse gas emissions

NA

7.3 Conflict with other practice(s)

Rooftops could be used for multiple purposed: green roofs (productive of edible biomass or not), solar panels, water retention (exclusively), supporting technical equipments etc. These different possible uses need to be compared carefully at the beginning of a project regarding their environmental and economic costs, the ecosystem services they provide as well as the local inhabitants demand regarding this space.

7.4 Negative impact on production

Urban environment present different types of soil, air and water contamination. This pollution sources could affect the growing systems and more specifically the quality of edible production. In terms of vegetables quality, the results obtained in this study confirmed generally the well-known characteristic of plant-specific contamination dealing with higher levels of trace metals in root and leafy vegetables compared to fruit vegetables (Grard *et al.*, 2015, 2020; Rahmanian *et al.*, 2016). The obtained results are in agreement with other studies showing low level of trace metals in produces growing on organic substrates with natural trace metal occurrence (Pennisi *et al.*, 2016). All vegetables respect the European norms for regulated trace metals (Cd and Pb)[7]. Certainly, the quality of urban vegetables remains a public health concern and needs to be further investigated for a wider range of vegetable species and also for other pollutants present in urban environments such as polycyclic aromatic hydrocarbons.

8. Recommendations before implementing the practice

We showed that by-products and urban organic wastes such as crushed wood, green waste composts and spent mushroom substrate were valuable substrates to build highly fertile Technosols leading to acceptable and even high yields per square meter of vegetables respecting the European norms for trace metals (e.g. Cd and Pb ; Grard *et al.*, 2015, 2020).

Before any implementation of a productive rooftop, the roof characteristics (load, access and security) should be carefully studied. The weight of type of growing system we propose here is 350 to 500 kg/m^2 when water saturated. Beside, the design of the productive rooftop should be achieved paying attention to the locally available organic resources and with a necessary characterization of their quality especially regarding contaminants. The combination of different urban organic wastes to construct Technosols should valorize their complementary towards the functions expected for plant development, i.e provision of water, air and nutrients.

[7] Commission Regulation E.C., No 1881/2006) (https://eur-lex.europa.eu/legal-content/FR/TXT/?uri=CELEX:02006R1881-20180319).

9. Potential barriers for adoption

Table 87. Potential barriers to adoption

Barrier	YES/NO	
Biophysical/ Natural resource	No	It is recommended to use local resources from urban organic wastes (Dorr *et al.*, 2017; Grard *et al.*, 2015, 2018, 2020).
Cultural / Social	Yes	The reasons for using the selected urban wastes should be clearly communicated to avoid a reject of this type of use by citizens.
Economic	No	Local wastes should be used to reduce transportation cost.
Legal (Right to soil)	Yes	Specific regulations applies to the use of wastes and some of them are not allowed to be directly used.
Knowledge	Yes	“New” or local type of urban organic waste should be documented in order to be properly used.

Photos

Photo 55. Crushed wood (left) and green waste compost (right) used in the experiment

Photo 56. AgroParisTech's experimental rooftop in November 2015 (top) and in April 2017 (bottom)

Photo 57. Overview of the experiment on the roof of AgroParisTech in January 2016

Photo 58. A Technosol profile of the experiment after 7 years of growing

References

Artmann, M. & Sartison, K. 2018. The Role of Urban Agriculture as a Nature-Based Solution: A Review for Developing a Systemic Assessment Framework. *Sustainability*, 10(6): 1937. https://doi.org/10.3390/su10061937

AgroParisTech. 2020. T4P: an innovative research project for productive Parisian roofs. In : *AgroParisTech* [onlin]. Paris. [Cited 18 August 2021]. http://www2.agroparistech.fr/T4P-un-Projet-de-recherche-innovant-pour-des-Toits-Parisiens-Productifs.html

Baptiste, G & Mathieu, U. 2018. BD : Sur le toit, des légumes et de la science. In: *The Conversation*, 26 November 2018. (also available at https://theconversation.com/bd-sur-le-toit-des-legumes-et-de-la-science-107386)

Collaert, J.-P. 2010. *L'art du jardin en lasagnes*. Edisud edition. 144 pp.

Cultures en Ville. 2021. *Cultures en Ville* [online]. Cachan, France [Cited 18 August 2021]. https://www.culturesenville.fr/

Dorr, E., Sanyé-Mengual, E., Gabrielle, B., Grard, B.J.-P. & Aubry, C. 2017. Proper selection of substrates and crops enhances the sustainability of Paris rooftop garden. *Agronomy for Sustainable Development*, 37(5): 51. https://doi.org/10.1007/s13593-017-0459-1

Grard, B.J.P., Bel, N., Marchal, N., Madre, F., Castell, J.F., Cambier, P., Houot, S., Manouchehri, N., Besancon, S., Michel, J.C., Chenu, C., Frascaria-Lacoste, N. & Aubry, C. 2015. Recycling urban waste as possible use for rooftop vegetable garden. *Future of Food: Journal on Food, Agriculture and Society*, 3(1): 21–34. (also available at http://futureoffoodjournal.org/index.php/journal/article/view/141).

Grard, B.J.P., Chenu, C., Manouchehri, N., Houot, S., Frascaria-Lacoste, N. & Aubry, C. 2018. Rooftop farming on urban waste provides many ecosystem services. *Agronomy for Sustainable Development*, 38(1). https://doi.org/10.1007/s13593-017-0474-2

Grard, B.J.P., Manouchehri, N., Aubry, C., Frascaria-Lacoste, N. & Chenu, C. 2020. Potential of Technosols created with urban by-Products for rooftop edible production. *International Journal of Environmental Research and Public Health*, 17(9): 3210. https://doi.org/10.3390/ijerph17093210

Harada, Y., Whitlow, T.H., Todd Walter, M., Bassuk, N.L., Russell-Anelli, J. & Schindelbeck, R.R. 2018. Hydrology of the Brooklyn Grange, an urban rooftop farm. *Urban Ecosystems*, 21: 673-689. https://doi.org/10.1007/s11252-018-0749-7

IUSS Working Group WRB. 2015. FAO - World reference base for soil resources 2014. Update 2015 International soil classification system for naming soils and creating legends for soil maps. *World Soil Resources Reports No. 106 - FAO*, p. 1–191 pp.

Joimel, S., Cortet, J., Jolivet, C.C., Saby, N.P.A., Chenot, E.D., Branchu, P., Consalès, J.N., Lefort, C., Morel, J.L. & Schwartz, C. 2016. Physico-chemical characteristics of topsoil for contrasted forest, agricultural, urban and industrial land uses in France. *Science of The Total Environment*, 545–546: 40–47. https://doi.org/10.1016/j.scitotenv.2015.12.035

Joimel, S., Grard, B., Auclerc, A., Hedde, M., Le Doaré, N., Salmon, S. & Chenu, C. 2018. Are Collembola "flying" onto green roofs? *Ecological Engineering*, 111: 117–124. https://doi.org/10.1016/j.ecoleng.2017.12.002

Orsini, F., Gasperi, D., Marchetti, L., Piovene, C., Draghetti, S., Ramazzotti, S., Bazzocchi, G. & Gianquinto, G. 2014. Exploring the production capacity of rooftop gardens (RTGs) in urban agriculture: the potential impact on food and nutrition security, biodiversity and other ecosystem services in the city of Bologna. *Food Security*: 781–792. https://doi.org/10.1007/s12571-014-0389-6

Pennisi, G., Orsini, F., Gasperi, D., Mancarella, S., Sanoubar, R., Vittori Antisari, L., Vianello, G. & Gianquinto, G. 2016. Soilless system on peat reduce trace metals in urban-grown food: unexpected evidence for a soil origin of plant contamination. *Agronomy for Sustainable Development*, 36(4): 56. https://doi.org/10.1007/s13593-016-0391-9

Rahmanian, M., Daniel, A., Grard, B., Juvin, A., Bosch, A., Aubry, C., Cambier, P. & Manouchehri, N. 2016. Edible production on rooftop gardens in Paris ? Assessment of heavy metal contamination in vegetables growing on recycled organic wastes substrates in 5 experimental roofgardens. In *Proceedings of The IRES 26th International Conference*. Paris, France,30th January 2016. ISBN: 978-93-85973-07-9

Séré, G., Schwartz, C., Ouvrard, S., Renat, J.-C., Watteau, F., Villemin, G. & Morel, J.L. 2010. Early pedogenic evolution of constructed Technosols. *Journal of Soils and Sediments*, 10(7): 1246–1254. https://doi.org/10.1007/s11368-010-0206-6

Specht, K., Siebert, R., Hartmann, I., Freisinger, U.B., Sawicka, M., Werner, A., Thomaier, S., Henckel, D., Walk, H. & Dierich, A. 2013. Urban agriculture of the future: an overview of sustainability aspects of food production in and on buildings. *Agriculture and Human Values*, 31(1): 33–51. https://doi.org/10.1007/s10460-013-9448-4

Topager. 2012. *Topager* [online]. [Cited 18 August 2021]. http://topager.com/

TEDx Talks. 2019. Des toits cultivés pour une ville végétale | Baptiste Grard | TEDxUNamur. [Cited 18 August 2021]. https://www.youtube.com/watch?v=FdyqW7VnB1s&t=19s

Vidal-Beaudet, L. 2018. Du déchet au Technosol fertile : l'approche circulaire du programme français de recherche SITERRE. *VertigO* (Hors-série 31). https://doi.org/10.4000/vertigo.21887

Weidner, T., Yang, A. & Hamm, M.W. 2019. Consolidating the current knowledge on urban agriculture in productive urban food systems : Learnings , gaps and outlook. *Journal of Cleaner Production*, 209: 1637–1655. https://doi.org/10.1016/j.jclepro.2018.11.004

24. Organic amendments for soils rehabilitation of open-pit mines in Spain

Vicenç Carabassa[2], Josep M. Alcañiz[1,2], Xavier Domene[1,2]

[1]Universitat Autònoma de Barcelona, E08193 Bellaterra (Cerdanyola del Vallès), Catalonia, Spain

[2]CREAF, E08193 Bellaterra (Cerdanyola del Vallès), Catalonia, Spain

1. Related practices and hot-spots

Sewage sludge additions; Technosols

2. Description of the case study

Construction of Technosols mixing mineral (soils or mine debris) and organic materials is a reliable solution to recover the soil functions of the degraded mined lands. The addition of an organic amendment allows a faster establishment of vegetation cover, reduces soil erosion, and promotes higher carbon sequestration at both the medium and the long term in comparison to unamended Technosols (Sopper, 1993; Schad and Dondeyne, 2017).

The most suitable organic materials for this purpose are those subjected to a previous organic matter stabilization treatment, such as municipal sewage sludge, household compost, pig slurry, or compost-like-outputs (Carabassa *et al.*, 2020). By recycling those wastes, and by using mine wastes as the mineral component of these soils, this practice clearly contributes to the circular economy principles. However, care must be taken to ensure that their pollutant burden (e.g. heavy metals) do not exceed the legal threshold concentrations. For its maximum benefits, the organic fraction in Technosols require a homogeneous mixing with the mineral component, and to provide the initial supply of nutrients (e.g. N, P, K) and organic matter to accelerate the development of vegetation and biologically activate the soil, while avoiding excessive addition rates.

The dosage needs to be adjusted depending on the properties of either the mineral and organic components. Previous experience has indicated 30-40 t/ha (dry weight) as an optimum rate, while application over 50 t/ha are not recommended (Domene, Alcañiz and Andrés, 2007; Ojeda *et al.,* 2015; Carabassa, Ortiz and Alcañiz, 2018). Those approximately correspond to organic-to-mineral material ratios of 1:15 and 1:10 (v:v), respectively.

In the short term (1-2 years), a moderate but immediate increase of organic carbon in soil is achieved by the direct effect of the organic amendment, while the indirect effect on vegetation establishment cause a two- to three-fold increase in carbon in the medium (10 years) and the long term (>20 years) (Ojeda *et al.*, 2015; Carabassa *et al.*, 2019).

3. Context of the case study

This case study describes the decadal monitoring of rehabilitated areas in several limestone quarries located in Catalonia, NE Iberian Peninsula. Due to the predominant Mediterranean climate (xeric soil moisture regime), water is the main constraint for vegetation development. In this area, land denuded by mining activities is around 5 700 ha, that must be restored according the local regulation.

The scarcity of topsoils and their inherent nutrient scarcity (especially in terms of N and P) make the soil rehabilitation practices difficult; therefore, artificial soils (Technosols) using mining debris can represent a suitable alternative. This case study covers ca. 100 ha of rehabilitated land and reports 20 years of measurements (Photo 59).

4. Possibility of scaling up

This practice can be readily applied to the restoration of road slopes, landfills, to some contaminated soils, and other severely degraded lands where the development of a vegetation cover is intended, especially in semi-arid areas and dry tropics.

5. Impact on soil organic carbon stocks

This practice has a clear potential for C sequestration, as it has been proved at medium-long term (10-20 year) in several pilot trials (Table 88).

Table 88. Evolution of SOC stocks in the sewage sludge amended technosols

Location	Climate zone	Soil type	Baseline C stock, 0–20 cm (tC/ha)	Additional C storage potential 0–20 cm (tC/ha/yr)	Duration (Years)	More information	Reference
Catalonia, 7 quarries	Warm temperate dry (Mediterranean)	Skeletic (calcaric) Technosol	9.0	1.9	10	Sewage sludge-amended technosols	Carabassa *et al.* (2019)
Catalonia, Mont-ral		Calcaric Technosol	12.1	1.4	18		Gonzalez-Campistany, Carabassa and Alcañiz (2017) (Photo 59)
		Skeletic (calcaric) Technosol	14.8	2.1	6		Ortiz *et al.* (2012)

These results correspond to Technosols constructed using sewage sludge as soil amendment at addition rates below 50 t/ha (dry weight basis), to avoid the disadvantages found in previous experimental works (section 6.2). Over the evaluation period, the C-storage increase rates tended to slow down as the C saturation level of the soil approaches.

6. Other benefits of the practice

6.1. Benefits for soil properties

This practice contributes to improve soil aggregation. In the short term, sewage sludge addition improves the aggregates resistance to the raindrops impact, and significantly increases aggregate size (Sort and Alcañiz, 1999). In the longer term, the direct effect of this amendment is replaced by the formation of new aggregates associated to the development of vegetation. The positive effects on the soil structure decrease bulk density and favor water storage and infiltration (Ojeda, Alcañiz and Le Bissonnais, 2008), therefore reducing runoff and soil losses by erosion (Ojeda, Alcañiz and Ortiz, 2003). The application of sludge tends to lower soil pH due to the promotion of biological activity, although this effect is not usually significant in soils with a high carbonate content such as the ones in this case-study, but noticeable shifts in soil microbial and enzymatic activities have been demonstrated (Tarrasón *et al.*, 2010).

6.2 Minimization of threats to soil functions

Table 89. Soil threats

Soil threats	
Soil erosion	Clearly reduced by the initial organic amendment addition that favors soil aggregation, water infiltration and vegetation cover.
Nutrient imbalance and cycles	N, P and often K is directly supplied by the organic amendments and then kept in the rehabilitated ecosystem.
Soil salinization and alkalization	The moderate addition rates prevent salinization, restricted to slightly and transient increases just after the addition of the organic amendment. Excessive alkalization is not plausible even in calcareous soils.
Soil contamination /pollution	Caution must be taken to the selection of the organic materials used as amendment in terms of their origin and stabilization process. Attention should be paid to excessive heavy metals, organic pollutants and impurities (e.g. plastics or glass pieces), but also to excessive labile organic matter content.
Soil acidification	In acid soils, organic wastes can increase soil pH depending on their composition.
Soil biodiversity loss	Promotion of biological activity after the organic amendment.
Soil compaction	By inference from the effects on soil aggregation, this practice is expected to reduce soil compaction.
Soil water management	The improvement in soil aggregation and organic matter unavoidably increases water holding capacity and water availability.

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

Although the increase in primary production is not the main goal in mine-sites rehabilitation, clear increases had been demonstrated regarding plant biomass: in Technosols constructed with organic amendments plant biomass, as well as tree growth, is two or three times greater than unamended ones (Moreno-Peñaranda, Lloret and Alcañiz, 2004; Ortiz *et al.*, 2012).

6.4 Mitigation of and adaptation to climate change

Since this practice allows restoring degraded lands, increased CO_2 fixation by photosynthesis is expected, later partly sequestered as biomass and as soil organic C. As soil surface becomes protected from direct solar radiation, lower fluctuations of soil temperature and water losses by evaporation are expected.

6.5 Socio-economic benefits

The use of local organic and mineral materials for Technosols construction contributes to circular economy, reducing the costs for transportation, landfilling, or incineration.

