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The role of soils in provision of energy

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Summary

Soils have both direct and indirect impacts on available energy, but energy provision, in itself, has direct and indirect impacts on soils. Burning peats provides only ~0.02% global energy supply yet emits ~(0.7-0.8)% carbon losses from land use change and forestry (LUCF). Bioenergy crops provide ~0.3% energy supply and occupy ~(0.2-0.6)% harvested area. Increased bioenergy demand is likely to encourage switching from forests and pastures to rotational energy cropping, resulting in soil carbon loss. However, with protective policies, incorporation of residues from energy provision could sequester ~0.4% LUCF carbon losses. All organic wastes available in 2018 could provide ~10% global energy supply, but at a cost to soils of ~5% LUCF carbon losses; not using manures avoids soil degradation but reduces energy provision to ~9%. Wind farms, hydroelectric solar and geothermal schemes provide ~3.66% of energy supply and occupy less than ~0.3% harvested area, but if sited on peatlands could result in carbon losses that exceed reductions in fossil fuel emissions. To ensure renewable energy provision does not damage our soils, comprehensive policies and management guidelines are needed that (1) avoid peats, (2) avoid converting permanent land uses (such as perennial grassland or forestry) to energy cropping and (3) return residues remaining from energy conversion processes to the soil.

Introduction

The role of soils in energy provision is complex. Soils have both direct and indirect impacts on available energy, but methods used to provide energy, in themselves, have direct and indirect impacts on soils. These impacts can influence the evolution of the landscape and ecosystem services provided by soils (1). This occurs through four different processes associated with energy provision; acquisition of the energy source, conversion / storage, transport / transmission and end use / disposal of residues from the energy conversion process (2). Acquisition of energy from the soil itself is a direct impact of soil on energy provision; this includes burning of peat, either for heat or for production of electricity. Indirect impacts of soil on energy provision include the effects of soil fertility and water holding capacity on the potential yield of energy crops. Conversely, direct impacts of energy provision on the productivity of soils can occur through the removal of organic wastes that might otherwise have been incorporated into the soil to increase productivity (such as organic manures, residues from crop production and tree crowns), and their use instead as biomass fuels. Onshore windfarms indirectly impact soils through changes in biophysical characteristics of the soil, extraction and removal of soils to make room for turbine bases and covering of soils with foundations which excludes them from use for other purposes. Hydropower schemes similarly affect soils, often significantly changing the hydrological conditions of the surrounding areas. Solar and geothermal schemes remove the footprint of the energy generation infrastructure from other land use but have limited wider impact on soils. Here we consider the interaction between soils and energy provision, providing an estimate of the net contribution of soils to energy, and the impacts of energy provision on soils, both in terms of loss of soil carbon (C) and land area available for other uses. Oil spills and

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other pollution events are further indirect impacts of energy provision on soils, which can have profound consequences for soil productivity and its continued use in food production. However, these are not considered here as it is assumed that their impacts are temporary, with soils being remediated to restore productivity (3).

Peat extraction for energy

Global extent of peat extraction for energy

Peats are highly organic soils that occur due to historical partial decay of vegetation, usually under anaerobic conditions that slow decomposition. A functioning peatland is an area of peat that is continuing to grow and accumulate organic matter through the slow cycling of organic inputs. This requires a viable seedbank of specialised peatland species in a top layer of peat that may undergo fluctuations in conditions between anaerobic and aerobic (the acrotelm), as well as the presence of the conditions that limit C cycling. Peat extraction for fuel use has been occurring in many places around the world for centuries, and in northern treeless areas, such as Ireland and the Scottish islands, it is likely to have been occurring for millenia (4). Although peatlands cover only (3 to 4)% of the global land area (5), they store (26 to 44)% of the global soil organic C (6), so are highly vulnerable to C loss (7). It has been estimated that northern peatlands alone hold (4.55×10^{14}) t C, which is just under the amount of C held in the atmosphere (8). Globally, the annual rate of loss of the land area of active peatlands (where peat is accumulating) is estimated to be 0.1% (9) and approximately 10% of the non-tropical peat area loss can be attributed to fuel use (4). A relatively small proportion of the total peatland area (0.1% equivalent to ~$\underline{5 \times 10^3}$ km^2) has been used for peat extraction (10), so there remains a large pool of C in peatlands that could be emitted to the atmosphere.

In Europe, large scale use of peat for fuel started in the Middle Ages (1000 and 1500 AD) (1). In the Western Netherlands, peat extraction initially aimed to clear wetlands for settlements and agriculture, while the extracted peat was used as a fuel to replace the shortfall in wood fuel due to widespread deforestation (1). However, by the 17th century, peat had become a major national energy source in the Netherlands (11). The invention of peat-working machines in the 19th century allowed industrial scale extraction of peat in many areas across Europe (12). Peat extraction was a major industry in Russia up to the 1980s when competition with the coal industry resulted in its decline (13). However, there have been recent calls to revive its large-scale use in response to the increased prices of fossil fuels (13). In 2015, peat extraction in Ireland accounted for 4.1% of the country’s greenhouse gas emissions (14), and was legislated for under three pillars of energy policy; security, competitiveness and the environment (15). Although its use for energy provision is uncompetitive and is associated with loss of biodiversity, the practice continues because it provides an indigenous source of energy that reduces dependency on imports and so is important for national fuel security (15). Similarly, in Finland, peat is considered to be a natural resource that is vital for meeting national energy demands and achieving economic competitiveness (16). However, it is also understood to be an important source of biodiversity and global eco-security, so legal and policy frameworks control its use to avoid destruction of intact peatlands (16). In 2005, Finland was the highest user globally of peat for energy purposes, followed by Ireland, with both countries together accounting for 67% of global peat extraction (17). Other countries involved in large scale peat extraction were Belarus, Russia, Sweden, Ukraine and Estonia (18). The World Energy Council collated data on consumption of peat for energy purposes (10) which indicated that, in 2008, (1.7334×10^7) t peat were consumed globally, with the seven highest users (in decreasing order, Finland, Ireland, Belarus, Russia, Sweden, Ukraine and Estonia) accounting for 99% of all peat use for energy (Figure 1).

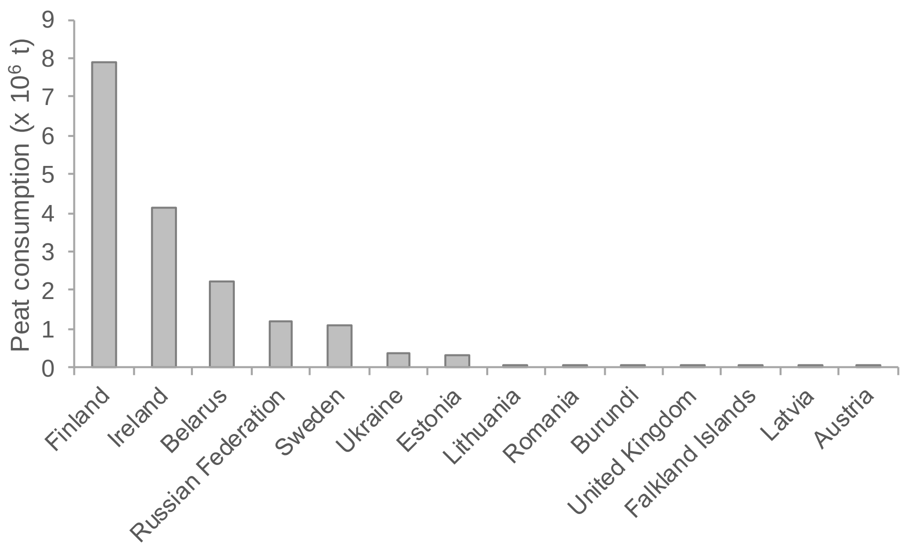

Figure 1 – Global peat consumption in 2008. Source: Adapted from values collated by the World Energy Council (10)

While European countries have led on peat consumption for fuel, the past 40 years have seen increasing interest in the use of peat for fuel in low to middle income countries. Within the tropics, the peatland resource has been more generally exploited for agricultural development (19), with peat use for energy being much less reported. The development of institutional cookstoves for use with peat fuel in Burundi was promoted in the 1980s, with incentivisation for stove use linked to the sale of peat (20). This programme was an attempt to reduce degradation of forest reserves, although deforestation has continued to be widespread (21). The cost effectiveness of peat powered electricity compared to the existing electricity supply is a critical factor for some developing countries (22), where equitable energy access is a priority (23). While the literature suggests that Rwanda will increase peat to power production in order to meet its national development targets (23), this does not align with the country's climate mitigation commitments under its Nationally Determined Commitments with the UNFCCC, in particular its commitment to low C energy from hydropower, solar power and sustainable biomass fuels (24). In neighbouring Uganda, highland peat deposits have not yet been exploited for energy, as the country has to date depended heavily on firewood from forest resources for its cooking fuel (25). However, with the national natural forest resource projected to be exhausted outside of protected areas by 2025 (25), peat may become a more attractive fuel source for communities adjacent to wetlands, although as with Rwanda this would not align with the country's climate commitments or environmental management regulations.