6.6 Other benefits of the practice

The rehabilitation of degraded land allows recovering the ecosystem services lost, by the direct provision and regulation of environmental benefits, while reducing the costs for the reversal of environmental degradation (e.g. the lower soil losses due to erosion improves the quality of surface waters and reduces the water purification costs downstream). Landscape restoration also improves cultural services linked to local population wellbeing, in turn linked to environmental engagement or tourism attraction.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 90. Soil threats

Soil threats	
Soil erosion	Negative side effects are not expected.
Nutrient imbalance and cycles	When high application rates are used or when very P-rich wastes are applied N/P ratio imbalances might occur. A transient initial N overfertilization is possible before the full development of the vegetation cover, when maximum nutrient availability is present.
Soil contamination / pollution	An analytical control of organic amendments is required. The use of organic amendments with high pollutants content must be avoided.
Soil biodiversity loss	Some negative effects have been described for soil fauna at high amendment rates (Barrera *et al.*, 2001; Domene *et al.*, 2007). Transient shifts effects on plant species richness had been also reported (Moreno-Peñaranda *et al.*, 2004).

Soil threats	
Soil water management	Since this practice favors plant development, losses by transpiration could increase. However, this is not a drawback since the ecological succession will effectively adjust the community present to the rainfall regime of the area.

7.2 Increases in greenhouse gas emissions

Despite higher CO_2 emissions are expected in organically-amended Technosols than in unamended soils, a clearly more positive balance of C sequestration is expected initially, and especially along time, as a result of primary production. No higher N_2O and CH_4 emissions are expected as amended Technosols are well-aerated environments with high redox potential.

8. Recommendations before implementing the practice

Although several methodological approaches exist for addition of organic amendments to mineral substrates for Technosol construction, our experience recommend mixing before the surface spreading. Furthermore, the amended soil layer should be of at least 20 cm but not more than 40 cm to avoid overfertilization. When thicker soils are required, the topsoil layer could be placed over unamended materials.

9. Potential barriers for adoption

Table 91. Potential barriers to adoption

Barrier	YES/NO	
Biophysical	Yes	Avoid in wetlands to avoid eutrophication of the aquifers.
Cultural	Yes	Rejection due to lack of habit, resistance to change and lack of knowledge regarding the practice.
Social	Yes	During transportation and mixing of organic waste with mineral substrates, bad smells can reach neighboring inhabited areas, so those steps have to be carried out in low population areas.
Economic	No	The organic wastes used as amendment are generally supplied at zero cost, and only the transportation and mixing costs need to be considered.
Institutional	No	No legal barriers exist for waste handling if carried out according to good practices codes. Reuse of mineral and organic wastes is promoted by environmental authorities, respecting the contamination thresholds.
Legal (Right to soil)	No	Mining companies have the legal obligation to restore the exploited areas so this low-cost and effective approach facilitate the compliance of those legal provisions.
Knowledge	No	This practice is supported by experiments and demonstrative pilot tests conducted in a variety of quarries. A detailed description of this practice using sewage sludge is provided in Carabassa, Ortiz and Alcañiz, (2018), and in Carabassa *et al.* (2020) for compost-like-outputs and digestates.
Natural resource	No	This practice does not compete for limited natural resources as it is based on waste materials.

Photos

Photo 59. View of a former limestone quarry before and after a Technosol setup using sewage sludge as amendment: a) before replacement of the soil (1994), b) just after the topsoil surface spreading (1996), and c) after 18 years (2014). The developed plant community corresponds to an Aleppo pine (Pinus halepensis) forest with Santolina chamaecyparissus and Helichrysum stoechas as predominant bushes

References

Barrera, I., Andrés, P. & Alcañiz, J.M. 2001. Sewage Sludge Application on Soil: Effects on Two Earthworm Species. *Water, Air, and Soil Pollution*, 129(1): 319–332. https://doi.org/10.1023/A:1010335816237

Carabassa, V., Ortiz, O. & Alcañiz, J.M. 2018. Sewage sludge as an organic amendment for quarry restoration: Effects on soil and vegetation. *Land Degradation & Development*, 29(8): 2568–2574. https://doi.org/10.1002/ldr.3071

Carabassa, V., Domene, X., Díaz, E. & Alcañiz, J.M. 2020. Mid-term effects on ecosystem services of quarry restoration with Technosols under Mediterranean conditions: 10-year impacts on soil organic carbon and vegetation development. *Restoration Ecology*, 28(4): 960–970. https://doi.org/10.1111/rec.13072

Carabassa, V., Domene, X. & Alcañiz, J.M. 2020. Soil restoration using compost-like-outputs and digestates from non-source-separated urban waste as organic amendments: Limitations and opportunities. *Journal of Environmental Management*, 255: 109909. https://doi.org/10.1016/j.jenvman.2019.109909

Domene, X., Alcañiz, J.M. & Andrés, P. 2007. Ecotoxicological assessment of organic wastes using the soil collembolan Folsomia candida. *Applied Soil Ecology*, 35(3): 461–472. https://doi.org/10.1016/j.apsoil.2006.10.004

Gonzalez-Campistany, R., Carabassa, V. & Alcañiz, J.M. 2017. Segrest de carboni en pedreres rehabilitades amb sòls adobats amb fangs de depuradora. *Quaderns Agraris*, 42: 7-24. https://doi.org/10.2436/20.1503.01.72

Moreno-Peñaranda, R., Lloret, F. & Alcañiz, J.M. 2004. Effects of Sewage Sludge on Plant Community Composition in Restored Limestone Quarries. *Restoration Ecology*, 12(2): 290–296. https://doi.org/10.1111/j.1061-2971.2004.00310.x

Ojeda, G., Alcañiz, J.M. & Ortiz, O. 2003. Runoff and losses by erosion in soils amended with sewage sludge. *Land Degradation & Development*, 14(6): 563–573. https://doi.org/10.1002/ldr.580

Ojeda, G., Alcañiz, J.M. & Le Bissonnais, Y. 2008. Differences in aggregate stability due to various sewage sludge treatments on a Mediterranean calcareous soil. *Agriculture, Ecosystems & Environment*, 125(1): 48–56. https://doi.org/10.1016/j.agee.2007.11.005

Ojeda, G., Ortiz, O., Medina, C.R., Perera, I. & Alcañiz, J.M. 2015. Carbon sequestration in a limestone quarry mine soil amended with sewage sludge. *Soil Use and Management*, 31: 270-278. https://doi.org/10.1111/sum.12179

Ortiz, O., Ojeda, G., Espelta, J.M. & Alcañiz, J.M. 2012. Improving substrate fertility to enhance growth and reproductive ability of a Pinus halepensis Mill. afforestation in a restored limestone quarry. *New Forests*, 43(3): 365-381. https://doi.org/10.1007/s11056-011-9286-4

Schad, P. & Dondeyne, S. 2017. World Reference Base for Soil Resources. *In Encyclopedia of Soil Science, Third Edition*. https://doi.org/10.1081/e-ess3-120053850

Sopper, W.E. 1993. *Municipal Sludge Use in Land Reclamation*. 163p. Lewis Publishers, Florida, USA.

Sort, X. & Alcañiz, J.M. 1999. Effects of sewage sludge amendment on soil aggregation. *Land Degradation & Development*, 10: 3-12.

Tarrasón, D., Ojeda, G., Ortiz, O. & Alcañiz, J.M. 2010. Effects of Different Types of Sludge on Soil Microbial Properties: A Field Experiment on Degraded Mediterranean Soils. *Pedosphere*, 20(6): 681-691. https://doi.org/10.1016/S1002-0160(10)60058-6

25. Urban forestry effects on soil carbon in Leicester, United Kingdom of Great Britain and Northern Ireland

Bryant C. Scharenbroch[1], Nancy Cavallaro[2], Geraldine N. Vega Pizarro[3], Anna Paltseva[4], Maxine J. Levin[5]

[1]University of Wisconsin, Stevens Point, Soil and Waste Resources, Stevens Point, WI, United States of America

[2]Carbon Cycle Interagency Working Group (U.S.), Washington, DC, United States of America

[3]USDA-Natural Resources Conservation Service, Tolland, Connecticut, CT, United States of America

[4]School of Geosciences, University of Louisiana, Lafayette, LA,United States of America and Department of Landscape Design and Sustainable Ecosystems, Agrarian-Technological Institute, RUDN University, Moscow, Russian federation

[5]University of Maryland, Department of Environmental Science and Technology, College Park, MD, United States of America

1. Related practices and hot-spots

Urban forestry; Urban soils

2. Description of the case study

This case study examines the effects of urban trees on soil organic carbon (SOC) in the upper 100 cm in Leicester, United Kingdom of Great Britain and Northern Ireland. The study observed the effects of three tree genera (*Quercus*, *Fraxinus*, and *Acer*) and mixed-species woodlands on soil properties compared to urban grasslands that are mowed about 25 times a year and not irrigated. The parks ranged in age from 20 to more than 100 years old. The study trees within the parks ranged in life stages from saplings to large, mature trees. The study found that aboveground biomass and belowground services of urban trees are indirectly coupled. SOC enhancement of urban trees relative to urban grasslands was genus-specific. The study found that SOC was greater under *Fraxinus* and *Acer* spp. compared to urban grasslands, but similar to urban grasslands under *Quercus* spp. and mixed woodlands. Soils under F. excelsior had the highest SOC, with 11 kg/m^2 above the park grasslands. This species is known to quickly establish extensive and deep root networks compared to other broadleaf trees (Kerr and Cahalan, 2004). The increased SOC under this species was particularly marked in the 40-100 cm depth range. The study concludes that genus selection should be a consideration for SOC storage under urban trees, e.g. the greater SOC enrichment under F. excelcior There were no significant effects of woodlands on C/N ratios or on soil bulk density compared to park grasslands. The study also showed that urban grasslands stored 23 percent more SOC than typical agricultural lands in this region.

3. Context of the case study

The case study was conducted in urban parks in the city of Leicester, UK. Leicester has a population of 310 000 and an area of approximately 73 km^2. The region has a temperate climate, with 620 mm of precipitation per year and average annual daily minimum and maximum temperatures of 6.1 °C and 13.9 °C, respectively. The study was designed to test the hypothesis that SOC stocks and C/N ratios would be increased while bulk density would be reduced in soil under trees compared to urban grasslands. Soil organic carbon storage was measured under three target genera (*Acer, Fraxinus,* and *Quercus*), six species (*Acer campestre, Corylus avellana, Crataegus monogyna, Salix caprea,* and *Tilia x europaea*), and mixed woodlands. The selected tree species are in the top 4 most abundant tree species in the UK and most common in the parks in Leicester. The selected trees ranged in diameter at breast height from 2.5 cm to 197 cm and in biomass from 1.3 kg to 61 tons. SOC storage under these urban trees was compared to soils in urban grasslands located near the woodland sites within the same urban parks. The urban tree SOC stocks are compared to the baseline SOC stock in the urban grasslands to assess the impact of urban trees on SOC storage. Soils are deep and wet clays and loams. Soil types included Hanslope, Whimple, Salop, Beccles 3, Ragdale, and Fladbury 1. Soil types are from the National Soil Map for England and Wales produced by Cranfield University.

4. Possibility of scaling up

This case study is adapted for scaling up. The study examined some important urban tree genera and species in one city. A more comprehensive examination of urban tree species would be useful to better elucidate the importance of species-specific SOC storage and sequestration of the urban trees. It might also be useful to compare deep soil horizons (>100 cm depth) since SOC storage can also be stored in deep subsurface layers. Selection of tree species is important given the range of SOC accumulation seen in this study but selection must also consider adaptability to the climatic and soil conditions (drought, flooding, heat, etc.) in a given urban area, the likely changes in the climate into the future and the species susceptibility to common and emerging pests and pathogens.

5. Impact on soil organic carbon stocks

In this study, the additional C storage is the difference in mean SOC for urban grassland and urban parks with trees. Ideally, this number would be divided by the tree age to determine the annual rate, but tree ages were not listed. The parks selected in this study were established from 20 to over 100 years ago

The baseline value of C stock was 150 tC/ha (100-210 tC/ha) in urban grassland control. Tree SOC values ranged from 140 tC/ha (110-230 tC/ha) for mixed woodland, 140 tC/ha (90-210 tC/ha) for Quercus, 190 tC/ha (150-250 tC/ha) for Acer, and 260 tC/ha (160-400 tC/ha) for Fraxinus (Table 92).

Table 92. Evolution of soil carbon stocks at 0-100 cm depth under urban trees vs. urban grassland in Leicester, United Kingdom of Great Britain and Northern Ireland

Climate is Temperate cool according to the IPCC.

Soil type	Baseline C stock (tC/ha)	Additional C storage (tC/ha)	More information	Reference
Deep and wet clays and loams and not listed	150 (100-210)	+50	*Acer* spp. vs. urban grassland control	Edmonson *et al.* (2014)
	150 (100-210)	+110	*Fraxinus excelsior* vs. urban grassland control	
	150 (100-210)	No significant difference	*Quercus robur* vs. urban grassland control	
	150 (100-210)	No significant difference	Mixed woodland vs. urban grassland control	

6. Other benefits of the practice

6.1. Benefits for soil properties

The study focused mostly on SOC storage and sequestration. However, the study did measure other soil properties important for ecosystem services such as nutrient cycling and flood mitigation such as soil bulk density and C:N ratio. These properties relate to various soil threats such as erosion, nutrient balance, soil compaction and water management were not found to be significantly affected by tree species type and were not significantly different from the grassland soils.

6.2 Minimization of threats to soil functions

There were no significant differences between the forested and the grassland soils in terms of the factors that were measured that would relate to such threats as erosion, nutrient balance, acidification, water infiltration and retention, and soil compactions.

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

This case study did not quantify effects on provision services.

6.4 Mitigation of and adaptation to climate change

This case study did not quantify effects on Greenhouse Gases (GHGs) not related to carbon. However, given that these results show appreciable carbon sequestration from urban forests receiving no irrigation or fertilization suggests that the practice produces net mitigation of climate change. The deep rootedness of some of these trees confers adaptation to projected weather extremes expected with climate change. The effects of climate change on the increasing and emerging pests of these trees is mentioned so adaptation to these threats may be needed.

6.5 Socioeconomic benefits

This case study did not quantify socioeconomic benefits. However, it is well-recognized in numerous publications that urban trees provide benefits of reducing air, water and noise pollution (Nowak and Heisler, 2010), reduce heat island effects (Armson, Stringer and Ennos, 2012), and provide wildlife habitat and aesthetic value. (McPherson *et al.*, 1997)

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Not assessed in the study

7.2 Increases in greenhouse gas emissions

Data is not provided in this case study to estimate the net GHG balance.

7.3 Conflict with other practice(s)

A conflict might exist with urban trees and urban grasslands due to the shading effects of urban trees. However, natural savanna systems have illustrated that trees and grasses can coexist within the same landscape. Trees may be inappropriate in certain arid areas where they may deplete water resources.

7.4 Negative impact on production

No negative impacts on production exist for urban trees.

8. Recommendations before implementing the practice

This case study did not provide recommendations on planting and caring for urban trees but the benefits documented would lead to recommendation of urban trees where appropriate for climate and soil

Photo

Photo 60. An aerial view of Abbey Park in Leicester Abbey in 2003. The image shows urban tree and grasslands in the urban landscape

References

Armson, D., Stringer, P. & Ennos, A.R. 2012. The effect of tree shade and grass on surface and globe temperatures in an urban area. *Urban Forestry & Urban Greening*, 11(3): 245–255. https://doi.org/10.1016/j.ufug.2012.05.002

Edmondson, J.L., Odhran S., O'Sullivan, R.I., Potter, J., McHugh, N., Gaston, K.J. & Leake, J.R. 2014. Urban tree effects on soil organic carbon. *PLoS One*, 9: e101872. https://doi.org/10.1371/journal.pone.0101872

Kerr, G. & Cahalan, C. 2004. A review of site factors affecting the early growth of ash (Fraxinus excelsior L.). *Forest Ecology and Management*, 188(1): 225–234. https://doi.org/10.1016/j.foreco.2003.07.016

McPherson, E.G., Nowak, D., Heisler, G., Grimmond, S., Souch, C., Grant, R. & Rowntree, R. 1997. Quantifying urban forest structure, function, and value: the Chicago Urban Forest Climate Project. *Urban Ecosystems*, 1(1): 49–61. https://doi.org/10.1023/A:1014350822458

Nowak, D.J. & Heisler, G.M. 2010. *Air quality effects of urban trees and parks*. Research Series National Recreation and Park Association. (also available at: https://www.nrpa.org/globalassets/research/nowak-heisler-research-paper.pdf)

26. Urban agriculture in Tacoma, Washington, United States of America

Anna Paltseva[1], Nancy Cavallaro[2], Bryant C. Scharenbroch[3], Geraldine Vegas Pizarro[4], Maxine J. Levin[5]

[1]School of Geosciences, University of Louisiana, Lafayette, LA, United States of America; RUDN University, Moscow, Russian Federation

[2]Carbon Cycle Interagency Working Group (U.S.) CCIWG/USGCRP, Washington, DC, United States of America

[3]University of Wisconsin-Stevens Point, Soil and Waste Resources, Stevens Point, WI, United States of America

[4]USDA-Natural Resources Conservation Service, Soil Scientist, Tolland, CT, United States of America

[5]University of Maryland, Dept. of Environmental Science and Technology, College Park, MD, United States of America

1. Related practices and hot-spot

Urban Agriculture; Urban soils

2. Description of the case study

Urban soils are commonly contaminated by legacy contaminants from the past industrial activities, leaded gasoline and paint, pesticides. Various organic (e.g. compost, mulch, biochar) and inorganic (bone meal, fish bone, fly ash, slag, zeolites) amendments are applied to contaminated soils to bind with heavy metals or dilute them. Biosolids, that is products of the wastewater treatment process, were chosen in this study (McIvor, Cogger and Brown, 2012) to improve urban soil for agriculture because of their high content of organic matter and nutrients. This case study evaluated the effect of two types of biosolids on soil physical and chemical properties in three urban community gardens in Tacoma, Washington during two growing seasons.

Two study areas, the Franklin and Proctor gardens, were managed as community gardens for at least 30 years and the third area of La Grande garden (Guadalupe Land Trust) was managed for only 10 years. The plot in Franklin garden was managed for years with the biosolids-based soil amendment TAGRO, (short for "Tacoma Grow") prior to the study described here. The Proctor plot underwent yearly tilling to prepare for gardening season. The La Grande garden (Guadalupe Land Trust) experienced minimal management being covered in well-established turf. Onions (*Allium cepa*), beets (*Beta vulgaris*), carrots (*Daucus carota*), potatoes (*Solanum tuberosum*), and beans (*Phaseolus vulgaris*) were planted in the same pattern in all the subplots.

The researchers assessed two types of organic amendments based on biosolids. One biosolid type used was the TAGRO soil product made from **Class A biosolids** cake mixed with sand and sawdust. Specifically, solids came from a dual-digestion wastewater treatment plant (thermophilic aerobic digestion followed by mesophilic anaerobic digestion). They were later dewatered and mixed with sawdust and sand in a volume ratio of 40 percent biosolids, 40 percent sawdust, and 20 percent sand (Photo 62). Another type was the GroCo biosolid compost produced from **Class B biosolids** that have been anaerobically digested (Photo 61). The solids were blended with wood chips at a 1:3 volume ratio (25 percent biosolids, 75 percent wood) and composted for 9 months. Prepared biosolids were applied at 200 Mg/ha (d.w.) in the first year of the study. Plots were split for the second year with half of the amendment plots receiving an additional 200 Mg/ha of the amendments.

Photo 61. GroCo used in the study

Photo 62. TAGRO mix used in the study

3. Context of the case study

This local study was conducted in three urban community gardens in Tacoma, Washington—Franklin, Proctor, and La Grande. Tacoma a mid-sized urban port city with a population of over 200 000 people and elevation of 74 m (243 ft) with annual high temperature of 16.7 °C (62 °F) and annual low temperature of 7.2°C (45 °F). Average annual precipitation is 996 mm (39.22 inch) (US Climate Data, 2020). In the context of circular economy, the authors assessed the use of organic amendments based on biosolids from wastewater treatment. In this case study, they intended to show its benefits and possible trade-offs when transformed into organic amendments for soil application.

4. Possibility of scaling up

This case study is adapted for scaling up, as in general approximately 500 dry t/yr of biosolids are produced in individual water treatment plants of medium sized cities. In the United States, as of 2013 about 55 percent of sewage solids are turned into fertilizer. Based on 2019 biosolids annual reports (US EPA, 2020)

- Generated: ~4.75 Million Dry Metric Tons (dmt) of biosolids
- Land applied: ~2.44 million dmt biosolids
  - Applied to agricultural land: ~1.4 million dmt
  - Applied to non-agricultural land: ~1 million dmt
- Incinerated: ~765,000 dmt biosolids
- Landfilled: ~1 million dmt biosolids
- Other management practices (examples include deep well injection and storage): ~498,000 thousand dmt biosolids

Required data to scale up include carbon content in different urban agricultural soils across the country and with different amendment applications. The practices used in this study can be scaled-up and used in many urban settings and where municipal compost is available. According to the International Solid Waste Association (ISWA), the total of municipal solid waste produced annually in the United States of America is 238 t of which about 30% (67 t/yr) is organic waste. Of that only 21 t is recycled annually via composting and/or anaerobic digestion processes. Thus, there is ample room to increase available compost. The ISWA estimates that in 2015 2.6 million t/day of municipal organic waste was produced globally and that, based on trends in population and urbanization, that this may increase to 4.5 million t/day by 2050 (Ricci-Jugensen *et al.*, 2020).

5. Impact on soil organic carbon stocks

Ten soil samples per plot were collected using a 2.5-cm diameter soil hammer probe at a depth of 10-15 cm and composited for further lab analyses (Table 93).

Table 93. Evolution of soil carbon stocks in the two-year-study

Location	Climate zone	Baseline C stock (tC/ha)*	Additional C storage (tC/ha/yr)**	Duration	Depth (cm)	More information	Reference
Tacoma, Washington	Cool temperate moist	51.5–77.6	29.8	2 growing seasons	0–15	Addition of GroCo biosolid	McIvor, Cogger and Brown (2012)
		51.5–77.6	30.7			Addition of Tagro biosolid	

*The baseline value is the range of pre-treatment C stocks in 3 urban community gardens where experiments were performed.

** Additional C storage was calculated by multiplying the depth of sampling, bulk density after the treatments and C storage from the published results. The results were converted into tones of C per hectare and divided by 2 seasons.

6. Other benefits of the practice

6.1. Benefits for soil properties

Physical properties

Biosolids increased the surface water infiltration rate from 10.1 ± 0.95 ml/min in control soils to between 51 ± 6.1 and 212 ± 34 ml/min. Bulk density from a depth of 0 to 8 centimeters was decreased from 1.07 ± 0.01 g/cm^3 to between 0.51 ± 0.08 and 0.77 ± 0.03 g/cm^3.

Chemical properties

From a depth of 0 to 15 centimeters, decreased pH (5.35 -5.75); increased total carbon, nitrogen, and available phosphorus; no effect on soil metal concentrations.

Additionally, EPA states that when applied to land at the appropriate agronomic rate, biosolids provide nutrients and improve soil structure, and water reuse (EPA, 2016).

6.2 Minimization of threats to soil functions

Table 94. Soil threats

Soil threats	
Soil erosion	By increasing water infiltration rate, the biosolids would reduce runoff and erosion.
Nutrient imbalance and cycles	Increased nitrogen and available phosphorus.
Soil contamination/pollution	According to E.P.A, biosolids reduce the bioavailability of many pollutants.
Soil compaction	Decreased bulk density.
Soil water management	Biosolids increased the water infiltration rate.

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

According to EPA, biosolids were found to promote rapid timber growth, allowing quicker and more efficient harvest (EPA, 2016). In these urban community gardens, the service of food production for the gardeners would be increased.

6.4 Mitigation of and adaptation to climate change

The case study did not address GHGs. However, in that this use of biosolids avoids their deposition into landfills or their incineration, and increases plant growth, this practice serves to mitigate both methane and CO_2 emissions.

6.5 Socio-economic benefits

The land application of biosolids benefits communities by diverting the waste from sanitary landfills, which is also cheaper compared with traditional landfilling or incineration. It also provides an opportunity for the farmers and cities to collaborate in a cooperative venture (Arnold *et al.*, 1996). Reduced demand on non-renewable resources (e.g. phosphorus) and a reduced demand for synthetic fertilizers are also associated with biosolids' application (US EPA, 2016).

6.6 Other benefits of the practice

Biosolids used in this study are pathogen free and suitable for use by the general public with low cost and a soil like appearance. According to EPA, biosolids promote sustainable vegetation, reduce the bioavailability of soil toxic substances, control soil erosion, and regenerate damaged soils (US EPA, 2016).

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 95. Soil threats

Soil threats	
Soil contamination / pollution	Biosolids did not have a significant effect on total soil metal concentrations.
Soil acidification	Decreased pH.

7.2 Increases in greenhouse gas emissions

The case study did not address GHG emissions. However, as stated above, the practice would reduce direct GHG emissions from the biosolids compared to other uses or disposal of these biosolids.

7.3 Other conflicts

Although this has not been studied in this research, it is important to note that EPA's Biosolids Program assesses the potential risk of pollutants found in biosolids. EPA has found about 400 pollutants in biosolids since 1993 (when 40 CFR Part 503 was promulgated, US EPA, 2018) but not all of them will be present in every wastewater treatment facility.

8. Recommendations before implementing the practice

Before any biosolids are applied, background soil needs to be tested to determine the amount of arsenic, lead, cadmium, copper, mercury, zinc, selenium, nickel already present in the soil and in the biosolids. Each time biosolid is applied to a field, the number of pounds per acre of each of these trace elements that are applied must be added to the initial background level and to all previous biosolids applications (Stehouwer, 2010). In addition, annual biosolids applications should not exceed the nitrogen needs of the crop being grown, meaning that nitrogen content and forms in soil and biosolids need to be assessed. Timing of application should be based on the local weather and type of soil. This way nutrients are taken up by plants instead of being leached by rain. It is preferred to apply biosolids in the fall and early spring when there are no standing crops. The application is not recommended on snow-covered or frozen ground (Stehouwer, 2010). pH of biosolids should be considered before application as it affects metal behavior in soils (Lu, He and Stoffella, 2020). Avoid applying biosolids to soils with high levels of heavy metals or toxic organic compounds. EPA suggests seeking assistance in calculating the agronomic rate from the local extension agent or the soil testing department at the Land Grant University in each state.

9. Potential barriers for adoption

This case study by McIvor, Cogger and Brown (2012) doesn't discuss these aspects; however, EPA has requirements for Class A and Class B biosolids determined by the federal regulation 40 CFR Part 503 (US EPA, 2018), which creates incentives for beneficial use of biosolids. Because biosolids are a municipal byproduct, they contain many different synthetic organic and metal contaminants. Only a few of the contaminants are regulated, which are only 9 metals and no synthetic organic pollutants, according to EPA 503 which came into effect in 1993. Individual states may have more rigorous requirements and additional criteria. Also, most states require permits to apply biosolids in addition to a site evaluation. Some other barriers include odor, excess loading of nutrients (e.g. phosphorus, nitrogen, molybdenum), organic pollutants and pathogens (Lu, He and Stoffella, 2020). Biosolids are available for public purchase from hardware stores, home and garden centers or local wastewater treatment plants (US EPA, 2016).

Photo

Photo 63. Urban Agriculture on Governor's Island, New York

References

Arnold, K., Magai, R., Hoorman, R. & Miles, R. 1996. Benefits and risks of biosolids. *Extension - University of Missouri* [online]. [Cited 30 September 2020]. https://extension2.missouri.edu/wq427

McIvor, K., Cogger, C. & Brown, S. 2012. Effects of biosolids based soil products on soil physical and chemical properties in urban gardens. *Compost Science & Utilization*, 20(4): 199-206. https://doi.org/10.1080/1065657X.2012.10737049

Lu, Q., He, Z.L. & Stoffella, P.J. 2020. Land application of biosolids in the USA: a review. *Applied and Environmental Soil Science*, 2012: 1-11. https://doi.org/10.1155/2012/201462

Ricci-Jurgensen, M., Gilbert, J. & Ramola, A. 2020. *Global assessment of organic municipal waste production and recycling*. International Solid Waste Association Rotterdam, The Netherlands.

Stehouwer, R. 2010. Use of biosolids in crop production. *PennState Extension* [online]. [Cited 27 December 2020]. https://extension.psu.edu/use-of-biosolids-in-crop-production

US Climate Data. 2020. Weather averages Tacoma, Washington. *U.S. Climate Data* [online]. [Cited 30 September 2020]. https://www.usclimatedata.com/climate/tacoma/washington/united-states/uswa0441

US EPA. 2016. Basic information about biosolids. *US Environmental Protection Agency* [online]. [Cited 30 September 2020]. https://www.epa.gov/biosolids/basic-information-about-biosolids

US EPA. 2018. Standards for the Use or Disposal of Sewage Sludge (40 CFR Part 503). In: *US EPA* [online]. [Cited 30 September 2020]. https://www.epa.gov/biosolids/biosolids-laws-and-regulations

27. Soil organic carbon in forested and non-forested urban plots in the Chicagoland Region, United States of America

Bryant C. Scharenbroch[1], Nancy Cavallaro[2], Maxine J. Levin[3]

[1]University of Wisconsin, Stevens Point, Soil and Waste Resources, Stevens Point, WI, United States of America

[2]Carbon Cycle Interagency Working Group (U.S.) CCIWG/USGCRP, Washington, DC, United States of America

[3]University of Maryland, Department of Environmental Science and Technology, Regents Drive College Park, MD, United States of America

1. Related practices and hot-spot

Urban forestry; Urban soils

2. Description of the case study

Soil organic carbon (SOC) was quantified throughout the seven-county Chicago region, Illinois, United States of America. With a city population near 3 million and a metro area population approaching 10 million, Chicago is the third largest city and metropolitan area in the United States. It is located in the Great Lakes region, along the southwestern shore of Lake Michigan. This study was conducted to understand and model SOC distribution across this urban landscape, including 7 counties and over 14 thousand square kilometers. The study found that most SOC (>75 percent) was at depths greater than 25 cm and that 100 cm may not be deep enough to fully account for stored SOC. Compared to other terrestrial ecosystems (e.g. non-urban forests, prairies), SOC across the Chicago region is relatively high (250-750 tC/ha). Chicago region SOC was highest in the most urbanized green areas (e.g. commercial, industrial, utility, transportation land use categories) (450±30 tC/ha) and lowest in agriculture lands (246±34 tC/ha). Soil organic C was intermediate in urban forests (365±30 tC/ha), urban parks (366±30 tC/ha), and residential areas (365±30 tC/ha). A state-factor model was used to predict SOC storage. Important predictors of SOC across the Chicago region were land use and other anthropogenic factors (e.g. impervious surface area; distance to the city center, nearest building, highway and street) as well as surface (0-25 cm) soil properties (e.g. SOC, pH, K, microbial biomass, electrical conductivity). The study did not find strong evidence to suggest vegetation parameters (e.g. aboveground biomass, basal area increment) as important predictors for SOC. Furthermore, the study did not find a significant difference in SOC when comparing the plots with trees (343 ± 30 tC/ha) to those without trees (383 ± 30 tC/ha). The wide range of tree cover (basal area ranged from 0.04-1.9 m^2/ha) on these plots are a likely cause of this non-significant difference. However, the study does provide strong evidence of SOC variation by land use. Of relevance to this case study the mean SOC in the non-forested land use (agriculture) is 246 ± 34 tC/ha compared to a mean for the forested land uses (forests and urban parks) is 365 ± 30 tC/ha. These land-use SOC data were used to estimate baseline SOC stocks and additional C storage in forested urban landscapes. The urban forest areas would be managed less intensely than the most urbanized urban green areas associated with commercial and industrial land uses, with fewer outside inputs of carbon. So even though the most urbanized areas had the highest SOC, these soils were associated with the thickest human altered and human transported materials. Thus, for carbon sequestration and net carbon uptake into the future, urban forests would be the most promising path forward.

3. Context of the case study

Soil organic carbon (SOC) in the 0-100 cm soil depth was quantified on 190 plots (0.04 ha) distributed across the seven-county (14 625 km^2) region of Chicago, IL, United States of America. The plots were randomly stratified across the seven counties and in five different land uses: (1) commercial, industrial, transportation (40 plots), utility right-of-ways and vacant urban sites, forest, which included natural and restored forest, wetland, and savanna sites (40 plots), (3) park, which included parks, golf courses, schools and institutions with grass or landscaped understory (40 plots), (4) residential, which included single and multi-family residences and neighbourhoods (40 plots), and (5) agriculture, which included current farm land, riparian and ditch buffer zones in farm fields (30 plots). Twenty plots for each land-use had trees and twenty did not. The agriculture land-use included twenty plots without trees, but only ten of the agriculture plots met the criteria of having trees. Plots with trees were defined as plots containing woody vegetation, with at least three trees of 10 cm diameter at breast height. Plots with trees had at least 1 tree near plot center and the mean number of trees per tree plot was 8. Tree age data is not provided, but tree ages are estimated to be 10-200 years (personal communication with author, B. Scharenbroch on 12/02/20).

Two soil cores were collected on each plot to a depth of 100 cm. Horizons were described and identified according to standard soil science practices. Soil bulk density and organic carbon was determined on each horizon to compute the SOC stock. The total SOC stock was summed for the total of each pedon. The mean SOC for each plot was computed. According to Soil Surveys conducted in the Chicago region, the major soil orders (US Soil Taxonomy) present are Alfisols, Mollisols, Entisols, Inceptisols and Histosols. The most common suborders in the region are Aqualfs, Udalfs and Udolls. Much of the area is mapped as urban land or Udorthents.

Seventeen climate, relief, parent material, vegetation, and anthropogenic explanatory factors were also measured on each plot. The vegetation factors included: aboveground biomass, basal area, basal area increment, and litter depth. Twenty-one surface (0-25 cm) soil properties were measured as explanatory factors. The explanatory factors were used to develop models to predict SOC on these plots as described in Scharenbroch *et al.*, 2017. The models included state factor models with soil forming factors as drivers of SOC.