Methods for peat extraction and use for energy

Mechanized methods of peat cutting include auger cutters, caterpillar-tracked diggers and vacuum harvesting (26)(27). The first stage of peat cutting usually involves initial drainage of the peatland to allow the heavy extraction machinery to access the site (27)(28). This process in itself aerates the peat and so results in an increased rate of organic matter decomposition and carbon dioxide (CO_2) emissions, with associated changes to the habitat and species composition. An auger cutter digs vertically through the peat and extrudes the turfs onto the surface, while a caterpillar-tracked digger cuts peat from a vertical bank and loads it into a trailed compressor pulled by a tractor (26). For vacuum harvesting, vegetation is first removed from the surface, the upper layers of peat are milled to enhance drying to a moisture content of ~45%, and then a large vacuum extraction vehicle sucks up the loose peat (29). The impact of auger cutting is usually to damage the vegetation and compact the peat on the first cut, so impeding drainage and reducing biodiversity, but multiple cuttings result in further damage, culminating in bare peat that is vulnerable to sheet erosion (26). Caterpillar-tracked digger extraction leaves only a shallow layer of peat above the mineral soil, and the cut-over land is usually then converted to agricultural use, so the peat ecosystem is permanently destroyed (26). Similarly, vacuum harvested sites usually cannot be restored to a functioning peatland system because the viable seed bank has been removed by the extraction process (27). Different methods for restoring peat-cut areas have been attempted. These include drain blocking, damming and levelling in cut-over raised bogs that have exposed deeper fen peat layers that could be saved (30), spreading of "hay" made from cut and dried plants from a nearby intact site at the time when seeds are present (31), rhizome and sphagnum transplantation (32) and preserving and transplanting the whole acrotelm in blocks (27). However, there remain questions over the potential for successful restoration of the hydrology (33), suitability of restoration techniques (34) and re-establishment of the peatland species (35), so it is usually assumed that peat extraction will result in full destruction of the peatland habitat (36).

The extracted peat is usually burnt to produce heat, either for direct use or for use in electricity generation (37). There is also potential to use fast pyrolysis of peats to produce synthetic gas, synthetic oils and other high C materials at the same time as releasing thermal energy (37). Peat has a relatively low energy density; (1.98×10^{10}) J t^{-1} dry weight, which is similar to wood ((1.85×10^{10}) J t^{-1}) (38) but lower than coal ((2.45×10^{10}) J t^{-1}) (39). It also has a lower bulk density than wood or coal, meaning that 1 m^3 peat provides only ~15% of the heat energy provided by 1 m^3 of coal (40). Peat used for electricity generation is usually in the form of milled peat, which produces (7.8×10^9) J t^{-1} peat (15). This is lower than sod peat ((1.31×10^{10}) J t^{-1}) or peat briquettes ((1.85×10^{10}) J t^{-1}), mainly due to the higher moisture content (41). The total annual fuel provision by peat use in the European Union between 2000 and 2010 was (3.37×10^6) t oil equivalent ($(\underline{1.41 \times 10^{17}})$ J y^{-1}, with 45% being used in central heating power plants, 38% for condensing power generation, 10% in district heating and 8% in residential heating (42). This is equivalent to ~0.03% of global energy consumption in 2005 ((4.77×10^{20}) J y^{-1} (43)) and ~$\underline{0.02}$% of the global energy supply in 2018 of (5.98×10^{20}) J y^{-1} (44))

Impacts of using peat for energy

Extraction of peat provides only a small contribution to the global energy consumption, but has greater importance in individual countries, amounting to (5 to 7)% of primary energy consumption in Finland and Ireland, 1.9% in Estonia and 0.7% in Sweden (42). It provides jobs to people in rural areas and acts as a short-term energy reserve (7 to 17 months in Finland and Estonia) which is important to cover interruptions in imported energy sources (42). Therefore, while peat extraction is of low importance to global energy provision, it has higher national importance, which is why its extraction continues. However, peat extraction has negative impacts on a wide range of other ecosystem services that are provided by these soils. Potential impacts include reduced net C storage with associated climate impacts, loss of habitats and biodiversity, reduced water quality and flow regulation, loss of wild species that may be used for other purposes, decline in ecotourism and loss of the unique information contained in the paleoenvironmental record (4). The net impacts of peat extraction are difficult to quantify as peat affects a range of different services that are valued in different ways and are important in some locations but not in others. However, the use of peat in energy provision always has an adverse impact on soil C storage as the combustion of peat releases, in a short period of time, C that accumulated over thousands of years. Even if peatland restoration is successful, any C sequestration possible due to subsequent plant inputs will provide negligible compensation over the short term for the burning of peat (45).

The impacts of this on the climate are complex (see Supplementary Materials (SM.)1), but in terms of loss of stored C alone, burning of peats for energy (globally (1.7334×10^7) t peat per year (10)) emits ($(2.86$ to $3.18) \times 10^7$) t y^{-1} CO_2 (assuming C is 45 – 50% by mass (46)) (C loss = $(\underline{(7.80 \text{ to } 8.67) \times 10^6})$ t y^{-1}). These emissions can be compared to the CO_2 emissions from all land use change and forestry in the decade 2000 to 2009 (LUCF) as a way of quantifying the relative impact of using peatlands for energy provision on stored soil C. This is equivalent to ($\underline{0.7 \text{ to } 0.8}$)% of all LUCF C emissions (~(4×10^9) t y^{-1} (47)). As shown in SM1, emissions due to burning dominate the C budget and have a much more significant impact on atmospheric C (in CO_2 equivalents) than reduced C sequestration due to loss of the peatland habitat or reduced CH_4 emissions due to draining boggy areas. Degradation of the peatlands surrounding the excavated areas may also contribute further to net greenhouse gas emissions. Drainage of the site to allow access to heavy peat cutting machinery increases losses of C by aerobic decomposition of the peat surrounding the excavation; for example, in one mined peatland, CO_2 emissions were observed to increase by 100 – 400% (48), and remained at this elevated level for over 20 years (49). Losses from draining peats are significant; despite covering only 3 – 4% of the global land area (5), in 2009 net greenhouse gas emissions from all drained peats (not only from peat extractions), in the form of CO_2 and nitrous oxide (N_2O – nutrient rich soils only), were estimated to amount to (2 to 3) × 10^9 t y^{-1} CO_2 eq. (50), ~(50 – 75)% of C emissions from all LUCF (47). Drainage ditches can also provide additional sources of CH_4 due to the supply of labile dissolved organic C held under anaerobic conditions (51). The impacts of drainage depend on the initial state of the peat and the extent of the drained area around the peat cutting and drains, with reported extents of drainage ranging from a few metres to over 200 m (52)(53). Therefore, while the losses of C associated with burning the peats are estimated to be (0.7 to 0.8)% of C emissions from LUCF (47), it is likely that the net emissions including drainage and degradation of the area around the peat cutting will be significantly higher.

Production of crops for energy

Global extent of energy crops

The main terrestrial crops used for energy provision include crops that produce oils (e.g. oilseed rape, sunflowers, soya, oil palm), sugar (e.g. perennial sugar cane, sugar beet and sweet sorghum), starch (e.g. maize, wheat, cassava) and lignocellulosic biomass (e.g. wood, straw and miscanthus) (54). Crops producing oils, sugar and starch are usually grown on land that would otherwise be used for food production, while lignocellulosic biomass crops can often be grown on more marginal land, which would be less suitable for producing food due to high slopes, erosion rates or levels of contamination, or due to low fertility or availability of water (47)(55).