4. Possibility of scaling up

This case study is adapted for scaling up. Required data to do so would include carbon density per unit tree cover and urban tree cover measurements. The area of this study was extensive, but scaling up in the sense of increasing forested areas in urban areas is certainly possible and generally recommended where practical in urban areas with appropriate climates for forest growth.

5. Impact on soil organic carbon stocks

The baseline C stock and additional C storage were estimated using the case study data and the calculations described here. The baseline C stock is the mean and SE of the mean for SOC storage for the plots in the agriculture (246 tC/ha) land use, which were the lowest baseline C stock. The additional C storage (tC/ha/yr) was determined by computing the mean SOC storage for the forested and urban

park land use (366 tC/ha). The difference between the forested and non-forested SOC was 120 tC/ha. This difference in SOC between forested and non-forested was divided by the range of tree ages to estimate the annual SOC sequestration. The computed additional C storage in the forested plots ranged from 0.6 to 12 tC/ha/yr (Table 96).

Table 96. Soil organic carbon stocks in urban landscapes with trees compared to those without trees

Location	Soil type	Baseline C stock (tC/ha)	Additional C storage from forest planting (tC/ha/yr)	Duration	Depth (cm)	Reference
Chicago region of IL, United States of America	Area is mapped as urban land or Udorthents.	246±34	0.6–12	The duration for accumulation is take as tree ages	0–100	Scharenbroc, Bialecki and Fahey (2017)

6. Other benefits of the practice

6.1. Benefits for soil properties

No mention has been made of benefits on soil properties. However, other studies have shown that urban trees are influenced by and improve soil physical, chemical, and biological soil properties related to urban soil quality.

6.2 Minimization of threats to soil functions

The study only measured SOC storage and sequestration. Consequently, the effects of the practice in reducing soil threats was not analyzed.

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

Not assessed in the study.

6.4 Mitigation of and adaptation to climate change

Soil organic C densities were found to be relatively high in the Chicago urban area in comparison to local non-urban soils. In addition, urban lands are increasing through urbanization. Because of these high SOC densities in urban lands and increasing urban land areas, it is essential to include urban SOC stocks in global and regional ecosystem C budgeting. Urban forests can be adaptive in providing shade to mitigate urban heat-island effects and mitigate low humidity during hot and dry periods. Their carbon uptake and storage helps mitigate the rising atmospheric $CO_{2,}$ and if excess nitrogen fertilizer is avoided and good drainage is maintained, nitrous oxide or methane emissions would be negligible.

6.5 Socioeconomic benefits

These were not assessed in this study but green areas in cities is known to provide recreational benefits as well as cleaner air for the urban population.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Not assessed in the study.

7.2 Increases in greenhouse gas emissions

No data is provided in this case study to estimate the net GHG balance.

7.3 Conflict with other practice(s)

No major conflicts with other practices are considered in the case study but any land use change or decision for one area could potentially influence future land use in the same and surrounding areas.

7.4 Negative impact on production

There are no negative impacts on production unless land use is changed from agriculture to urban forests.

8. Recommendations before implementing the practice

Soil organic C densities were found to be significant in deep soil layers. It is essential that studies and other characterizations of urban SOC stocks sample deep soil layers to at least 1-m depths. Surficial soil sampling and characterization of SOC may substantially underestimate the SOC stock in urban and other lands.

9. Potential barriers for adoption

A major challenge for characterization of urban SOC is understanding the storage and sequestration under hardscapes. Urban areas have the unique barrier of limited access under sealed surfaces. Future research should focus on innovative techniques to sample soils under sealed surfaces and modeling approaches to better estimate SOC under sealed surfaces.

Photo

Photo 64. Miles Schwarz-Sax (L) and Bryant Scharenbroch (R) collecting a soil core from a forested urban park plot in Chicago, IL United States of America

References

Scharenbroch, B.C., Bialecki, M.B. & Fahey, R.T. 2017. Distribution and Factors Controlling Soil Organic Carbon in the Chicago Region, Illinois, USA. *Soil Science Society of America Journal*, 81(6): 1436-1449. https://doi.org/10.2136/sssaj2017.03.0087

28. Compost application to restore post-disturbance soil health in Montgomery County, Virginia, United States of America

Nancy Cavallaro[1], Geraldine N. Vega Pizarro[2]. Bryant C. Scharenbroch [3], Anna Paltseva[4], Maxine J. Levin[5]

[1]Carbon Cycle Interagency Working Group (U.S.) CCIWG/USGCRP, Washington, DC,United States of America

[2]USDA-Natural Resources Conservation Service, Tolland, CT, United States of America

[3] University of Wisconsin-Stevens Point, Soil and Waste Resources, Stevens Point, WI,United States of America

[4] School of Geosciences, University of Louisiana, Lafayette, LA, United States of America; RUDN University, Moscow, Russian Federation

[5] University of Maryland, Dept. of Environmental Science and Technology, College Park, MD,United States of America

1. Related practices and hot-spot

Parks, lawns, and gardens (Horticultural), Urban trees, compost additions; Urban soils

2. Description of the case study

An urban soil rehabilitation technique known as "profile rebuilding" has been proposed in Montgomery County (Virginia, United States of America) in order to avoid the many problems of soil degradation associated with business-as-usual practices for urban and suburban development as cities expand into native as well as agricultural lands to build roads, buildings and urban infrastructure. The study was carried out to evaluate this technique compared to business-as-usual practices and the undisturbed soil, with measurements being taken from 2011 to 2014 to determine the changes in soil physical characteristics, soil carbon sequestration, greenhouse gas emissions, microbial carbon, tree establishment and growth, and stormwater mitigation. Soil carbon assessments were on soil samples taken in 2011. Business as usual practices that include vegetation clearing, topsoil removal, stockpiling, compaction, grading, and construction (Chen *et al.*, 2014a, 2014b; Layman *et al.*, 2016), can degrade the soil. For example, in soils from this area that were disturbed for construction, including grading for stormwater redirection, macroaggregates were reduced by 44 percent at the 5-15 cm depth, the total surface carbon was reduced by 35 percent and 47 percent of the mineral bound carbon was lost, and drainage was reduced from about 7.5 cm/hr to about 1 cm/hr, and after 3 years, there was tree dieback, and turf thinning, and nutrient deficiencies (Chen *et al.*, 2013). The practical purpose of the study was to see if this practice, applied once at the time of soil disturbance for urban development can allow for more successful and rapid establishment of vegetation, reduce carbon loss, greenhouse gas emissions, and hydrology issues, and increase the resilience of the area to weather extremes.

For this case study a full field experiment was carried out in an area similar to those undergoing new urban expansion. The field experiment was conducted to mimic typical urban land development practices (two options, described below) compared to a soil profile rebuilding technique, and all three treatments were compared to an undisturbed soil. The treatments were applied in 2007 and are described as follows:

1. Topsoil removal and compaction followed by replacing 10 cm of reserved topsoil,
2. Same as (1) with tillage to 15 cm after topsoil replacement
3. Soil profile rebuilding technique using the same standard practice (treatment 1) but incorporating compost to a 60 cm depth with deep tillage and then replacing the topsoil (**Photo 65** and **Photo 66**). The compost used was from a composting facility that used municipal leaf collection feedstocks and had a C/N ratio of 15.0.
4. These treatments were compared with undisturbed soil (control) plots.

The treatments were applied to 24 plots 4.6 m by 18.3 m (4 treatments and 6 replicates in a completely randomized experimental design) in the state of Virginia, United States of America. In 2008, the plots were planted with 5 different tree species (*Acer rubrum* L., *Quercus bicolor Willd.*, *Quercus macrocarpa Michx.*, *Ulmus 'Morton'* (Accolade®) (*Ulmus japonica (Rehd.) Sarg.* × *Ulmus wilsoniana Schneid.*), and *Prunus 'First Lady'* (*Prunus* × *incam Ingram* ex R. Olsen & Whittemore 'Okamé' × *Prunus campanulata Maximowicz*) with 5 trees per plot) to evaluate tree growth and canopy changes from the treatments. Results showed that the profile rebuilding techniques were effective in increasing the carbon storage in available and aggregate-protected carbon pools and microbial biomass in subsurface soils while the standard urban soil treatments released mineral-bound soil carbon (Chen *et al.*, 2013; Chen *et al.*, 2014a). The practice resulted in an increase of global warming potential due to increased CO_2 efflux. However, this CO_2 efflux was at least partially offset by increased biomass production as demonstrated in measurements of tree growth in these same plots (Layman *et al.*, 2016). It was shown that after 6 years, trunk cross-sectional area and canopy area matched or surpassed those in the undisturbed soil. The soil rehabilitation through profile rebuilding accelerated establishment and growth of these urban trees by up to 84 percent compared to the two standard (BAU) treatments. Continued enhanced growth of these trees and other vegetation is expected to continue to increase carbon storage and decrease the global warming potential. Effects on methane (CH_4) and nitrous oxide (N_2O) were not statistically significant for any treatment.

3. Context of the case study

The study site is in Montgomery County, Virginia, in the eastern part of the United States. The site was previously used as pasture until the above treatments were applied in 2007 to assess the potential effects of profile rebuilding on soil carbon, greenhouse gas emissions, microbial activity, soil physical properties, and vegetative growth in a controlled replicated experimental setting. The climate is humid subtropical with hot summers and mild winters (average annual temperature 11.3 °C). Rainfall is moderate throughout the year (an average of 1 007 mm rainfall per year, and 50 cm snowfall). The area under investigation included two closely related loamy soils: Shottower loam (fine, kaolinitc, mesic Typic Paleudult) and Slabtown loam (fine-loamy, mixed, mesic Aquic Paleudalf).

4. Possibility of scaling up

The practices used in this study can be scaled-up and used in many urban settings and where municipal compost is available. According to the International Solid Waste Association (ISWA), the total of municipal solid waste produced annually in the United States of America is 238 t of which about 30 percent (67 t/year) is organic waste. Of that only 21 t is recycled annually via composting and/or anaerobic digestion processes. Thus there is ample room to increase available compost. The ISWA estimates that in 2015 2.6 million t/day of municipal organic waste was produced globally and that, based on trends in population and urbanization, that this may increase to 4.5 million t/day by 2050 (Ricci-Jugensen, Gilbert and Ramola, 2020).

5. Impact on soil organic carbon stocks

Total soil carbon and carbon pools were measured using a standard soil carbon density fractionation technique (Six *et al.*, 1998). Microbial carbon was also determined according to the chloroform fumigation extraction method. The addition of compost and deep tilling as soil rehabilitation techniques increased the carbon storage, a key soil component that is lost during urban land development. Loss of carbon took place in the plots where the soil was treated as for urban development. Thus, to assess the potential of reducing this post-development loss, the baseline values in Table 97 are those from the most commonly used treatments in urban development in the region. Because there was no significant difference between the two BAU treatments (with and without rototilling) the values in Table 97 are the average of these two treatments. However, it is useful to see the values for the undisturbed soil, so they are included in Table 97. The profile rebuilding treatment was able to recover about half of the carbon that was lost over 4 years and this carbon was in the 15-30 cm depth, where carbon is less vulnerable to loss compared to surface soil. Although compost was incorporated to a 60 cm depth, this study did not sample this soil past 30 cm so it is likely that this underestimates the total carbon sequestered via this profile rebuilding management practice.

The practice resulted in higher total soil carbon in the profile (6.12 tC/ha at 15-30 cm depth alone and likely more at deeper depths), larger available carbon (1.59 tC/ha), and larger aggregate-protected carbon pool (2.04 tC/ha) than the other treatments including the undisturbed soils at 15-30 cm depth. The practice also had the greatest microbial biomass carbon at 15-30 cm.

In the table below, the baseline is the typical treatment (BAU) in urban development (scraping the soil to 25-30 cm depth followed by compaction via a compactor, then replacement of 10 cm of the topsoil) rather than the undisturbed soil.

Table 97. Evolution of soil organic carbon stocks on 0-5, 5-10 and 15-30 cm on the study site in Montgomery County, Virginia, United States of America

Climate zone is Warm temperate moist according to IPCC and soils are classified as Loamy soils.

The values in parentheses are the standard deviations.

Baseline C stock of BAU treatments (tC/ha)	Undisturbed soil (tC/ha)	Annual additional C storage in profile rebuilding treatment (tC/ha/yr)	Depth (cm)	Duration (Years)	Reference
6.51 (1.4)	12.1 (1.95)	-0.03 (0.29)	0–5	4	Chen *et al.*, 2013
5.5 (1.3)	7.73 (1.22)	0.49 (0.4)	5–10		
2.51 (1.25)	3.79 (0.92)	1.20 (0.58)	15–30		

6. Other benefits of the practice

6.1. Benefits for soil properties

Physical properties

Compost application and deep tilling increase hydraulic conductivity (Chen *et al.*, 2014b). The saturated soil hydraulic conductivity in the subsurface was improved and helped mitigate stormwater runoff in urban areas (Chen *et al.*, 2014b). Soils subjected to profile rebuilding had about twice the saturated hydraulic conductivity (14.8 and 6.3 cm/h at 10-25 cm and 25-40 cm, respectively) of undisturbed soils and approximately 6-11 times that of soils subjected to typical land development practices (Chen *et al.*, 2014b). Bulk density was reduced at 0-15 cm (from 1.33 to 1.22 g/cm^3) and at 15-30 cm (from 1.44 to 1.25 g/cm^3).

Chemical properties

Adding compost can increase the soil pH and carbon to nitrogen (C/N) ratio. The increases in carbon dioxide flux and tree growth/nutrient availability suggested an increase in microbial activity (Chen *et al.*, 2013, Chen *et al.*, 2014a). Aggregate stabilized carbon also increased.

Biological properties:

Microbial biomass was measured in the 0-5 cm and 5-10 cm depths but was highly variable so that no statistically significant effect was detected. However, as noted above, nutrient availability and carbon dioxide flux increases

(Chen *et al.*, 2014a) suggest increased microbial activity at depths below 10 cm, and increased tree growth in the compost plots would also cause increased root respiration.

6.2 Minimization of threats to soil functions

Table 98. Soil threats

Soil threats	
Soil erosion	Increased soil hydraulic conductivity and reduced compaction mitigates the soil erosion potential. This area is mostly gently rolling terrain, primarily broad, gently sloping ridges bordered by long steep sideslopes (USDA, 1985).
Nutrient imbalance and cycles	Compost increases the nutrient content (N, P, Ca, Mg, several micronutrients) of the amended soils (Chen *et al.*, 2013).
Soil acidification	Increased soil organic matter often mitigates the effects of soil acidity or otherwise improving soil pH.
Soil biodiversity loss	Healthier and more fertile soils help preserve and enhance biodiversity, although this was not specifically assessed in this study.
Soil compaction	Compaction level decreased as evidenced by decreased bulk density and increased hydraulic conductivity (Chen *et al.*, 2013).
Soil water management	Improved stormwater capture (Chen *et al.*, 2013).

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

The practice restores or enhances soil health indicators related to tree growth, carbon content and stabilization, water relations and bulk density, allowing for land uses that can provide food, fiber, and habitat. Enhanced tree establishment and growth demonstrated in this study suggest fuel, timber and habitat enhancement.

6.4 Mitigation of and adaptation to climate change

All treatments demonstrated a CH_4 sink, but CH_4 uptake was too variable to demonstrate a treatment effect. Likewise, N_2O fluxes were not statistically different across treatments. N_2O flux was not affected by the treatment most of the year. The N_2O was only higher in spring when moisture levels were higher.

However, the improved soil physical properties and carbon accumulation contribute to mitigation as well as decreased vulnerability to extreme weather events such as drought, flooding and erosion. The faster establishment of trees and their enhanced growth is expected to continue to increase in the next decade and sequester carbon from the atmosphere. Trees also provide shade that can reduce energy needed for air conditioning when there are nearby buildings and residences.

6.5 Socio-economic benefits

Although the study did not specifically assess drought resilience or nutrient needs, based on other studies on the effects of compost and carbon accumulation in soils as well as the increased tree growth that was measured, it seems clear that implementing the practice would help to reduce the need for additional nutrients and water in the future, thus decreasing the maintenance cost of urban forests and landscapes in the long run. Healthier soils within the urban landscape allow for the provision of important social values such as recreation and the experience and appreciation of natural systems and cleaner air.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 99. Soil threats

Soil threats	
Nutrient imbalance and cycles	The compost quality must be high. The compost used in this study had a C:N ratio of 15. Compost amendments with a C/N ratio > 20 could immobilize nitrogen and cause N deficiencies in plants (Day, 2016).
Soil salinization and alkalinization	The soils in this study did not have salinization or alkalization issues. The compost will not address high pH in soils, and the practice may increase the pH (Day, 2016) in some cases, depending on the quality of the compost.
Soil contamination/pollution	The compost will have little or no effect in soils contaminated with heavy metals (Day, 2016) but the soils in this study gave no indication of potential pollution issues. Compost used for this purpose should be screened for any risk of introducing contaminants to the soil.

Soil threats	
Soil acidification	Compost effects on soil pH vary according to the original soil pH and the quality of the compost. There is seldom any adverse effect since the effect tends to be a mitigation of acidity in acid soils and mitigation of alkalinity in alkaline soils.

7.2 Increases in greenhouse gas emissions

The mean greenhouse gas (GHG) emission was 4.6 g C/m^2/day (16.9 g CO_2eq/m^2/d) of CO_2-equivalent after 4-5 years of experiment installation. As noted above, this analysis does not consider the offsetting of some of this GWP by enhanced biomass growth and carbon accumulation below 30 cm depth. These measurements do not take into account any additional emissions due to fuel needed for incorporating the compost to 60 cm. This would be a one-time emission that could be compensated in post-treatment changes in potential fertilizer, irrigation and tillage practices that could be needed in the non-composted plots that had poorer growth and soil quality. Also, GHG measurements were not taken during treatment application nor in the first 3-4 years following soil disturbance.

Table 100. Percent contribution to GWP

Treatment	CO_2 equivalents (gC/m^2/d)	Percent contribution to GWP (%)		
		CO_2	N_2O	CH_4
Compost, deep tilling and tree planting	4.6	98.7	1.5	-0.2

7.3 Conflict with other practice(s)

This practice can be used with many other practices that can disturb soils (such as timber harvest operations and soil disturbance for water management) and would not be in conflict.

7.4 Negative impact on production

No negative impact.

8. Recommendations before implementing the practice

The compost application depth is greater than normally applied by most contractors. Communication with the contractor and equipment operator must be clear to avoid the compost application at the wrong depth (Day, 2016).

9. Potential barriers for adoption

Table 101. Potential barriers to adoption

Barrier	YES/NO	
Biophysical/ Natural resource	Yes	As described above, the compost is from leaf compost that is commonly available and plentiful in most areas of the US where autumn leaves are collected and composted by the municipalities. However, infrastructure must be in place.
Economic	Yes	The additional cost of deep tillage and incorporation of compost may be a barrier to adoption by land and urban developers.

Photos

Photo 65. In the profile rebuilding technique employed in the Virginia case study in 2007, the compost is spread 4 inches (10 cm) deep over the soil surface

Photo 66. Compost application. During the compost application and backhoe subsoiling, the operator must scoop a bucket of soil with compost on the top, lift it several feet in the air, and drop it. This breaks up the compacted soils into large clods and creates veins of compost down

References

Chen, Y., Day, S.D., Wick, A.F., Strahm, B.D., Wiseman, P.E. & Daniels, W.L. 2013. Changes in soil carbon pools and microbial biomass from urban land development and subsequent post-development soil rehabilitation. *Soil Biology and Biochemistry*, 66: 38–44. https://doi.org/10.1016/j.soilbio.2013.06.022

Chen, Y., Day, S.D., Shrestha, R.K., Straum, B.D. & Wiseman, P.E. 2014a. Influence of urban land development and soil rehabilitation on soil-atmosphere greenhouse gas fluxes. *Geoderma*, 226-227: 348-350. https://doi.org/10.1016/j.geoderma.2014.03.017

Chen, Y., Day, S.D., Wick, A.F & McGuire, K.J. 2014b. Influence of urban land development and subsequent soil rehabilitation on soil aggregates, carbon, and hydraulic conductivity. *Science of the total environment*, 494-495, 329-336. https://doi.org/10.1016/j.scitotenv.2014.06.099

Day, S.D. 2016. Soil Profile Rebuilding: An Alternative to Soil Replacement. *City Trees*. (also available at: https://www.urbanforestry.frec.vt.edu/SRES/).

Layman, R.M., Day, S.D., Mitchell, D.K., Chen, Y., Harris, J.R. & Daniels, W.L. 2016. Below ground matters: Urban soil rehabilitation increases tree canopy and speeds establishment. *Urban Forestry and Urban Greening*, 16: 25-35. https://doi.org/10.1016/j.ufug.2016.01.004

Ricci-Jurgensen, M., Gilbert, J. & Ramola, A. 2020. Global assessment of organic municipal waste production and recycling. International Solid Waste Association Rotterdam, The Netherlands (also available at:
https://www.iswa.org/uploads/media/Report_1_Global_Assessment_of_Municipal_Organic_Waste_Compressed_v2.pdf)

Six, J., Elliott, E.T., Paustian, K. & Dorian, J.W. 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Science Society of America Journal*, 62: 1367-1377. https://doi.org/10.2136/sssaj1998.03615995006200050032x

United States Department of Agriculture (USDA). 1985. Soil Survey of Montgomery County Virginia. (also available at:
https://www.nrcs.usda.gov/Internet/FSE_MANUSCRIPTS/virginia/montgomeryVA1985/montgomeryVA1985.pdf)

29. Management of ornamental lawns and athletic fields in California, United States of America

Geraldine N. Vega Pizarro[1], Nancy Cavallaro[2], Maxine J. Levin[3], Bryant C. Scharenbroch[4], Anna Paltseva[5]

[1]USDA-Natural Resources Conservation Service, Tolland, Connecticut, Tolland, CT, United States of America

[2]Carbon Cycle Interagency Working Group (U.S.) CCIWG/USGCRP, Washington, DC, United States of America

[3]University of Maryland, Department of Environmental Science and Technology, College Park, MD, United States of America

[4]University of Wisconsin-Stevens Point, Soil and Waste Resources, Stevens Point, WI, United States of America

[5]School of Geosciences, University of Louisiana, Lafayette, LA,United States of America and Department of Landscape Design and Sustainable Ecosystems, Agrarian-Technological Institute, RUDN University, Moscow, Russian federation

1. Related practices and hot-spots

Gardens, parks, and lawns; Drylands, Urban soils

2. Description of the case study

Turfgrass is a common element in urban areas and covers more than 1.9 percent of land in the United States (Townsend-Small and Czimczik, 2010). Carbon sequestration and nitrous oxide emissions (N_2O) were quantified in four parks in Irvine of southern California, United States (May 2008-2009). Carbon dioxide emissions generated by lawn management practices, fertilizer production, and irrigation were also measured. The parks were established between 1975 and 2006. Each park under study contained two turf types: ornamental lawns (Photo 67) and athletic fields (Photo 68). Ornamental lawns were established from seed on existing soil. Athletic fields were constructed from imported turfgrass sods that add allochthonous OC to the system, then are normally renovated extensively every year, including tilling and re-sodding to replace dead grass, and frequent aeration to offset compaction, similar to practices employed in conventional agriculture. The management for both lawns in this case study included trimming and mulching grass clippings weekly and watering with recycled wastewater based on local estimates of evapotranspiration: soils consistently had high soil moisture (20 – 50 percent of pore space). This study had a total of five fertilizer events (May 2008-2009) in both turf types and sampled daily starting one or two days prior to fertilization and continuing until fluxes returned to baseline levels (about 8 days). Fertilizers included sulfur-coated urea, calcium nitrate, Nitra King (19-4-4), and Turf Supreme (16-6-8).

The average SOC sequestration was 1.4 tC/m^2/yr. N_2O emission was 45 to 145 gN/m^2/yr. The global warming potential (GWP) for ornamental lawns was -108 to 285 g CO_2/m^2/yr (depending on fertilization rate) and 405 to 798 g CO_2/m^2/yr for athletic fields. Urban ornamental lawns have the potential to sequester atmospheric carbon if managed properly with conservation practices and supplemented with irrigation. In the ornamental lawns in the current study, which are not subject to intense tilling or intense compaction from manipulation, the high productivity of the perennial vegetation overwhelms preexisting differences between parks in soil type, density, and initial OC content, resulting in significant storage of SOC in soils. These high SOC stocks are typical for ornamental lawns, which have the highest SOC density of urban soils and can accumulate SOC rapidly (Pouyat, Yesilonis and Golubiewski, 2009).

3. Context of the case study

The case study was conducted in four parks located within a 7 km radius in Irvine, CA, United States (33°41′N, 117°47′W). The area has an average annual temperature of 19°C and rainfall 350 mm/yr. The area was under agriculture for more than 100 years before the parks' establishment. Soils are loamy and formed from moderately alkaline and calcareous parent material with less than 1 percent organic carbon. Representative Soil series are the Chino silty clay loam (Mollisol), Omni silt loam (Mollisol), Calleguas clay loam (Entisol), and Mocho loam (Mollisol).

4. Possibility of scaling up

The practices used in this study can be scaled up and applied in many urban settings, however these practices may be specific to warm temperate dry IPCC climate zones, as soils within the warm temperate dry zones have relatively low baseline stocks (less than 1 percent) and do not sequester added soil carbon easily through dry seasons.

5. Impact on soil organic carbon stocks

There is a potential for carbon sequestration in ornamental lawns. In athletic fields there is no carbon storage due to the constant surface restoration. To assess SOC sequestration, the study sampled 8–12 cores to 20 cm depth in each park and each treatment every 10 m in linear transects. The number of samples taken in each turf type reflected the relative area covered by each type. Samples were dried at 60 °C, weighed, and sieved to remove rocks >2 mm. Soil bulk density was measured on every sample and corrected for the mass of particles >2 mm. A sub-sample was ground to powder and acidified with 2M HCl for 24 hrs to remove carbonates. Total OC and nitrogen contents were quantified with an elemental analyzer (EA), and stocks were calculated using EA data and bulk density.

Ornamental lawns had an initial carbon stock of 12 tC/ha for the top 20 cm of soil in 2008, considered low; however the organic carbon sequestration rate (2008-2009) averaged 1.4 tC/ha/yr. Athletic fields had an initial organic carbon content of 35 tC/ha in 2008. The 4 parks had varied years of establishment: in ornamental lawns and athletic fields from 2 to 33 years (average ± standard error). Measurements showed a significant linear relationship between SOC stock and lawn age in ornamental lawns, corresponding to accumulation rate of 1.4 tC/ha/yr. There was no consistent trend over the study period, however, the oldest athletic field had more organic carbon. These fields do not store organic carbon *in situ* until more than 30 years after establishment (Table 102).

Table 102. Evolution of soil organic carbon stocks at 0-20 cm depth in 4 parks of Irvine, California, United States of America

Climate is Warm Temperate Dry according to the IPCC.

<table>
<tr><th>Location</th><th>Soil type</th><th>Baseline C stock* (tC/ha)</th><th>Additional C storage (tC/ha/yr)</th><th>Duration** (Years)</th><th>More information</th><th>Reference</th></tr>
<tr><td>4 parks (Ornamental lawns)</td><td rowspan="2">Entisol, Mollisols</td><td>12</td><td>1.4</td><td rowspan="2">2–33</td><td></td><td rowspan="2">Townsend-Small and Czimczik (2010)</td></tr>
<tr><td>4 parks (Athletic fields)</td><td>35</td><td>NA</td><td>Reconditioned yearly, no basis for consistent accumulation or storage.</td></tr>
</table>

*Baseline value of C stock estimated 2008-9 with substitute measurement;

**4 ages of establishment of park fields and lawns: 2, 10, 20, 33 years

6. Other benefits of the practice

6.1. Benefits for soil properties

There is no data about benefits on soil properties.

6.2 Minimization of threats to soil functions

Table 103. Soil threats

Soil threats	
Soil erosion	Turfgrass prevents soil erosion (Monteiro, 2017).
Nutrient imbalance and cycles	There was no significant accumulation of N over time. Total N was 3.5 t/ha. It could be due to a high uptake by roots or by leaching (Townsend-Small and Czimczik, 2010).
Soil water management	Turfgrass promotes water infiltration with the improvement of soil structure and porosity (Monteiro, 2017).

6.3 Mitigation of and adaptation to climate change

Through irrigation and maintenance of park urban infrastructure (ornamental lawns, athletic fields, trees), these conservation practices enhance carbon storage and improve physical structure and sustainability of the soil structure and soil health. Good management improves soil health by improving water holding capacity, infiltration and soil structure thus allowing better sustainability and mitigation to extreme climate events such as drought or intense rainfall.

6.4 Socio-economic benefits

A well-maintained lawn in public parks provides recreation and open space as well providing economic enhancement to surrounding neighborhoods.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 104. Soil threats

Soil threats	
Nutrient imbalance and cycles	Constant management in athletic fields can cause loss of nutrients like P. This will promote the use of more fertilizer to maintain the quality of the turfgrass (Monteiro, 2017).
Soil contamination / pollution	Excess use of fertilizer can cause nutrients leaching and be a source of groundwater pollution (Qian *et al.*, 2003).
Soil compaction	Lawn management can cause soil compaction by using heavy equipment to mow the grass (Matthieu *et al.*, 2011).

7.2 Increases in greenhouse gas emissions

N_2O emissions are between 0.45 to 1.45 tN/ha/yr, determined primarily by fertilization rate. It contributes to a GWP of 45 to 145 g $CO_2/m^2/yr$.

This rate is offset in ornamental lawns by the organic carbon accumulation of 1.4 tC/ha/yr, offsetting 513 g $CO_2/m^2/yr$ of GWP. For organic carbon storage in athletic fields there is no net storage of CO_2 to offset these N_2O emissions. However, during lawn management, organic carbon sequestration is outweighed by the N_2O and CO_2 emissions released from fuel use, fertilization, and irrigation. Thus, the total GWP for organic lawns was -108 to 285 g $CO_2/m^2/yr$ (depending on fertilization rate) and in athletic fields the GWP was 405 to 798 g $CO_2/m^2/yr$. These numbers do not include methane emissions or uptake since these were not measured in this study. The organic carbon accumulation will eventually reduce the amounts of energy needed for both fertilizer and irrigation needs but clearly the main problem with these lawn management practices with regard to global warming potentials is related to the use and type of fertilization and the energy consumed in irrigation and other practices. While mulching reduces the GWP, in order to accomplish greater mitigation and sustainability, renewable energy sources and by use of carbon neutral fertilizer sources and/or biological nitrogen fixation for nutrient needs.

8. Recommendations before implementing the practice

Athletic fields and ornamental lawns require intensive management for conventional esthetic standards. Business as usual includes removal of grass clipping, and constant mowing which reduce the organic carbon and soil nutrients. The constant application of fertilizer to maintain the turf quality can contribute to groundwater pollution and greenhouse gas emission. It is recommended to implement conservative management practices to reduce the impact of lawns in the environment. These include compost and compost extract applications that could be used to improve the turfgrass quality to limit the use of synthetic fertilizers. Several authors such as Johnson (2018), Glab *et al.* (2018), and Boulter, Boland and Trevors (2000, 2002) reported that if managed properly and considering the soil type, a mature compost application improves the soil quality by increasing the microbiodiversity which increases the soils potential to sequester carbon; improves soil bulk density, porosity, and water infiltration; reduces soil compaction; increases available water capacity; suppresses turfgrass diseases and supports turfgrass growth. There is no documentation as to whether these practices were permanently adopted as a follow-up to the research study. Returning the grass clippings to the soil during mowing is a good conservation practice but may detract from conventional esthetics of well-groomed and maintained Athletic fields and ornamental lawns.

9. Potential barriers for adoption

Table 105. Potential barriers to adoption

Barrier	YES/NO	
Biophysical / Natural resource	Yes	Athletic fields and ornamental lawns require intensive management and water irrigation. The amount of water needed to maintain the good quality of lawns represents a natural resource conservation concern (Monteiro, 2017).
Economic	Yes	Lawns require intensive management and water irrigation. (Monteiro, 2017). The amount of water needed to maintain the good quality of lawns represents an economic concern.

Photos

Photo 67. Ornamental lawn in Irvine, California, United States of America

Photo 68. Lawn in a baseball park in Irvine, California United States of America

References

Boulter, J.I., Boland, G.J. & Trevors, J.T. 2000. Compost: a study of the development process and end-product potential for suppression of turfgrass disease. *World Journal of Microbiology & Biotechnology*, 16: 115-134. https://doi.org/10.1023/A:1008901420646

Boulter, J.I., Trevors, J.T. & Boland, G.J. 2002. Microbial studies of compost: bacterial identification, and their potential for turfgrass pathogen suppression. *World Journal of Microbiology & Biotechnology,* 18: 661-671. https://doi.org/10.1023/A:1016827929432

Glab, T., Zabinski, A., Sadowska, U., Gondek, K., Kopec, M., Mierzwa-Hersztek, M. & Tabor, S. 2018. Effects of co-composting maize, sewage sludge, and biochar mixture on hydrological and physical qualities of sandy soil. *Geoderma*, 315: 27- 35. https://doi.org/10.1016/j.geoderma.2017.11.034

Johnson, D. 2018. *Compost for soil regeneration: Johnson-Su composting bioreactor* [online]. [Accessed 30 November 2020]. Regeneration International. https://regenerationinternational.org/bioreactor/

Matthieu, D.E., Bowman, D.C., Thapa, B.B., Cassel, D.K. & Rufty T.W. 2011. Turfgrass Root Response to Subsurface Soil Compaction. C*ommunications in Soil Science and Plant Analysis Journal*, 42(22): 2813–2823. https://doi.org/10.1080/00103624.2011.622826

Monteiro, J.A. 2017. Ecosystem services from turfgrass landscapes. *Urban Forestry and Urban Greening*, 26: 151–157. https://doi.org/10.1016/j.ufug.2017.04.001

Pouyat, R.V., Yesilonis, I.D. & Golubiewski, N.E. 2009. A comparison of soil organic carbon stocks between residential turf grass and native soil, *Urban Ecosyst.*, 12: 45–62. https://doi.org/10.1007/s11252-008-0059-6.