If sustainably managed, energy crops have potential to significantly reduce C emissions from deforestation and fossil fuel use, sequester C in degraded land, reduce emissions of black C and short-lived greenhouse gases (e.g. CH_4 and carbon monoxide) and provide opportunities for regional economic development (54). However, a key challenge is to achieve this idealised sustainable management of energy crops and avoid the high potential for negative impacts, such as loss of C and emissions of greenhouse gases from soils and vegetation, competition with food crops for productive land, reduction in biodiversity and loss of land tenure for local populations (54).

The soil impacts the potential terrestrial production of biomass energy by controlling the supply of nutrients and water to plants. This is dependent on soil texture, organic matter content, water holding capacity, structure and slope (56), factors that are reflected in the total potential terrestrial supply of biomass (54). Haberl et al. estimated that, accounting for biophysical limitations only, the potential terrestrial biomass supply is (1.26 × 10^{21}) J y^{-1} (57). The world energy supply in 2018 was (5.98 × 10^{20}) J y^{-1} (44), so if all energy supplied were provided by biomass, this would require 47.5% of the world's net primary production, which represents an unrealistic exploitation of natural resources (58) that would significantly impact global food production and biodiversity (54). Exploitation of more than 45% to 47% of net primary production is predicted to represent a planetary boundary, beyond which global net primary production will begin to fall (59)(60). Therefore, in practice, only a small proportion of energy requirements can be supplied by energy crops.

Competition with food crops is perhaps the key limitation to energy cropping. In 2019, around 820 million people world-wide, ~11% of the global population, were undernourished (61). Energy crops have a larger spatial footprint than most other forms of energy provision (62), and if productive agricultural land is used for energy cropping, land available to grow food will be reduced. However, many foods depreciate, and provision of food depends on supply chains and markets, so in areas where there is no market for food crops, growing energy crops can provide a useful diversification opportunity for farmers (62).

One way proposed to avoid competition between energy and food crops is to grow energy crops on marginal land that is unsuitable for food production (63)(64). Fast growing energy crops, such as the woody crops, *Salix* and *Populus*, and energy grasses, *Miscanthus* and *Arundo*, have high potential to provide phytoremediation of contaminated areas (65). Reforestation schemes could contribute ((8 × 10^{18}) to (1.1 × 10^{20})) J y^{-1}, equivalent to ~(1 to 18)% of the 2018 world energy supply (44). They could also provide additional benefits, such as regeneration of soils by increasing the organic matter content with associated C sequestration, improved soil water retention and protection of soils from erosion (54). However, use of marginal land to grow energy crops could also increase potential conflicts with loss of biodiversity because the traits that characterise an ideal energy crop (rapid growth, tolerance to drought and low soil fertility) also make it highly invasive (66).

In 2018, bioenergy provided (5.56×10^{19}) J y^{-1}, 9.3% of the annual global energy supply ((5.98×10^{20}) J y^{-1}) (44). Energy crops represented ~3% of the total biomass energy (54) (0.28% of the global supply), which is equivalent to a supply of ~(1.67×10^{18}) J y^{-1} (Figure 2). The area of land dedicated to producing (1.51×10^{18}) J biofuel feedstocks in 2007 was estimated to be (2.51×10^{5}) km^2, 1.6% of the harvested area (67). By 2017/18, the land area used for biofuel production had increased to (7.40×10^{5}) km^2 (Figure 2), ~4% of the total harvested area (68) and ~0.5% of the global land area.

Estimates of the environmentally sustainable technical potential for bioenergy provision (including energy crops, biomass fuels and organic wastes) assume that only land surplus to food and fibre requirements can be used and exclude land use change that results in deforestation or loss of wetlands or biodiversity (54)(69). Most estimates for 2050 agree a technical potential for bioenergy of at least (1×10^{20}) J y^{-1} (16% of the global energy supply) with large variations in estimates due to assumptions on the importance of different constraints (47). Deng et al. (70) estimated a technical potential for liquid biofuel by 2070 of ((0.40 – 1.90) $\times 10^{20}$) J y^{-1}, with 75% of that, ((0.32 – 1.43) $\times 10^{20}$) J y^{-1}, coming from energy crops, the remaining 25% being provided by agricultural and forestry residues (Figure 2). This required a total land area of ((3.7 to 13.2) $\times 10^{6}$) km^2 (70), which is equivalent to (2 to 9)% of global land area and (22 to 80)% of the arable area in 2017 (68) (Figure 2). From an economic perspective, the latest market trends project that global biofuel production will increase from the 2018 production values by 25% by 2024 to 11.6% of the global energy supply (71). Assuming the proportion of biofuels obtained from energy crops remains unchanged, this would represent an increase in cropped energy supply from ~(1.67×10^{18}) J y^{-1} in 2018 to ~(2.08×10^{18}) J y^{-1} in 2024 (0.35% of global energy supply), on a land area of (9.25×10^{5}) km^2 (~5.5% of the harvested area and ~0.5% of the global land area). Therefore, the projections suggest that the harvested area under energy cropping has potential for significant expansion without impacting food production or the environment (from ~4 to ~16%), and energy crops are already showing economic potential. However, if energy cropping is to expand to this extent, policies will need to be implemented to ensure protection of food production and biodiversity.

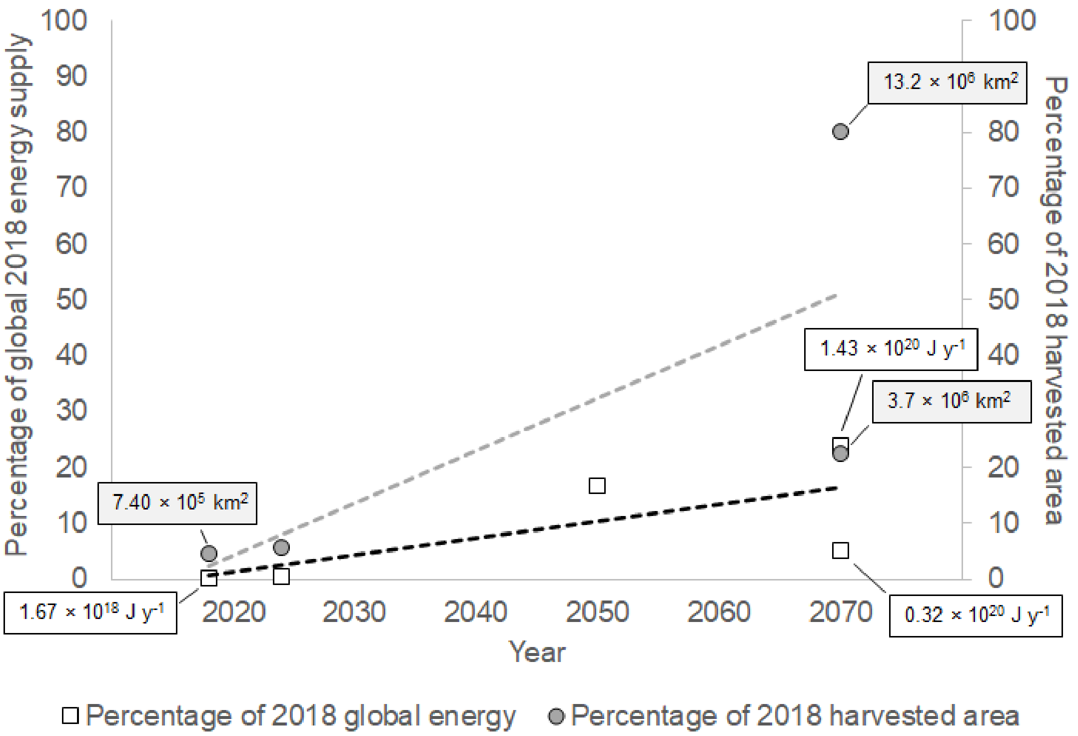

Figure 2 – Projected energy provision and land area required for energy crops. Black dotted line is the best fit to percentage of global energy supply produced by energy crops. Grey dotted line gives the best fit to percentage of harvested area occupied by energy crops.