Qian, Y.L., Bandaranayake, W., Parton, W.J., Mecham, B., Harivandi, A.M. & Mosier, A.R. 2003. Long-term effects of clipping and nitrogen management in turfgrass on soil carbon and nitrogen dynamics: The CENTURY model simulation. *Journal of Environmental Quality*, 32(5): 1694–1700. https://doi.org/10.2134/jeq2003.1694

Townsend-Small, A. & Czimczik, C.I. 2010. Carbon sequestration and greenhouse gas emissions in urban turf. *Geophysical Research Letters*, 37: L02707. https://doi.org/10.1029/2009GL041675

30. Water and residues management on a golf course, Nebraska, United States of America

Nancy Cavallaro [1], Maxine J. Levin[2], Geraldine N. Vega Pizarro[3], Bryant C. Scharenbroch[4], Anna Paltseva[5]

[1] Carbon Cycle Interagency Working Group (U.S.) CCIWG/USGCRP, Washington, United States of America

[2]University of Maryland, Department of Environmental Science and Technology, College Park, MD, United States of America

[3] USDA-Natural Resources Conservation Service, Tolland, Connecticut, United States of America

[4]University of Wisconsin-Stevens Point, Soil and Waste Resources, Stevens Point, United States of America

[5]School of Geosciences, University of Louisiana, Lafayette, LA, United States of America; Department of Landscape Design and Sustainable Ecosystems, Agrarian-Technological Institute, RUDN University, Moscow, Russian federation

1. Related practices and hot-spot

Gardens (horticulture), parks, and lawns, adequate irrigation practicesirrigation, organic mulching; Urban soils

2. Description of the case study

This case-study presents results obtained in Qian, Follet and Kimble (2010) in which soil organic carbon (SOC) changes, sequestration, and decomposition rates were studied, using C isotope techniques in four different grass plots on a golf course in Nebraska, United States, from 2002-2005:

1. Fine fescue (*Festuca spp.*), irrigated;
2. Fine fescue (*Festuca spp.)*, non-irrigated;
3. Kentucky bluegrass (*Poa pratensis L.*), irrigated; and
4. Creeping bentgrass (*Agrostis palustris Huds.*), irrigated

Treatments included irrigation (presume central pivot, 90-100 percent evapotranspiration (ET) applied as needed) in three of the four grass plots, weekly mowing, and annual fertilization. All plots were fertilized with 150 kgN/ha annually from 2002 to 2005. Grass clippings remained after mowing, creating a mulch layer that naturally decomposed and returned to the soil.

After 4 years of establishment, turfgrass provided 17 to 24 percent of SOC at 0 to 10 centimeters (cm) and 1 to 13 percent at 10 to 20 cm. All turfgrasses showed a significant carbon sequestration of 0.32 to 0.78 tC/ha/yr. Irrigation increased the soil organic carbon input and decomposition. Irrigated fine fescue (*Festuca spp.*) had the highest carbon input with 3.35 tC/ha/yr and the highest carbon decomposition rate with 2.61 tC/ha/yr.

3. Context of the case study

The study was conducted in Arbor Links Golf Course in Nebraska City, Nebraska in Central United States 2002-2005. The site was initially a prairie grassland dominated by various warm- and cool-season plants. In the 1860s, the area was cleared and planted to cropland for wheat, oat, sorghum, and corn production. In 2000, the area was converted to a golf course for recreation. The annual precipitation rate is 860 mm. High temperature ranges from 0 to 31 °C, low temperature ranges from −21 °C to 19 °C. Soil type were Mollisols with dark surfaces Mollisols (Aksarben silty clay loam; a fine, smectitic, mesic Typic Argiudoll) indicating native high organic matter, with an average pH of 6.8. Some areas were likely prone to soil erosion before conversion but that is not documented in the case study.

4. Possibility of scaling up

The practices used in this study can be scaled up and applied in many urban settings, however, these practices may be specific to cool temperate moist IPCC climate zones, such as Central North America (35 °-50 °N 85 °-105 °W) and most crops grown in cool, temperate, or moist habitats are C3 plants, as are all trees.

5. Impact on soil organic carbon stocks

There is potential for SOC sequestration with the turfgrasses in the study. SOC was measured from cores at 0-10 cm and 10-20 cm for different grass types and irrigation at the beginning and end of four-year period. Values are given below for each depth and the total for 0-20 cm.

Using C isotopic analysis, the study was able to determine the proportion of soil carbon derived from the turfgrass. Four years after establishment, about 17 to 24 percent of SOC at 0 to 10 cm and 1 to 13 percent from 10 to 20 cm was derived from turfgrass. Irrigated fine fescue added the most SOC (3.35 tC/ha/yr) to the 0- to 20-cm soil profile but also had the highest rate of SOC decomposition (2.61 tC/ha/yr). The corresponding additions and decomposition rates for unirrigated fine fescue, Kentucky bluegrass, and creeping bentgrass in the top 20-cm soil profile were 1.39 and 0.87, 2.05 and 1.73, and 2.28 and 1.50 tC/ha/yr, respectively.

Irrigation increased both SOC input and decomposition but when averaged overall it increased SOC sequestration.

As shown in Table 106, all turfgrasses exhibited significant C sequestration (0.32–0.78 t/ha/yr) during the first 4 years after turf establishment. The net C sequestration rate was higher, however, for irrigated fine fescue and creeping bentgrass than for Kentucky bluegrass. The Kentucky bluegrass and the unirrigated fine fescue showed no significant sequestration at the 10-20cm depth whereas the irrigated fine fescue showed little increase in carbon at 0-10cm but significant a significant increase at 10-20cm. This likely relates to rooting depth as influenced by grass species and water availability. Organic nitrogen levels were not significantly changed over the course of this study but models suggest that total N will increase over time (Qian *et al.*, 2003).

Table 106. Evolution of soil carbon stocks in the 4-year study in Nebraska City, Nebraska, United States of America

Values extracted from Qian et al. (2010)

Soils are Mollisols, climate is Cool Temperate Moist according to the IPCC (IPCC, 2014).

Treatment	Baseline SOC (tC/ha)	Additional annual C storage (tC/ha/yr)
0-20 cm		
Fine fescue, unirrigated	27.8	0.52
Fine fescue, irrigated	30.7	0.74
Kentucky bluegrass, irrigated	36.3	0.32
Creeping bentgrass, irrigated	30.2	0.77
0-10 cm		
Fine fescue, unirrigated	16.9	0.68
Fine fescue, irrigated	19.4	0.06
Kentucky bluegrass, irrigated	19.3	0.52
Creeping bentgrass, irrigated	16.2	0.67
10-20 cm		
Fine fescue, unirrigated	10.9	-0.16
Fine fescue, irrigated	11.4	0.68
Kentucky bluegrass, irrigated	17.0	-0.2
Creeping bentgrass, irrigated	13.9	0.1

6. Other benefits of the practice

6.1. Benefits for soil properties

There is no data about benefits on soil properties, other than carbon stocks.

6.2 Minimization of threats to soil functions

Table 107. Soil threats

Soil threats	
Soil erosion	Turfgrass prevents soil erosion (Monteiro, 2017).
Nutrient imbalance and cycles	Returning the clippings to the soil after mowing helps with the recycling of nutrients and reduces the amount of fertilizer needed (Qian *et al.*, 2003). Measurement of total N, P, and K were not reported in this study, but models suggest that long-term sustainability is enhanced by this practice (Qian *et al.*, 2003).
Soil compaction	No information was provided by this study.
Soil water management	Turfgrass promotes water infiltration with the improvement of soil structure and porosity compared to undeveloped bare ground (Monteiro, 2017).

6.3 Socio-economic benefits

A well-maintained lawn adds economic value to the property (Photo 69). However, this study did not document economic benefits of well-maintained golfing greens.

7. Potential drawbacks to the practice

7.1 Tradeoffs with other threats to soil functions

Table 108. Soil threats

Soil threats	
Nutrient imbalance and cycles	Although this was not considered in the study, constant management in golf fields can cause loss of nutrients, such as P. Nutrient loss will require the use of more fertilizer to maintain the quality of the turfgrass (Monteiro, 2017). Mulching could mitigate this loss but is not covered in this case study.
Soil contamination / pollution	Although this was not considered in the study, excess use of fertilizers with business as usual on high maintenance golf lawns can cause nutrients leaching and can be a source of groundwater pollution (Qian *et al.*, 2003).
Soil compaction	Golf lawn maintenance is weekly for esthetics of courses. Although it was not considered in this study, lawn management can cause soil compaction, such as by using standard motorized equipment to mow the grass (Matthieu *et al.*, 2011).

7.2 Increases in greenhouse gas emissions

There is no data for greenhouse gas emissions in the study.

7.3 Conflict with other practice(s)

Conservative mulching on golfing fields could be in conflict with standard practices of removing mulch to have a groomed course. This is standard in the esthetics of golfing greens to have no visible mulching as if a carpet.

8. Recommendations before implementing the practice

Golf lawns require intensive management (Photo 69). Removal of grass clippings, and constant mowing reduce the organic carbon and soil nutrients. The constant application of fertilizer to maintain the turf quality can contribute to groundwater pollution and greenhouse gas emission. It is recommended to implement conservative soil erosion and mulching management practices to reduce the impact of lawn management in the environment. However, there is no documentation as to whether these practices were permanently adopted as a follow-up to the research study. Returning the grass clippings to the soil during mowing is a good conservation practice. Compost and compost extraction applications could be used to improve the turf quality, and to limit the use of synthetic fertilizers. This was partially considered in the case study but could be expanded to other products and techniques of application. Several authors–Johnson (2018); Glab *et al.* (2018); and Boulter, Boland, and Trevors (2000, 2002)–reported that if managed properly, and considering the soil type, a mature compost application improves the soil quality by increasing the micro-biodiversity. Micro-biodiversity increases soil potential to sequester carbon; improves soil bulk density, porosity, and water infiltration; reduces soil compaction; increases available water capacity; suppresses turfgrass diseases; and supports turfgrass growth.

9. Potential barriers for adoption

Table 109. Potential barriers to adoption

Barrier	YES/NO	
Biophysical / Natural resource	Yes	Lawns require intensive management and water irrigation. The amount of water needed to maintain the good quality of lawns represents a natural resource conservation concern (Monteiro, 2017).
Economic	Yes	Lawns require intensive management and water irrigation. (Monteiro, 2017). The amount of water needed to maintain the good quality of lawns represents an economic concern.

Photo

Photo 69. Golf lawn in Southbride, Massachusetts, United States of America. Establishment of golf courses is a common practice for urban soils

References

Boulter, J.I., Boland, G.J. & Trevors, J.T. 2000. Compost: a study of the development process and end-product potential for suppression of turfgrass disease. *World Journal of Microbiology & Biotechnology*, 16: 115-134. https://doi.org/10.1023/A:1008901420646

Boulter, J.I., Trevors, J.T. & Boland, G.J. 2002. Microbial studies of compost: bacterial identification, and their potential for turfgrass pathogen suppression. *World Journal of Microbiology & Biotechnology,* 18: 661-671. https://doi.org/10.1023/A:1016827929432

Glab, T., Zabinski, A., Sadowska, U., Gondek, K., Kopec, M., Mierzwa-Hersztek, M. & Tabor, S. 2018. Effects of co-composting maize, sewage sludge, and biochar mixture on hydrological and physical qualities of sandy soil. *Geoderma*, 315: 27-35. https://doi.org/10.1016/j.geoderma.2017.11.034

IPCC. 2014. *Climate change 2014: mitigation of climate change: Working Group III contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. O. Edenhofer, R. Pichs-Madruga, Y. Sokona, J.C. Minx, E. Farahani, S. Kadner, K. Seyboth, A. Adler, I. Baum, S. Brunner, P. Eickemeier, B. Kriemann, J. Savolainen, S. Schlömer, C. von Stechow & T. Zwickel (Eds.) New York, NY, Cambridge University Press. 1435 pp.

Johnson, D. 2018. *Compost for soil regeneration: Johnson-Su composting bioreactor.* [online] Regeneration International. [Cited 13 October 2020] https://regenerationinternational.org/bioreactor/

Matthieu, D.E., Bowman, D.C., Thapa, B.B., Cassel, D.K. & Rufty T.W. 2011. Turfgrass Root Response to Subsurface Soil Compaction. C*ommunications in Soil Science and Plant Analysis Journal*, 42(22): 2813–2823. https://doi.org/10.1080/00103624.2011.622826

Monteiro, J.A. 2017. Ecosystem services from turfgrass landscapes. *Urban Forestry and Urban Greening,* 26: 151–157. https://doi.org/10.1016/j.ufug.2017.04.001

Qian, Y., Follet, R.F. & Kimble, J.M. 2010. Soil Organic Carbon Input from Urban Turfgrasses. *Soil Science Society of America Journal,* 74(2): 366-371. https://doi.org/10.2136/sssaj2009.0075.

Qian, Y.L., Bandaranayake, W., Parton, W.J., Mecham, B., Harivandi, A.M. & Mosier, A.R. 2003. Long-term effects of clipping and nitrogen management in turfgrass on soil carbon and nitrogen dynamics: The CENTURY model simulation. *Journal of Environmental Quality*, 32(5): 1694–1700. https://doi.org/10.2134/jeq2003.1694.

31. Maintenance of marshlands in urban tidal wetlands in New York City, United States of America

Yiyi Wong[1], Ellen Kracauer Hartig[2,3], Nancy Cavallaro[4], Maxine J. Levin[5]

[1]Wong & Associates, United States of America LLC, New York,United States of America

[2]City of New York Parks & Recreation

[3]City University of New York (CUNY), United States of America

[4]Carbon Cycle Interagency Working Group (U.S.) CCIWG/USGCRP, Washington, DC,United States of America

[5]University of Maryland, Dept. of Environmental Science and Technology, College Park, MD,United States of America

1. Related practices and hot-spots

Wetland Conservation, Salt Marsh Restoration; Urban soils, Wetlands

2. Description of the case study

The objective of this study is to examine the effects of short and long-term climate and anthropic controls on the different types of carbon sequestration in urban tidal marshes. Coastal wetlands which include coastal and riverine marshes, seagrass meadows, mangroves, and tidal wetlands are known as "blue carbon" wetlands (Peteet *et al.*, 2020). Although these ecosystems are small in area, they are among the world's largest carbon sinks per unit area because of their continuous carbon influx (Chmura *et al.*, 2003; Bridgham *et al.*, 2006). In North America alone, tidal wetlands remove up to 27 Tg C/yr from the atmosphere and have a net uptake of as much as of 17 Tg C/yr (Windham-Meyers *et al.,* 2018 from the State of the Carbon Cycle Report (SOCCR-2)). However, as these marshes are threatened with sea-level rise and increased storm frequency, we may lose up to 45% of these carbon sinks by the end of the century (Craft *et al.,* 2009).

Historically, tidal wetlands in urban areas were seen as mosquito-ridden wastelands to be drained and filled. Since the 1970s, United States environmental policies have recognized the ecological function and socio-economic value of wetlands (Section 404 of the Clean Water Act). Many states including New York promulgated their own legislation (NYS Article 25 of the Environmental Conservation Law, 6 NYCRR Part 661, Tidal

Wetlands Act). While current legislative and public policies have effectively reduced direct dredge and fill activity in wetlands, we now realize that wetland peat and soils accumulate and store a significant amount of carbon belowground in thick organic layers and these are at risk from indirect anthropogenic activity.

Despite the changes in environmental policies in urban municipalities, historic imagery indicates that marshes are continuing to recede at the shoreline edge, and internal pools are becoming larger and coalescing. The rapid deterioration from sea level rise and other human-induced stressors - most importantly, overloading of nutrients from sewage plant effluent and other sources - has further exacerbated this trend (Hartig *et al.,* 2002; Krause *et al.,* 2019; Watson *et al.,* 2014; Wigand *et al.,* 2014;Deegan *et al.,* 2012; Bernice *et al.,* 2016).

Without active management, our urban marshes will not keep pace with sea level rise and we will lose ecosystem functions and values such as erosion control, storm impact mitigation, habitat for birds, fish, and other wildlife, water quality improvement by sequestering toxins and heavy metals through sedimentation (Peteet *et al.,* 2018; Callaway *et al.,* 2012), carbon sequestration, and opportunities for recreation and education. Therefore, prevention of additional marsh loss, maintenance of current marsh area, and potentially increasing marsh habitat through restoration, conservation, and enhancement is important (Hartig *et al.*, 2002).

Determining the amount of C sequestered and methods to increase accumulation of C within urban coastal ecosystems will enable coastal communities to develop adaptation, mitigation, and resilience strategies to combat the impacts of global climate change. Climate adaptation plans should include (Peteet *et al.,* 2020; Haight *et al.,* 2019; NYC Parks, 2018) :

1. thin mineral layer placement in subsiding and/or eroded former marsh areas where appropriate; in NYC we suggest 15-30 cm sand placement across the marsh surface,
2. potential dam removal to enhance sedimentation rates,
3. enhanced nitrogen removal, and
4. conservation of proposed locales for planned marsh migration inland.

Preserving the existing blue carbon storage of wetlands and possibly increasing their capacity can reduce our carbon footprint while creating an additional carbon sink.

3. Context of the case study

The study site is Piermont Marsh, located 40 km north of New York City, along the Hudson River (Figure 12). Piermont Marsh has an average depth of 10 m and is 109 ha in size. It is one of four National Estuarine Research Reserve (NERR) sites within New York State. Through well documented, long-term historical and scientific data, pollen and sedimentation rates show settlement by Europeans around 1700, reforestation in the 20th Century, and the development of urban areas adjacent to the river (Pederson *et al.,* 2005).

The southern Hudson River Valley climate regime is similar to that of New York City where mean temperatures for January and July are -0.4 °C and 25 °C, while the mean annual precipitation is 1080 mm (Central Park Observatory from www.ncdc.noaa.gov; Kunkel *et al.*, 2013).

Figure 12. Vegetation image of Piermont Marsh, NY courtesy of Hud. Vegetation is now dominated by the invasive Phragmities australis (common reed) and Typha angustifolia with a remainder of native grasses of Spartina patens and Distichlis spicata (native high marsh grasses)

4. Possibility of scaling up

The practices used in this study can be scaled-up and used in many urban settings along coastal waterways, riverine locales, lakes, and reservoirs. The most effective and practical methods of preserving and restoring coastal marsh functions and extent will depend on geographical locations and site specific requirements, but the general recommendation of practices include 1) marsh restoration through thin mineral layer placement and buildout into unvegetated intertidal zones where appropriate, 2) potential dam removal to enhance sedimentation rates, 3) enhanced mitigation of reactive nitrogen inputs and other pollutants, and 4) conservation of proposed locales for planned marsh mitigation inland (Peteet *et al.*, 2020; Peteet *et al.*, 2018; Bulesco *et al.*, 2019; Haight *et al.*, 2019, NYC Parks, 2018, Davis *et al.*, 2017).

5. Impact on soil organic carbon stocks

For this study, field data collection and laboratory experiments were carried out in Piermont Marsh located along the Hudson River, upriver from New York City. A 13.7 m sediment core was extracted from the northern portion of Piermont Marsh using a modified Livingston piston corer and an additional core was retrieved using a Dachnowski side-opening corer (Peteet *et al.,* 2020; Photo 70 and Photo 71). Carbon dating was conducted in previous studies using a 1 m and 15 m core (Peteet *et al.*, 2018).

Cores were used to reconstruct rates of accumulation in mineral and organic matter within this marsh over the last three centuries. A detailed accretion model using density measurements shows historic land use, pollution markers, geochemical, isotopic and paleobotanical analyses. Piermont Marsh along with many other marshes in the New York City vicinity have degraded and been losing ground due to sea level rise and reduced sediment inputs as well as pollution.

Cores were sliced into 4.0 cm thick segments and subsamples were burned at 550 degrees C for two hours. The Loss-on-Ignition (LOI) and bulk density measurements were multiplied to calculate ash-free bulk density. Carbon content was calculated from the ash-free bulk density by using a conversion factor of 0.5 - appropriate specifically for sedge peat (Loisel *et al.,* 2014). Chemical analyses of the cores were determined by taking 3.0 grams of dried sediment using protocol developed by Croudace and Rothwell (2010) and scanning the surface of a smoothed peat core via Croudace *et al.* (2006).

Results of the data show C burial in Piermont Marsh varies greatly with climate. Because the potential C sequestration depends on the climatic and environmental conditions, the amount of sequestration or additional storage achieved by maintenance of this and similar coastal marshes is highly variable. Although Piermont Marsh is only 109 ha, Peteet *et al.* (2020) has found that throughout the history of Piermont Marsh, the range of C sequestration ranged from 0.5 to almost 3 tC/ha/yr (Table 110). They also found that warm and dry climate coincided with very low carbon storage and an increase in charcoal. It was speculated that as soils warmed, conditions favored the growth of upland vegetation such as pine and hickory in the uplands and less marsh production due to lower moisture content at the site. During warmer periods, in addition to fire, upland vegetation also respired more C into the atmosphere than marsh vegetation could incorporate into biomass because of the longer growing season. Conversely, climate change leading to extremely cold temperatures, as seen during the Little Ice Age, caused a decrease in C sequestration, possibly due to cooler summers and a shorter growing season (Peteet *et al.,* 2020). Recent decades have shown C sequestration rates of only 0.5 tC/ha/yr similar to the historic lows during long, dry, cold winters, despite the warmer and wetter current climate.

When climate conditions are optimal, where the marsh is growing and accumulating sediment, the potential carbon storage is almost 3 tC/ha/yr. When climate conditions are extremely cold or dry, where growth of vegetation is restricted, potential carbon storage is only 0.50 tC/ha/yr. However, even under these sub-optimal conditions, C sequestration is still significant, provided the marsh area can be maintained in the face of sea level rise.

Overall, nitrogen (N) inputs have increased since colonial times due to agricultural and sewage inputs (Watson *et al.,* 2014; Bulseco *et al.,* 2019). With nitrogen no longer a limiting factor for underground roots and rhizomes, the plants no longer formed dense root systems to accumulate N and the peat tended to disintegrate, with subsequent decreases in belowground C retention (Deegan *et al.,* 2012; Wigand *et al.,* 2014). With sea level rise and sedimentation rates unable to keep pace, the capacity of the marsh for C sequestration is also diminishing.

Table 110. Evolution of SOC stocks with wetland conservation

Baseline C Sequestration Rate (tC/ha/yr)	Current Land Area (ha)	Potential C Sequestration Rate (tC/ha/yr)	Depth (cm)	Duration (Years)	Reference
<0.5	109	2.5	0-100*	100*	Peteet *et al.* (2020)

* SOC measurements depth taken with depths to from 0-2.5 meters and 0-13.7 meters (two cores). However, for this paper, stocks were recalculated for the last 100 years or 1.0 meter to reflect the urban changes due to NYC expansion.

6. Other benefits of the practice (preserving, enhancing, and restoring tidal wetlands)

6.1. Benefits for soil properties

Physical properties

Preserving, enhancing, and restoring tidal wetlands allows for long term organic matter accumulation within the soil profile. Maintaining and/or increasing belowground root structure enables long term peat storage (Bulseco *et al.,* 2019). Peat layers bind to C, heavy metals, and other toxins, preventing them from entering the atmosphere and water column (Peteet *et al.,* 2018).

Chemical properties

Preserving, enhancing, and restoring tidal wetlands, prevents C biomass conversion to greenhouse gases such as methane and carbon dioxide (Callaway *et al.,* 2012; Chmura *et al.,* 2003). Enhanced tidal wetland protection acts to reduce delivery of N to more aquatic habitats (Valiela and Cole, 2002).

Biological properties

Preserving, enhancing, and restoring tidal wetlands increases above ground and belowground biomass of wetland vegetation (Callaway *et al.,* 2012).

6.2 On minimizing threat to wetlands and climate change

Table 111. Soil threats

Soil threats	
Soil erosion	Maintenance of existing wetlands through various recommended practices mitigates erosion and losses of these soils from storm events.
Nutrient imbalance and cycles	Anthropogenic activities such as sewage effluent have caused marshes in and surrounding NYC to be inundated with increased N concentrations (Bernice *et al.*, 2016). Increased N concentrations are suspected of reducing peat cohesiveness (Turner, 2011). When N is not a limiting factor, root growth is reduced and fibrous peat networks containing high concentrations of C are weakened. Among the results of increased N concentrations in coastal waters is the reduction of belowground C sequestration through loss of dense root formation and a release of C into the water column (Krause *et al.*, 2019; Watson *et al.*, 2014; Wigand *et al.*, 2014; Deegan *et al.*, 2012).
Soil salinization and alkalinization	With sea level rise, inland coastal forests are increasingly vulnerable to salt spray and saltwater flooding. Maintenance of saltwater marshes reduces this risk by mitigating storm surges and inland flooding from sea water.
Soil acidification	Generally, increased soil organic matter often mitigates the effects of soil acidity or otherwise improves soil pH. However, there are exceptions such as Sulfaquents with histic epipedons.
Soil biodiversity loss	Maintaining and enhancing tidal wetlands preserves its biodiversity as well as that of nearby other coastal habitats they buffer.
Soil water management	Improved water quality by preventing re-release of heavy metals and toxins buried in peat and organic layers (Peteet *et al.*, 2018).

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

In the North-eastern portion of the U.S., coastal high marsh vegetation such as *Spartina patens* (salt hay grass) and *Distichlis spicata* (spike grass), and in tidal freshwater systems (such as the more northern portions of the Hudson River), *Typha latifolia* (cattail), were farmed to feed cattle "salt hay". Although this is no longer practiced, it is an alternative feed option that could be used when other feed becomes limited or expensive due to climate and other stressors. (Peteet *et al.,* 2020; Burdick *et al.,* 2018; Vincent, Burdick and Dionne, 2014; Adamowicz *et al.*, 2004).

6.4 Mitigation of and adaptation to climate change

Wetlands are natural carbon sinks. With recent intensification and frequency of storms, due to climate change, managers are concerned that marshes will erode and disappear. However, wetlands have been shown to both accumulate and/or release carbon. Following hurricanes and similar large-scale storm events, organic rich, peat, and sediment complexes within wetlands can cause carbon distributions to shift when: 1) wave action strips peat from wetlands, 2) wave action disintegrates peat and releases its carbon components into the water column and/or atmosphere, and 3) high tides redistribute sediment containing organic carbon throughout the marsh (McKee and Cherry, 2009).

For example, in Jamaica Bay, NY, Wang *et al.* (2018) reported the effect of Hurricane Sandy on marsh islands varied from a 30 mm loss to a 15 mm gain in elevation. Additionally, Yeates *et al.* (2020) found that Hurricane Sandy resulted in elevation gain and loss depending on the orientation of the feature (e.g. south or north) and the angle of storm impact.

The mapped area of tidal marshes in the New York City area as of 2012 is estimated to be between 1 600 and 3 000 hectares (City of New York, 2012). While the carbon contents and sequestration rates for tidal marshes is highly variable, the top meter of Piermont Marsh, which covers only 109 ha, sequesters a total carbon content of 353 000 tons (Chen and Peteet, 2020) (equivalent to 1 294 333 tCO_2eq) or nearly 3 percent of New York City's annual total greenhouse gas emissions of 50 692 927 tCO_2eq (NYC Mayor's Office of Sustainability, 2019). This is the amount of carbon that could be considered at risk from future extreme weather events if conservation practices are not carried out. While some managers are concerned about the costs of being "green," maintaining and expanding environmental conservation programs can actually financially benefit the community. For example, municipalities and climate managers may submit an application for their wetlands to be included as a source of carbon stock within emerging carbon markets or they may sell their wetlands as carbon offsets for a public or private project. Furthermore, prevention of additional marsh loss, maintenance of current marsh area, and potentially increasing marsh habitat through restoration, conservation, and enhancement yields additional socio-economic benefits (Hartig *et al.*, 2002).

6.5 Socio-economic benefits

Worldwide, wetlands contribute up to USD 14.9 trillion in functional ecosystem services and value from ecotourism to flood prevention to commercial fisheries (Adair, 2020). Maintaining and increasing coastal wetlands are a more economical solution in preventing destruction related to increased hurricanes, typhoons, and Nor'easters. A Nor'easter is named for storm systems that are generated along the Eastern Coast of North America by winds typically coming from the northeast. These storms may occur at any time of the year but are seen most frequently from September to April (NOAA 2020). Hard engineering structures such as seawalls, tide-gates, jetties, and groins, are substantially more expensive than soft-structures such as restored wetlands because natural ecosystems when established are self-sustaining (Pilkey and Dixon, 1996). For example, New York City invested USD 1.5 billion to protect and restore wetlands to save the city anywhere from USD 3-8 billion in updating and/or building new water treatment plants and storm-protection infrastructure, creating new protection policies, and reducing construction costs (Adair, 2020).

7. Potential drawbacks to the practice

7.1 Tradeoffs with other climate threats

Table 112. Soil threats

Soil threats	
Soil contamination / pollution	Wetlands may increase the concentration of contaminants trapped within the wetland as they sequester contaminants from stormwater runoff.
Soil acidification	Generally, increasing the amount of organic matter accumulated in the wetland will maintain or provide a buffer for further reduction in soil pH. However, in instances where Sulfaquents are involved, organic amendments may further reduce soil pH.
Soil biodiversity loss	The application of a thin mineral layer to a wetland may reduce soil microbial communities and biodiversity if the layer applied is too thickly. Appropriate thickness of the thin mineral layer depends on the soil type and site location.

7.2 Increases in greenhouse gas emissions

Piermont Marsh was calculated to store a total of 353 000 tons of carbon based on the average marsh depth of 10 meters and 109 ha (Chen and Peteet 2020). This study did not include the measurement needed to determine the Global Warming Potential (GWP). However, in zones where the soil water is sufficiently saline, methane emissions would be negligible (Poffenbarger, Needelman and Megonigal, 2011). Nitrous oxide emissions may be enhanced in expanded areas of tidal wetlands but generally would not offset the benefit of reduced carbon emissions (Bulesco *et al.*, 2019; Doroski, Helton and Vadas, 2019).

7.3 Conflict with other practice(s)

Rebuilding or expanding wetlands might occur at the expense of forest lands. Thus, it is recommended that any planned expansion take into consideration effects on other valuable resources.

7.4 Negative impact on production

No negative impact.

8. Recommendations before implementing the practices

Careful planning is always important before implementation of practices. In the case of adoption of thin layer placement (practice #1) – managers need to consider, for example if plants will be expected to poke through from below, or if new planting will be done (a more expensive and labor-intensive option). How thin the layer needs to be varies depending on extent of subsidence and the source of sediments. In all cases, post-restoration monitoring should be part of the planning process.

Managers should review the jurisdictional requirements regarding environmental policies covering permitting and other regulations. Wetland restoration should only be conducted to increase habitat value and function and not at the expense of other high value habitats. For example, within the United States, dredge and fill construction activity within tidal wetlands are covered by the Clean Water Act, Section 404 (federal). While federal wetland regulation is limited to the wetland itself, many states have added wetland protections. For example, New York State and many of its municipalities regulate a jurisdictional adjacent (buffer) area that can extend to a maximum of 91.44 m beyond the delineated wetland boundary (NYSDEC, 1973).

New York City has already implemented design protocols whereby new low marsh and high marsh installations are designed at the high-end elevations to account for accelerated sea-level rise. As part of its Comprehensive Long-Term Sustainability Plan, NYC government agencies use the sea level rise projections of the NPCC (New York City Panel on Climate Change) for planning purposes (Horton *et al.*, 2015).

Policy makers may also consider registering their wetlands as carbon credits/carbon offsets within applicable carbon markets. If the wetland is storing a significant amount of carbon, it may be developed as a carbon offset according to voluntary market Standards or compliance market Standards according to regional carbon accounting and eligibility rules. The feasibility of the project would depend on the carbon revenues at the time, the timing in which verification can be accomplished, and when the carbon credit is sold. Profits from carbon trading may be reinvested in restoring or maintaining wetlands.

9. Potential barriers for adoption

Table 113. Potential barriers to adoption

Barrier	YES/NO	
Biophysical / Natural resource	Yes	Expansion of a wetland may depend on the available sources of mineral component and native vegetation. Wetland expansion might occur at the expense of forest lands.
Cultural/Social	Yes	Depends on the social will of the location.

Barrier	YES/NO	
Economic	Yes	Wetland restoration and expansion may be costly.
Institutional	Yes	Depends on the political and social will of the location, as well as legal jurisdiction.