Methods for use of crops for energy

Different methods can be used to provide energy from crops (Figure 3)(54). Direct combustion releases heat from oilseed and lignocellulosic biomass. Transesterification or hydrogenation produces biodiesel, syn-diesel or renewable diesel from oilseeds. Fermentation converts sugar or starch into ethanol, butanol and a range of other hydrocarbons. A biohydrogen fuel may also be produced by light or dark fermentation or in microbial fuel cells using the products of fermentation. Anaerobic digestion of sugar and starch produces biogas, which can either be burnt to provide heat and electricity, purified to produce biomethane which substitutes for natural gas applications, such as transport, or reacted by steam reforming to produce biohydrogen. Gasification of lignocellulosic biomass provides direct heat and produces a range of different liquid and gaseous fuels. Pyrolysis produces syngas, bio-oil and biochar from lignocellulosic biomass crops, which can be used to provide direct heat, diesel and other fuels and fuel additives. All of these processes leave residues, that could be incorporated into the soil to increase the organic matter and nutrient content, sequester C and improve productivity.

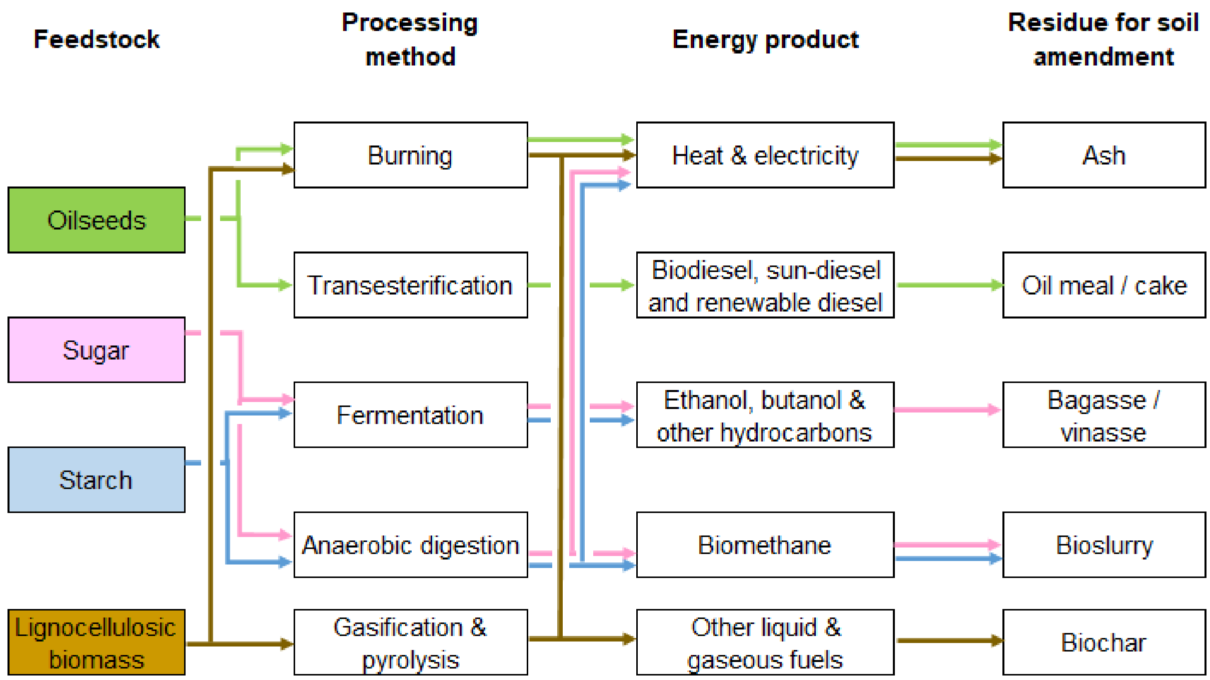

Figure 3 – Bioenergy routes and residues remaining from energy crops

Impacts on soils of producing crops for energy

While the productivity of soils impacts the potential provision of bioenergy, converting land to energy cropping in turn affects the C content and the productivity of the soils. The impact of energy cropping on soils depends on the category of land before conversion (forest, grassland, marginal or cropland), the energy crops grown (annual arable crops or perennial grasses and trees), how these integrate with or displace the existing land use, and use of the residues produced from the different methods of energy provision (47).

The impact of converting land to energy crops is highly site specific and depends on the plant inputs and management of the energy crop. Richards et al. (72) used the ECOSSE model to estimate greenhouse gas emissions and C sequestration resulting from land use transitions to energy crops in the UK; rotational crops - oilseed rape, wheat and sugar beet, and perennial crops – *Miscanthus*, short rotation coppiced willow and short rotation forestry poplar. They found reduced greenhouse gas emissions and increased C sequestration over a significant area of the UK when rotational arable cropping was converted to perennial *Miscanthus*, willow or poplar. Note that this study only accounts for direct impacts; potential indirect impacts due to land use change resulting from displacement of arable cropping were not considered. Growing perennial warm season grasses and short-rotation woody crops on marginal land has also been observed to reduce water and wind erosion and sequester (0.25 to 4) t ha^{-1} y^{-1} C (63). By contrast, conversion of permanent grass or forest to *Miscanthus*, poplar or rotational energy crops in the UK was simulated to result in increases in greenhouse gas emissions and losses of soil C mainly due to cultivation and reduced C inputs (72). Conversion of peatlands into land uses for energy cropping can also result in high and continuing losses of C (7)(73)(74); conversion of tropical virgin peat swamp forests in Southeast Asia to oil palm plantation has been observed to result in increased heterotrophic respiration of soil C between 7 t ha^{-1} y^{-1} and 95 t ha^{-1} y^{-1} (74) due to drainage and cultivation of the peats. While the impact of bioenergy on soils is highly dependent on the land selected to grow the energy crops, a market analysis of the economic and land-use consequences of biofuels using the GTAP-BIO model concluded that the major market-mediated responses are likely to include switching from food to energy crops, increases in intensification and conversion of forests or pastures to energy cropping (75). Therefore, without policy intervention to protect vulnerable soils, the overall impacts on soil C of land conversions for energy cropping are likely to be negative.

By contrast, incorporation into the soil of the residues from bioenergy provision can improve C sequestration and soil productivity. This might be used to increase yields of energy crops or to compensate for losses in food production areas by improving the productivity of the remaining areas cropped for food.

Ash residues from combustion of wood and biomass mixtures show significant variation in physical-chemical properties and elemental composition, depending on the type of biomass fuel burnt and the technology and temperature of combustion (76). However, ash is generally suitable for soil incorporation (76), increasing the pH and availability of phosphorus and micronutrients in the soil, although it has limited impact on the C content as, during efficient combustion, most of the C is oxidised and emitted as CO_2 (76).

Pressing and extraction of oils from oilseeds for transesterification produces oil cake (8% oil by weight) or oil meal (1 to 3 % oil by weight) (77). These residues are high in protein (15 to 50% by weight) so can either be fed to livestock (if edible), used for further energy generation (by combustion, pyrolysis or anaerobic digestion), or applied to soils as a nitrogenous fertilizer (71). If used as a fertiliser, they add C and N to the soil (as well as phosphorus and potassium), so improving soil properties and productivity (71). Globally, ~5% of oilseed production is used for biofuels (68), and production of biofuels from the three major oilseed crops (soybean, rapeseed and palm oil) produces 3.87×10^6 t y^{-1} oil cake or meal. Mazzoncini et al. studied a range of oilseed cakes and meals and measured the C content by dry matter weight in the range of (35 to 50)% and N content (5 to 6)% (78). Therefore, if the globally available oil cake/meal was used as a fertilizer it could add approximately ((1.4 to 1.9) × 10^6) t y^{-1} C and (1.9 to 2.3) × 10^5 t y^{-1} N. The amount of C that can be retained in the soil depends on the soil texture and initial C content (79), but for residues applied to C poor soils, on average ~10% of the applied C will be retained over the next 100 years (80). This assumption is also supported by long-term experimental data on C inputs to the soil and observed increases in soil C, such as given in the electronic Rothamsted Archive for both crop residues and animal manures (81). Assuming this is the proportion of C sequestered, applying the oil cake / meal residue from biofuel production from energy crops as an organic fertilizer could sequester ((1.4 to 1.9) × 10^5) t y^{-1} C. Note that if livestock numbers remain unchanged, diverting oilseed cake/meal that is currently fed to animals to use as a soil improver could have an indirect impact on soils due to more land being required to produce animal feeds.

The by-products available from bioethanol production depend on the feedstock and treatment method used (82). Ethanol production from starchy crops (cereals, millets, root and tuber crops) can produce vegetable oils, gluten (protein) meal and fibre (wet-milling) and "distillers grains with solubles" (dry milling), which are generally used as animal feeds (82). Key residues from production of ethanol from sugar are the fibres bagasse (sugar cane) and vinasse (sugar beet), which are widely used for production of biochemicals (furfural, xylitol, enzymes, vanillin and biopolymers), materials (paper, boards, textile fibre, construction materials and adsorbents) and animal feeds, but can also be used as soil conditioners and fertilisers (83). Each litre of ethanol produced by 7.9 kg sugar beet produces 600 g vinasse residue and 600 g dried beet pulp (82). In 2019, (1.32 × 10^{11}) dm^3 ethanol was produced (84) and approximately 60% of the ethanol was produced from sugar crops (85).

Therefore, assuming similar proportions for vinasse and bagasse residues, the amount of vinasse or bagasse available from bioethanol production would be approximately $((60/100) \times (1.32 \times 10^{11}) \times (600 / 10^6) = 4.75 \times 10^7)$ t y^{-1}. Assuming a C content of 40% (86), the amount of C that could be supplied to the soil globally in vinasse or bagasse would be approximately (1.9×10^7) t y^{-1}. Again, assuming an average of ~10% of applied C is retained over the next 100 years (80), this could sequester up to $(\underline{1.9 \times 10^6})$ t y^{-1} C. Note that this does not consider other essential uses of bagasse or vinasse, so the actual amount available for application to soils and C sequestered is likely to be less than this estimate.

Global biogas production from energy crops in 2017 was (1.33×10^{19}) J y^{-1} (87), with (3.35×10^{17}) J y^{-1} being produced from energy crops (88). Assuming a heating value of biogas containing (typically) 60% CH_4 of (2.41×10^7) J m^{-3}, this is equivalent to (1.39×10^7) m^3 y^{-1} biogas. Energy crops are often dry digested with any liquid recycled back to the digestion process and the solid part (bioslurry) used as a soil amendment (89). Biogas yield is positively correlated to the crude protein, crude lipid, cellulose and hemicellulose content (90)(91)(92), and negatively correlated to the lignin (acid detergent lignin, (92)) and water-soluble carbohydrate content of the feedstock (91)(93). The biogas yield from energy crops, therefore, has a large range; depending on composition of the feedstock, it can range from (39 to 70) m^3 t^{-1} (beet leaves and cut grass) to (550 to 650) m^3 t^{-1} (rapeseed) (89). Therefore, the global production of biogas $((3.35 \times 10^{17})$ J $y^{-1})$ requires $((2.1 \times 10^7)$ to $(3.6 \times 10^8))$ t energy crops, containing $((9.6 \times 10^6)$ to $(1.6 \times 10^8))$ t C (assuming an average C content of the feedstock of 45% (94)). The efficient conversion of organic C to CH_4 results in a reduction in the C content to only (6 to 29)% of the feedstock (95)(96)(97). Therefore, the C retained in bioslurry from global biogas production from energy crops is likely to be $((5.8 \times 10^5)$ to $(4.7 \times 10^7))$ t y^{-1}. The C that remains is usually highly stabilized, so C retention when applied to C poor soils is likely to be at least the 10% assumed for other residues (80), equivalent to $(\underline{(5.8 \times 10^4) \text{ to } (4.7 \times 10^6)})$ t y^{-1}. The retention and availability of nutrients (N and P) in the bioslurry is high, so bioslurry also acts as an excellent organic fertiliser (98)(99)(100), potentially replacing the production of fertilizers using fossil fuels.

The global use of dedicated biomass crops for pyrolysis and gasification is currently relatively small, but Woolf et al. estimated a technical potential for pyrolysis of (1.4×10^{16}) J y^{-1} biomass from agroforestry crops (0.002% of global energy supply) (101). Pyrolysis and gasification produce a biochar residue that can either be used for further energy provision or incorporated into the soil (102). Biochar is a highly recalcitrant form of C and so has high potential to permanently sequester C (101). The proportion of biochar produced and its stability depend on the conditions of the process, especially the temperature and rate of heating (103)(104)(105)(106)(107). Pyrolysis occurs between 350 and 900 °C in the absence of oxygen, but is usually performed in the temperature range 475 – 575 °C (108). The proportion of C retained in the biochar after pyrolysis can range from 20% at high temperatures (575 °C) to 50% at low temperatures (475 °C) (109). At 475 °C and low rates of heating (slow pyrolysis), Yang et al. observed that most of the carbohydrates were volatilised, leaving behind only recalcitrant compounds (110), whereas at 475 °C and high rates of heating (fast pyrolysis), limited heat transfer resulted in a fraction of the biomass (3 to 12%) remaining as cellulosic and hemi-cellulosic materials which are more rapidly degraded in the soil (109)(111). Increasing the temperature reduces the proportion of degradable compounds to zero, but also reduces the amount of C retained in the biochar (109). Gasification occurs at higher temperatures ((800 to 1200) °C (112)) and retains a much lower proportion of the biomass C in the biochar, only (3 to 7)% (102). Therefore, while the amount and degradability of C in the biochar varies widely (depending on feedstock and production temperature), we can estimate that the C retained ranges from (3 to 50)% with a proportion of recalcitrant C from (((100 – 12) = 88) to 100)% (111), meaning that (2 to 50)% of the feedstock will be processed into recalcitrant C and sequestered when applied to the soil. Therefore, although currently of limited extent, this is a technology that has high potential for future C sequestration (101).

The availability and concentration of nutrients in biochar is dependent on the temperature, rate of heating and the nutrient content of the feedstock, with higher N and P concentrations in biochars produced at lower temperatures (113), higher availability of the nutrients in biochars produced by slow processes (114), and the nutrient concentration being linearly dependent on the nutrient content of the feedstock (115). However, compared to other processes, losses of nutrients during pyrolysis are relatively high (98), so new methodologies are also needed to avoid losses of nutrients from the feedstock during the pyrolysis process. The temperature and rate of heating also impact the porosity of the biochar, with high temperature fast pyrolysis producing more porous biochars (116)(117). This high porosity and the presence of both positively and negatively charged exchange sites makes biochar effective at reducing losses of both cationic and anionic nutrients, especially from highly weathered soils that are deficient in exchange sites (118), so further impacting the availability of nutrients for growing crops. Therefore, this technology also has high potential to improve the future productivity of such soils (e.g. tropical soils (119)), although yield penalties have been observed at higher rates of application (over 50 t ha^{-1} biochar) in temperate or alkaline soils (118).

Bringing all this together, if national policies are designed to avoid C losses due to land use change on vulnerable soils (at worst resulting on no net change in soil C), then the potential global impacts of energy crops on soil C sequestration can be assumed to be equivalent to the impacts of using the residues as a soil amendment, currently ((2.1×10^6) to (6.8×10^6)) t y^{-1}, (0.2 to 0.6)% of C emissions from LUCF (47). The global impacts of amending soils with the residues currently available from energy crops are summarised in Figure 4.

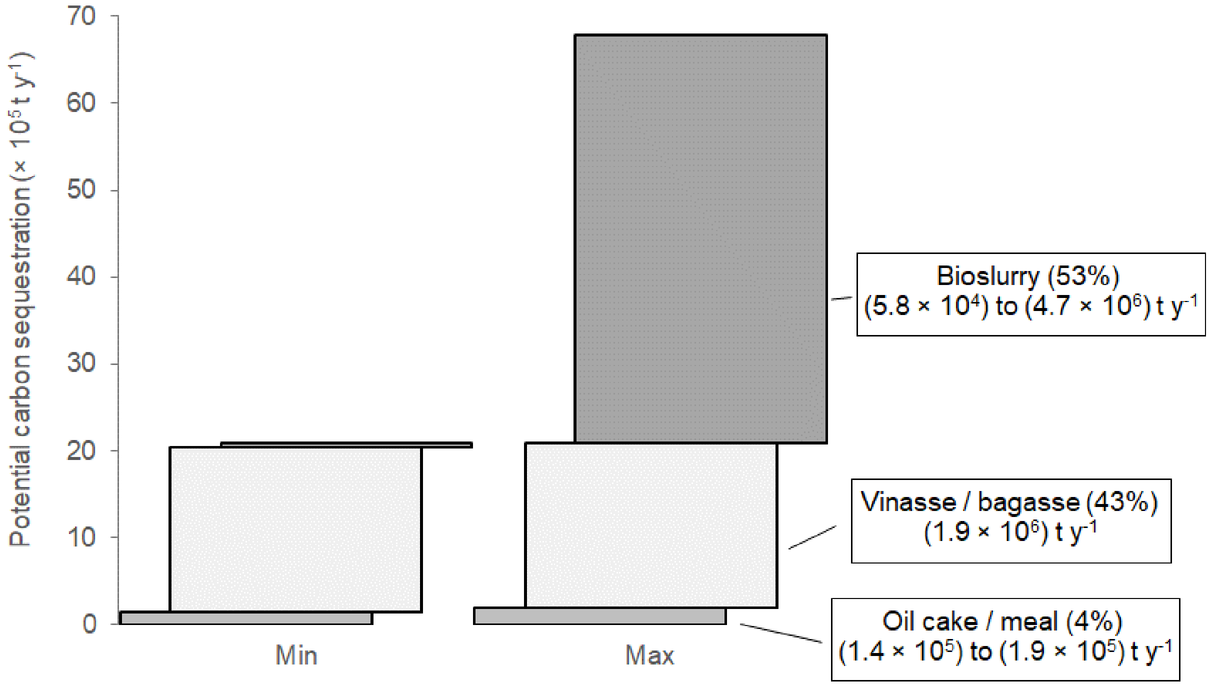

Figure 4 – Global impacts on C sequestration of incorporating residues from energy crops