Photos

Photo 70. Members of Dr. Dorothy Peteet's New Core Laboratory extracting a peat core using the Dachnowski corer (a.k.a. Russian peat corer)

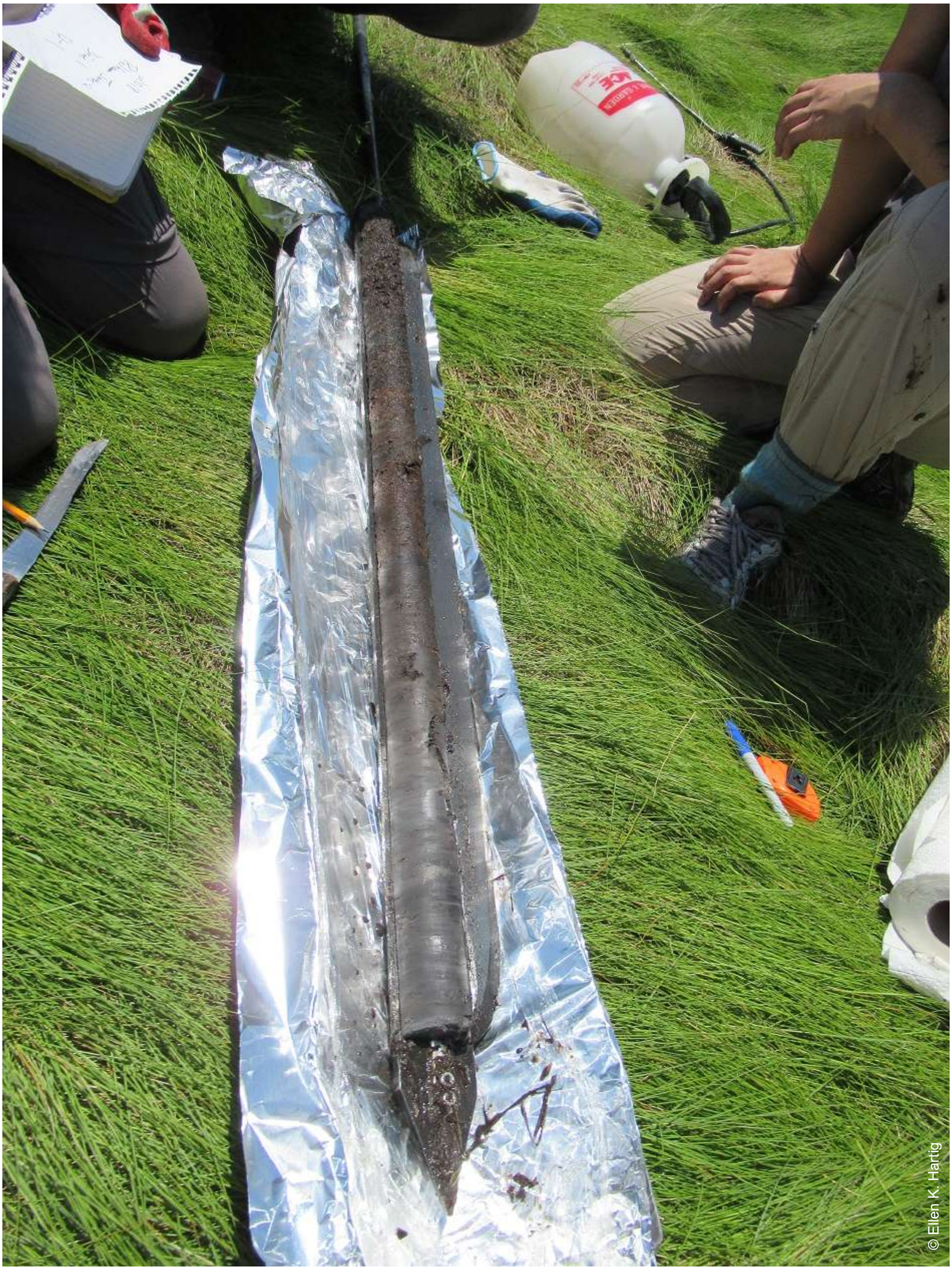


Photo 71. Example of a wetland core collected using the Dachnowski corer (a.k.a. Russian corer)

References

Adair, S. 2020. What's a duck marsh really worth? [online] Ducks Unlimited. [Accessed 19/12/2020]. https://www.ducks.org/conservation/waterfowl-habitat/whats-a-duck-marsh-really-worth.

Adamowicz, S.C., Roman, C.T., Taylor, G., O'Brien, K. & James-Pirri, M.J. 2004. Initial response of salt marshes to ditch plugging and pool creation (Maine). Ecological Restoration, 22(1): 53–54.

Bernice, R., Gordon, A.L., John, M., Robert, C., Zappa, C.J. & Parris, A.S. 2016. Resilience Indicators and Monitoring: An Example of Climate Change Resiliency Indicators for Jamaica Bay. In E.W. Sanderson, W.D. Solecki, J.R. Waldman & A.S. Parris (Eds.) Prospects for Resilience: Insights from New York City's Jamaica Bay, pp. 141–165. Washington, DC, Island Press/Center for Resource Economics. (also available at https://doi.org/10.5822/978-1-61091-734-6_7).

Bridgham, S.D., Megonigal, J.P., Keller, J.K., Bliss, N.B. & Trettin, C. 2006. The carbon balance of North American wetlands. Wetlands, 26(4): 889–916. https://doi.org/10.1672/0277-5212(2006)26[889:TCBONA]2.0.CO;2

Burdick, D., Adamowicz, S., Moore, G., Peter, C., Batchelder, D., Pau, N. & Wilson, G. 2018. Pilot efforts to mitigate ditching impacts at a northeast National Wildlife Refuge. (also available at: https://estuaries.org/download/summit/2018/proceedings/session_8/Burdick.pdf)

Bulseco, A.N., Giblin, A.E., Tucker, J., Murphy, A.E., Sanderman, J., Hiller-Bittrolff, K. & Bowen, J.L. 2019. Nitrate addition stimulates microbial decomposition of organic matter in salt marsh sediments. Global Change Biology, 25(10): 3224–3241. https://doi.org/10.1111/gcb.14726

Callaway, J.C., Borgnis, E.L., Turner, R.E. & Milan C.S. 2012. Carbon sequestration and sediment accumulation in San Francisco Bay tidal wetlands. Estuaries Coasts, 35: 1163–81.

Chen, M. & Peteet, D. Comparing Piermont Marsh blue carbon storage capacity to our carbon footprint. Columbia University Earth Institute Spring 2020 Research Showcase. (also available at: https://www.earth.columbia.edu/sitefiles/file/students/showcase/2020/posters/Mengbi-Chen.pdf)

Chmura, G.L., Anisfeld, S.C., Cahoon, D.R. & Lynch, J.C. 2003. Global carbon sequestration in tidal, saline wetland soils. Global Biogeochemical Cycles, 17(4). https://doi.org/10.1029/2002GB001917

City of New York. 2012. New York City Wetlands Strategy. PLANYC 2030 Mayor's Office of Long-term Planning and Sustainability. (also available at: http://www.nyc.gov/html/planyc2030/downloads/pdf/nyc_wetlands_strategy.pdf)

Craft, C., Clough, J., Ehman, J., Joye, S., Park, R., Pennings, S., Guo, H. & Machmuller, M. 2009. Forecasting the effects of accelerated sea-level rise on tidal marsh ecosystem services. Frontiers in Ecology and the Environment, 7(2): 73–78. https://doi.org/10.1890/070219

Croudace, I.W., Rindby, A. & Rothwell, R.G. 2006. ITRAX: description and evaluation of a new multi-function X-ray core scanner. Geological Society, London, Special Publications, 267(1): 51–63. https://doi.org/10.1144/GSL.SP.2006.267.01.04

Croudace, W. & Rothwell, R.G. 2010. Micro-XRF sediment core scanners: important new tools for the environmental and earth sciences. Spectrosc. Eur., 22: 6–13.

Davis, D.S., P.M. Weppler, P. Rafferty, D.G. Clarke & D. Yozzo. 2017. Elders Point East Marsh Island Restoration Monitoring Data Analysis. US Army Corps of Engineers, Engineer Research Development Center (ERDC). Report no. ERDC/EL;CR-17-1. (also available at: https://erdc-library.erdc.dren.mil/jspui/handle/11681/23952)

Deegan, L.A., Johnson, D.S., Warren, R.S., Peterson, B.J., Fleeger, J.W., Fagherazzi, S. & Wollheim, W.M. 2012. Coastal eutrophication as a driver of salt marsh loss. Nature, 490(7420): 388–392. https://doi.org/10.1038/nature11533

Doroski, A.A., Helton, A.M. & Vadas, T.M. 2019. The intersection of urban pollution and saltwater intrusion: a soil core experiment. Soil Biology and Biochemistry, 131: 44-53. https://doi.org/10.1016/j.soilbio.2018.12.023

Haight, C., Larson, M. Swadek, R.K. & Hartig, E.K. 2019. Toward a Salt Marsh Management Plan for New York City: Recommendations for Strategic Restoration and Protection In Perillo, G.M.E., Wolanski, E., Cahoon, D.R., Hopkinson, C.S. (Eds.) Coastal Wetlands: An Integrated Ecosystem Approach. Chapter 29, 2nd Edition, Elsevier, Amsterdam. pp 997-1022.

Hartig, E. K., Gornitz, V., Kolker, A., Mushacke, F. & Fallon, D. 2002. Anthropogenic and climate-change impacts on salt marshes of Jamaica Bay, New York City. Wetlands, 22(1): 71-89. https://doi.org/10.1672/0277-5212(2002)022[0071:AACCIO]2.0.CO;2

Horton, R., Little, C., Gornitz, V., Bader, D. & Oppenheimer, M. 2015. New York City Panel on Climate Change (NPCC) 2015 Report. Chapter 2: Sea Level Rise and Coastal Storms. Annals of the New York Academy of Science, 1336: 36-44. https://doi.org/10.1111/nyas.12593

Krause, J.R., Watson, E.B., Wigand, C. & Maher, N. 2020. Are Tidal Salt Marshes Exposed to Nutrient Pollution more Vulnerable to Sea Level Rise? Wetlands, 40(5): 1539–1548. https://doi.org/10.1007/s13157-019-01254-8

Loisel, J., Yu, Z., Beilman, D.W., Camill, P., Alm, J., Amesbury, M.J., Anderson, D., Andersson, S., Bochicchio, C., Barber, K., Belyea, L.R., Bunbury, J., Chambers, F.M., Charman, D.J., De Vleeschouwer, F., Fiałkiewicz-Kozieł, B., Finkelstein, S.A., Gałka, M., Garneau, M., Hammarlund, D., Hinchcliffe, W., Holmquist, J., Hughes, P., Jones, M.C., Klein, E.S., Kokfelt, U., Korhola, A., Kuhry, P., Lamarre, A., Lamentowicz, M., Large, D., Lavoie, M., MacDonald, G., Magnan, G., Mäkilä, M., Mallon, G., Mathijssen, P., Mauquoy, D., McCarroll, J., Moore, T.R., Nichols, J., O'Reilly, B., Oksanen, P., Packalen, M., Peteet, D., Richard, P.J., Robinson, S., Ronkainen, T., Rundgren, M., Sannel, A.B.K., Tarnocai, C., Thom, T., Tuittila, E.-S., Turetsky, M., Väliranta, M., van der Linden, M., van Geel, B., van Bellen, S., Vitt, D., Zhao, Y. & Zhou, W. 2014. A database and synthesis of northern peatland soil properties and Holocene carbon and nitrogen accumulation. The Holocene, 24(9): 1028–1042. https://doi.org/10.1177/0959683614538073

McKee, K.L. & Cherry, J.A. 2009. Hurricane Katrina sediment slowed elevation loss in subsiding brackish marshes of the Mississippi River delta. Wetlands, 29(1): 2–15. https://doi.org/10.1672/08-32.1

NOAA (National Weather Service). 2020. What is a Nor'easter? [online]. [Cited 26 March 2021]. https://www.weather.gov/safety/winter-noreaster.

NYC Parks (City of New York Parks & Recreation). 2018. Salt Marsh Restoration Design Guidelines: Strategies for Assessment and Restoration of Resilient Urban Tidal Wetlands. (also available at: https://www.nycgovparks.org/pagefiles/132/NYCParks-SaltMarshRestorationDesignGuidelines-FINAL-20180925__5bbe25b575534.pdf)

NYC Mayor's Office of Sustainability. 2019. Inventory of New York City Greenhouse Gas Emissions in 2017. (also available at: https://www1.nyc.gov/assets/sustainability/downloads/pdf/GHG_Inventory_2017.pdf)

NYS Article 25 of the Environmental Conservation Law. 1973. 6 NYCRR Part 661, Tidal Wetlands.

Pederson, D.C., Peteet, D.M., Kurdyla, D. & Guilderson, T. 2005. Medieval Warming, Little Ice Age, and European impact on the environment during the last millennium in the lower Hudson Valley, New York, USA. Quaternary Research, 63(3): 238–249. https://doi.org/10.1016/j.yqres.2005.01.001

Peteet, D.M., Nichols, J., Kenna, T., Chang, C., Browne, J., Reza, M., Kovari, S., Liberman, L. & Stern-Protz, S. 2018. Sediment starvation destroys New York City marshes' resistance to sea level rise. Proceedings of the National Academy of Sciences, 115(41): 10281–10286. https://doi.org/10.1073/pnas.1715392115

Peteet, D., Nichols, J., Pederson, D., Kenna, T., Chang, C., Newton, B. & Vincent, S. 2020. Climate and anthropogenic controls on blue carbon sequestration in Hudson River tidal marsh, Piermont, New York. Environmental Research Letters, 15(6): 065001. https://doi.org/10.1088/1748-9326/ab7a56

Pilkey, O.H. & Dixon, K.L. 1996. The Corps and the Shores. Island Press, California and Washington D.C.

Poffenbarger, H.J., Needelman, B.A. & Megonigal, J.P. 2011. Salinity Influence on Methane Emissions from Tidal Marshes. Wetlands, 31(5): 831–842. https://doi.org/10.1007/s13157-011-0197-0

Turner, R.E. 2011. Beneath the Salt Marsh Canopy: Loss of Soil Strength with Increasing Nutrient Loads. Estuaries and Coasts, 34(5): 1084–1093. https://doi.org/10.1007/s12237-010-9341-y

Valiela, I. & Cole, M.L. 2002. Comparative Evidence that Salt Marshes and Mangroves May Protect Seagrass Meadows from Land-derived Nitrogen Loads. Ecosystems, 5(1): 92–102. https://doi.org/10.1007/s10021-001-0058-4

Vincent, R.E., Burdick, D. M., & Dionne, M. 2014. Ditching and Ditch-Plugging in New England Salt Marshes: Effects on Plant Communities and Self-Maintenance. Estuaries and Coasts, 37(2): 354–368. https://doi.org/10.1007/s12237-013-9671-7

Wang, H., Chen, Q., Hu, K., Snedden, G.A., Hartig, E.K., Couvillion, B.R., Johnson, C.L. & Orton, P.M. 2017. Numerical modeling of the effects of Hurricane Sandy and potential future hurricanes on spatial patterns of salt marsh morphology in Jamaica Bay, New York City. U.S. Geological Survey Open-File Report 2017–1016, 43 p. https://doi.org/10.3133/ofr20171016

Watson, E.B., Oczkowski, A.J., Wigand, C., Hanson, A.R., Davey, E.W., Crosby, S.C., Johnson, R.L. & Andrews, H.M. 2014. Nutrient enrichment and precipitation changes do not enhance resiliency of salt marshes to sea level rise in the Northeastern U.S. Climatic Change. https://doi.org/10.1007/s10584-014-1189-x

Wigand, C., Roman, C.T., Davey, E., Stolt, M., Johnson, R., Hanson, A., Watson, E.B., Moran, S.B., Cahoon, D.R., Lynch, J.C. & Rafferty, P. 2014. Below the disappearing marshes of an urban estuary: historic nitrogen trends and soil structure. Ecological Applications, 24(4): 633–649. https://doi.org/10.1890/13-0594.1

Windham-Myers, L., Cai, W.-J., Alin, S., Andersson, A., Crosswell, J., Dunton, K.H., Hernandez-Ayon, J.M., Herrmann, M., Hinson, A.L., Hopkinson, C.S., Howard, J., Hu, X., Knox, S.H., Kroeger, K., Lagomasino, D., Megonigal, P., Najjar, R., Paulsen, M.-L., Peteet, D., Pidgeon, E., Schäfer, K., Tzortziou, M., Wang, Z.A., Watson, E.B., Cavallaro, N., Shrestha, G., Birdse, R., Mayes, M.A., Najjar, R., Reed, S., Romero-Lankao, P. & Zhu, Z. 2018. Chapter 15: Tidal Wetlands and Estuaries. Second State of the Carbon Cycle Report. U.S. Global Change Research Program. (also available at https://carbon2018.globalchange.gov/chapter/15/).

Yeates, A.G., Grace, J.B., Olker, J.H., Guntenspergen, G.R., Cahoon, D.R., Adamowicz, S., Anisfeld, S.C., Barrett, N., Benzecry, A., Blum, L., Christian, R.R., Grzyb, J., Hartig, E.K., Leo, K.H., Lerberg, S., Lynch, J.C., Maher, N., Megonigal, J.P., Reay, W., Siok, D., Starke, A., Turner, V. & Warren, S. 2020. Hurricane Sandy Effects on Coastal Marsh Elevation Change. Estuaries and Coasts, 43(7): 1640–1657. https://doi.org/10.1007/s12237-020-00758-5

The Global Soil Partnership (GSP) is a globally recognized mechanism established in 2012. Our mission is to position soils in the Global Agenda through collective action. Our key objectives are to promote Sustainable Soil Management (SSM) and improve soil governance to guarantee healthy and productive soils, and support the provision of essential ecosystem services towards food security and improved nutrition, climate change adaptation and mitigation, and sustainable development.

Thanks to the financial support of

European Commission

Ministry of Finance of the Russian Federation

ISBN 978-92-5-134899-4

9 789251 348994

CB6605EN/1/09.21