Use of organic wastes for energy

The main sources of organic wastes available for energy provision are agricultural and forest residues, and municipal and industry wastes. Agricultural residues include animals manures (120) and crop harvest residues, such as straw, haulms and seed husks (121). Forest residues include dead wood and the remnants from wood processing (sawdust, bark and black liquor) (47)(122). Municipal and industry wastes include wastes from the food industry (123)(124) including animal rendering (125), municipal solid wastes (126) and sewage sludge (127). On average, the per capita rate of waste generation is 1.22 t y^{-1} for agricultural residues and 0.27 t y^{-1} for municipal solid wastes (128), which over a world population in 2018 of (7.59×10^9) capita (129), amounts to approximately (9.54×10^9) t y^{-1} agricultural wastes and (2.05×10^9) t y^{-1} municipal solid wastes (compares to (2.01×10^9) t y^{-1} quoted for municipal solid waste by Kaza et al. (130)). This gives a total of (1.16×10^{10}) t y^{-1}, with forestry and industry wastes further adding to this total. Crop residues and manures are the main sources of untreated or composted wastes applied to soils (131). Other wastes require pre-treatment, by an increasing range of methods, including composting, anaerobic digestion, pyrolysis and gasification (132). This is required to avoid immobilisation of nutrients (106) and to reduce pathogen levels before application to the soil (132)(133). Therefore, use for energy provision of organic wastes that are not widely applied to crops (i.e. forest, municipal and industry wastes) could actually facilitate and incentivise application of organic wastes to cropland and so increase the potential C sequestration and nutrient availability in soils. By contrast, using crop residues and livestock manures for energy provision is more likely to directly compete with their use as organic fertilisers, so could impact future soil productivity unless the residues from energy conversion processes are returned to the soil to compensate for this. Note that removal of heavy metals from some wastes will require additional extraction treatments, such as complexation with EDTA, uptake by heavy metal tolerant plants or bioelectrochemical extraction (134)(135).

Crop harvest residues

Lal estimated that in the early 2000s crop harvest residue production was (3.76×10^9) t y^{-1}; 74% cereals, 8% legumes, 3% oil crops, 10% sugar crops and 5% tubers (136). Similarly, Smil (137) estimated that in the mid-90s global crop residue dry matter production was (3.74×10^9) t y^{-1}. The average ratio of crop residues to production for these two estimates is 0.61 (138). Assuming this ratio has remained unchanged since 2000 (evidence for this assumption provided by e.g. (139)(140)(141)), the total amount of crop residues produced can be estimated from the FAO crop production data (138). This extrapolates crop harvest residue production for 2018 to (5.83×10^9) t y^{-1} (Figure 5). Lal (136) estimated the potential bioenergy provision from (3.76×10^9) t y^{-1} crop residues to be ~(6.99×10^{19}) J y^{-1} (using an approximate fuel value of crop residues of (1.86×10^{10}) J t^{-1} (142)). Assuming the same fuel value for 2018 crop residues, the total bioenergy available from crop residues would be (1.07×10^{20}) J y^{-1}, nearly 18% of the 2018 global energy supply ((5.98×10^{20}) J y^{-1}) (44). Note, this value is higher than the bioenergy estimated to be potentially available from crop harvest residues by Smeets et al., in 2050 ((5.04 to 7.02) $\times 10^{19}$) J y^{-1} (143), as alternative uses of the residues have not yet been subtracted. Conversion of crop residues into bioethanol is discussed further in SM.2.

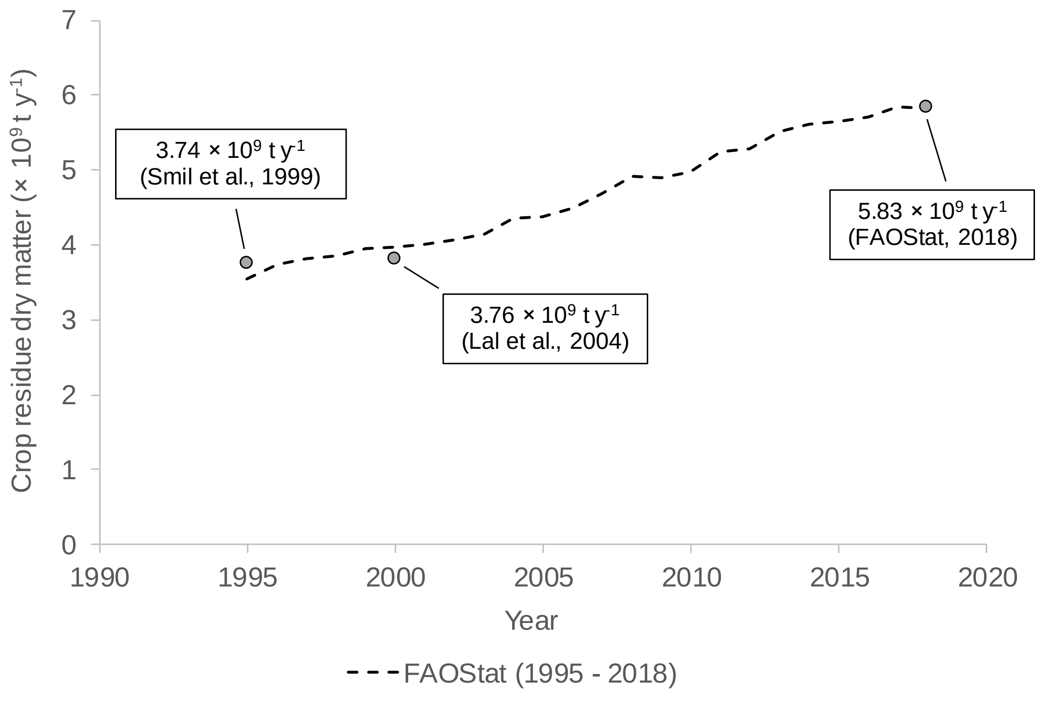

Figure 5 – Total crop harvest residue production estimated from FAOStat (138) using residue : production ratios provided by Lal et al. (136) and Smil (137)

Evidence from a range of authors suggests that (30 to 60)% of crop residue dry matter can be removed from land without impacting sustainable crop production (144)(145). This is required to reduce erosion, but does not ensure maintained soil organic matter and nutrient content, which is becoming critical to continued crop production in many places in the world (146)(147). Other analyses suggest that if 20% of the soil surface is covered by crop residues, soil erosion will be reduced by 50% (148), and if 90% is covered this increases to a reduction in water erosion of 93% compared to the uncovered soil (149).

Other competing uses for crop residues include use as animal feed or bedding (approximately (25 to 40)% (143)), burning for fuel (~(7% to 16)% (137)(150)), and other minor uses, such as mushroom composts, pulp-making for paper and bio-chemicals (137). A significant proportion is burnt in the field in order to quickly prepare for subsequent crops; between 1995 and 2017, this ranged from 5.4% of total crop residues in 1995, declining by 0.0005% each year ($R^2 = 0.94$) to 4.1% in 2017 (151). Therefore, in 2018, ~4% of crop residues ((2.36 × 10^8) t y^{-1}) were burnt in the fields that, if the necessary supply chains had been established, could have been used to provide energy without impacting soil amendments. Assuming the fuel value of (1.86 × 10^{10}) J t^{-1} (142), this could provide an extra (4.39 × 10^{18}) J y^{-1} energy (~0.7% of global energy supply (44), (Figure 7)). Alternatively, these unused crop residues could be incorporated into the soil to increase its organic matter content and improve recycling of nutrients. Assuming at least 40% of the crop residues must be incorporated in the soil either mechanically or by biological processes to maintain sustainable production (144)(145), this leaves up to 24% of the crop residues unaccounted for (Figure 6), equivalent to (2.60 × 10^{19}) J y^{-1} energy (~0.7% of global energy supply (44), Figure 7). Note that if these unaccounted-for residues are currently incorporated in the soil, using them for energy provision could reduce the organic matter content of the soil and impact future soil productivity.

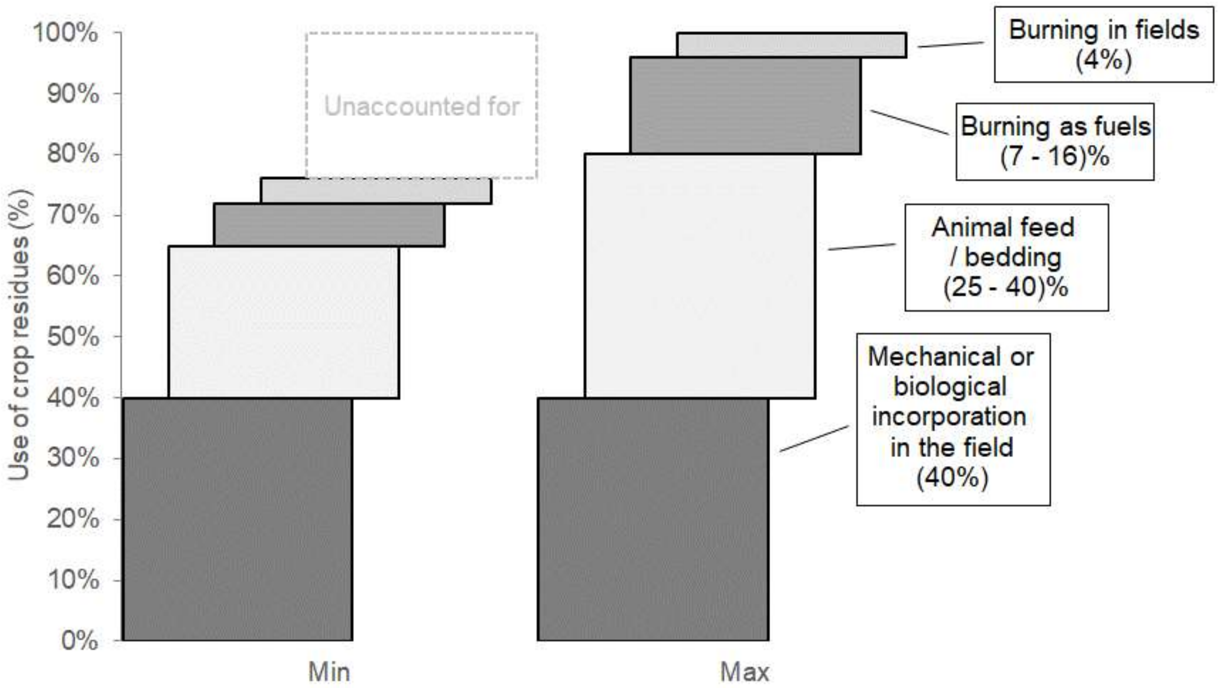

Figure 6 – Global uses for crop residues

Assuming the C content of crop residues is (40 to 50)% (152) and that an average of ~10% of the applied C is retained over the next 100 years (80), incorporating the ~4% of crop residues that are currently burnt in the fields would sequester an extra ((9.4 × 10^6) to (1.2 × 10^7)) t y^{-1}; ~(0.9 to 1.1)% of C emissions from LUCF (47). Unaccounted for residues could sequester up to an additional ~3.2%; (3.5 × 10^7) t y^{-1}. Total from burnt and unaccounted for residues = (9.4 × 10^6) to (4.6 × 10^7) t y^{-1} (~(4.0 to 4.3)% of C emissions from LUCF) (Figure 8).

Data from FAO on global fertilizer applications (153) and N in crop residues (154) show that between 1995 and 2017, N contained in crop residues of 11 major crops (barley, beans, maize, millet, oats, potatoes, paddy rice, rye, sorghum, soybeans, wheat) was 33% (standard error ± 0.2%) of fertiliser N applied globally. Therefore, given (1.09 × 10^8) t y^{-1} N applied in fertiliser in 2018 (153), the N content of the crop residues is likely to be ~(3.55 × 10^7) t y^{-1}. Similarly, Smil estimated that in the 90s crop residues globally contained ~33% of the N taken up by the crop (with ~30% of P and ~65% of K) (139). While this suggests fertilizer inputs could be significantly reduced by recycling crop residues, organic inputs with a high C:N ratio (over ~10) tend to temporarily immobilize N in the soil and so decrease availability of N to the subsequent crop (106). The average C:N ratio of crop residues between 1995 and 2018 for these 11 major crops was between 23 to 29. While the composition of crop residues will vary between crop types, with some crop residues (e.g. legumes) having a lower C:N ratio than others (e.g.

cereals), this suggests that on average global availability of N to the subsequent crop will be reduced by incorporation of fresh residues, suggesting the need for pre-treatment.

Composting and mixing with wastes with lower C:N ratios are methods frequently used to make crop residues more suitable for soil incorporation (98). Composting typically retains (26 to 48)% of the C in the starting material (155), so assuming the same ~10% is retained over the next 100 years (80), this equates to global C sequestration of (2.45×10^6) to (5.32×10^7) t y^{-1} from composting of the crop residues currently burnt in the fields or unaccounted for. Using crop residues for bioenergy by methods that retain the nutrients in the waste, such as anaerobic digestion (98), could help to increase availability of nutrients while also retaining some of the C in the soil and providing energy. Anaerobic digestion typically retains (6 to 29)% of the C in the feedstock (95)(96)(97), so would sequester a little less than composting, ((5.66×10^5) to (2.37×10^7)) t y^{-1} (up to ~2.2% of C emissions from LUCF (47)). This would provide ((2.22×10^{17}) to (1.54×10^{18})) J y^{-1} (up to 0.3% of the 2018 global energy supply (44)).

Livestock manures

Current uses of livestock manures include use as organic fertilisers, as fuels and in small-scale construction activities (156). Livestock manures are a major source of crop nutrients in both high- and low-income countries, contributing ~(37 to 61)% of the total global N input to the land surface (157). Zhang et al. (158) used data on the spatial distribution of livestock from the Global Livestock Impact Mapping System (159) together with country specific annual livestock populations to provide a disaggregated dataset of global manure production from 1860 to 2014. Manure N production increased at a rate of (7×10^5) t y^{-1} ($p < 0.01$) from (2.14×10^7) t y^{-1} in 1860 to (1.31×10^8) t y^{-1} in 2014 (158); by extrapolation, manure N in 2018 would be (1.34×10^8) t y^{-1}. In 2014, only 19% of manure was applied to cropland, (2.45×10^7) t y^{-1} N (158); this would be equivalent to an application of (2.50×10^7) t y^{-1} N in 2018. Estimates provided by FAO on the amount of manure N applied to all soils (including both croplands and grasslands) in 2018 were just 9% higher than this, at (2.73×10^7) t y^{-1} (160). However, Gerber et al. estimated a much lower rate of manure N application to crops; (7.8×10^6) t y^{-1} N in 2000, which is just 6% of the manure N produced (161). They attributed their lower estimate to using more refined animal and region-specific management factors (161). In 2018, this would translate to a lower value of manure N application to crops of (8.61×10^6) t y^{-1} N.

In a meta-analysis of 521 observations, Liu et al. (162) characterised the C:N ratio of manures as ~18 (±2) for cattle, ~12 (±1) for pigs and ~8 (±2) for poultry. Assuming the same manure C:N ratio as cattle for asses, buffaloes, camels, goats, horses, llamas, mules and sheep (18 ± 2), the total C applied to soils as manure in 2018 would be between ((1.2 ± 0.5) $\times 10^8$) t y^{-1} (cropland only) (161) and ((3.9 ± 0.5) $\times 10^8$) t y^{-1} (all soils) (160). If ~10% of this C is assumed to be sequestered over the next 100 years (80), this represents C sequestration of ((1.2 to 3.9) $\times 10^7$) t y^{-1} (~(1.1 to 3.6)% of C emissions from LUCF (47)) (Figure 8). However, if a higher proportion of the manure produced could be captured, additional C sequestration in cropped soils from application of manures could be up to ((1.68 to 1.77) $\times 10^8$) t y^{-1} C; note this would reduce C sequestration from manure deposited on pastures. Application of manures to soils increases the C content of the rapidly turning over organic matter pools by ~88% compared to only ~27% in the recalcitrant pools (162). Therefore, in addition to sequestering soil C, the organic matter continues to decompose and release nutrients to crops. Manure application has also been demonstrated to increase aggregate stability and soil porosity (163)(164), and decrease bulk density (165), so further improving the conditions for root growth and crop production.

In 2018, biogas and biomethane production from livestock manures provided (4.6×10^{17}) J y^{-1} energy worldwide, but the *technical potential* for biogas production from manures considering only feedstocks that do not compete with applications to agricultural land is estimated to be over 16 times the current use, (7.5×10^{18}) J y^{-1} (1.25% of global energy supply (44), Figure 7), and is expected to increase by a further 40% by 2040 (166). Livestock manures were the major feedstock for biogas production in 2018, providing 34% of the total production (166). However, if both anaerobic digestion and gasification processes are considered, the yield of biogas and/or biomethane from manures is much lower than for many other feedstocks, partly due to the high moisture content of manures. The average biogas production yield is only (3.35×10^8) J t^{-1} for sheep and cattle manure, and (1.63×10^9) J t^{-1} for poultry and pig manure, compared to (6.70×10^9) to (1.06×10^{10})J t^{-1} for bioenergy crops, (7.45×10^9) J t^{-1} for wood residues, (9.30×10^9) J t^{-1} for food and green waste, and (1.51×10^{10}) J t^{-1} for industry wastes (166). The global potential for biogas and biomethane production in 2018 was (2.34×10^{18}) J y^{-1} for municipal solid wastes and (6.82×10^{18}) J y^{-1} for woody biomass, totalling 32% of the potential production from wastes (166). Therefore, there is high potential for food, green and industry wastes to make up a larger share of biogas production in the future, leaving a larger proportion of livestock manures for incorporation in the soil. If manures currently incorporated in soils were instead diverted to energy provision, this would result in a loss of soil C of up to (3.9×10^7) t y^{-1} (160), ~3.6% of C emissions from all LUCF (47)(Figure 8).

In low-income countries, manure is often dried to produce dung cakes that are burnt to provide household energy (167)(168). For example, in the Northern Highlands of Ethiopia, as much as 80% of household energy consumption is provided by crop residues and dung (169). Negash et al. (169) discussed the potential positive impacts of introducing household scale anaerobic digesters on the C and nutrient stocks of soils. This is in part due to anaerobic digestion preventing dung from being used or sold as a fuel; burning of dung leaves very little C or nutrients for soil incorporation, whereas anaerobic digestion will retain 6 to 29% of the C in the digestate and most of the nutrients (95)(96)(97). In addition to this, the organic matter that is incorporated may be more recalcitrant than in untreated manure. Smith et al. (146) used simulation modelling to consider the impact of different treatments on C sequestration in soils and crop production, and surmised that anaerobic digestion of manures before incorporation actually increases C sequestration compared to untreated manures due to the stabilisation of organic matter by the digestion process (98)(111).

Inputs to soils due to energy provision with organic wastes

The production as biogas and biomethane from all feedstocks worldwide in 2018 provided (1.47×10^{18}) J y^{-1} energy (166). The feedstock was composed of 25% crop residues, 34% livestock manure, 25% municipal solid waste, 3% forestry and 13% unspecified residues (166). However, the *technical potential* for biogas production from only feedstocks that do not compete with food or organic waste applications to agricultural land was estimated by IEA to be 16 times this value, (2.39×10^{19}) J y^{-1} (166). If gasification is included to produce biomethane, which allows forestry residues to be included, this increases to a total technical potential for biogas and biomethane of (3.06×10^{19}) J y^{-1}, ~5.1% of the 2018 global energy supply (44)(166) (Figure 7).

Assuming the biogas production yield for municipal solid waste given by IEA (166), (2.22×10^7) t y^{-1} of municipal solid waste would have been used for biogas and biomethane production in 2018, which after treatment by anaerobic digestion to reduce pathogens (133) and further processing to remove heavy metals (134)(135) could be suitable for application to soils. Assuming an average C content in the feedstock of 45% (94), with (6 to 29)% of feedstock C retained in the digestate (95)(96)(97) and ~10% of this waste sequestered over the next 100 years (80), this represents additional C sequestration of only ((5.98×10^4) to (2.89×10^5)) t y^{-1} globally, but with a *technical potential* of (2.34×10^{18}) J y^{-1} (~0.4% of the 2018 global energy supply (44)(166)). If applied to soils, this would sequester ((4.19×10^5) to (2.02×10^6)) t y^{-1} C globally (up to 0.2% of C emissions from LUCF (47)).

Food waste is a particularly good feedstock for anaerobic digestion, leading to consistent biogas production that is higher than achieved with energy crops (480 ± 88) dm^3 CH_4 per kg of volatile solids (170)). Globally, approximately (1.3×10^9) t y^{-1} food waste is disposed of with no further use (171). Abundant quantities of food supply chain wastes are produced every year and have significant potential for valorisation through production of fuels and other chemicals (123). In 2010 in the UK, 10% of food wastes from the Federation of Food and Drink producers were used as animal feed, 75% for soil incorporation, 5% for energy provision (1% by anaerobic digestion and 4% by incineration) and 9% went to landfill (123). The global production of used cooking oil was ~(5×10^6) t y^{-1} (123). Used cooking oils are burnt in fuel boilers, used as for lubricants / surfactant precursors and for biodiesel production (123). Smeets et al. estimated that the bioenergy potentially available from food wastes will increase by 2050 to (1.6×10^{19}) J y^{-1} (143).

Assuming the production yield for biomethane from woody residues given by IEA (166), (5.62×10^6) t y^{-1} of woody biomass would have been required for biomethane production in 2018. Assuming a C content of woody biomass of 45% (94) and (3 to 7)% of the C is retained after gasification (102), the C content of the biochar residue would be ((7.58×10^4) to (1.77×10^5)) t y^{-1}. This is a low value, but biochar is highly recalcitrant, so a large proportion of this would be permanently sequestered into the soil (111).

Other methods used to release energy from municipal solid and industry wastes include fermentation to produce bioethanol (172) and biohydrogen (173), gasification, pyrolysis, torrefaction and hydrothermal carbonization (174). In countries with poor waste collection facilities, it has been suggested that municipal biowaste could be used to produce charcoal for use as a household fuel (SM.3) or biochar for soil improvement, providing cost recovery for waste collection as well as contributing to sustainable farming and energy provision (175).

Impacts on soils of using organic wastes for energy

In summary, energy provision from organic wastes in 2018 could have provided up to (6×10^{19}) J y^{-1} from municipal wastes, livestock manures, woody biomass and from crop residues currently burnt as fuels, disposed of by burning in the fields and unaccounted for (Figure 7). This is nearly 10% of the 2018 global energy supply (44). Incorporating bioslurry and biochar from biogas and biomethane production would increase soil C by nearly (2×10^7) t y^{-1}, but reduce inputs from livestock manures and crop residues (assuming unaccounted for crop residues are currently incorporated in the soil) with associated loss of soil C of over (7×10^7) t y^{-1}, resulting in net losses of over (5×10^7) t y^{-1} soil C (Figure 8) (~4.6% of C emissions from LUCF (47)). If organic wastes are to be used for energy provision while also protecting soils, use of livestock manures for energy provision should be avoided. However, anaerobic digestion of crop residues could be beneficial as it reduces the C:N ratio, so allowing crop residue N to be released. This would provide (4×10^{19}) J y^{-1} (6.7% of the 2018 global energy supply (44) (Figure 7)), while also reducing loss of soil C (Figure 8). Note that crop residues with very high C:N ratios would need to be mixed with more N rich waste sources (such as food wastes) to optimise the C:N ratio for the digestion process.

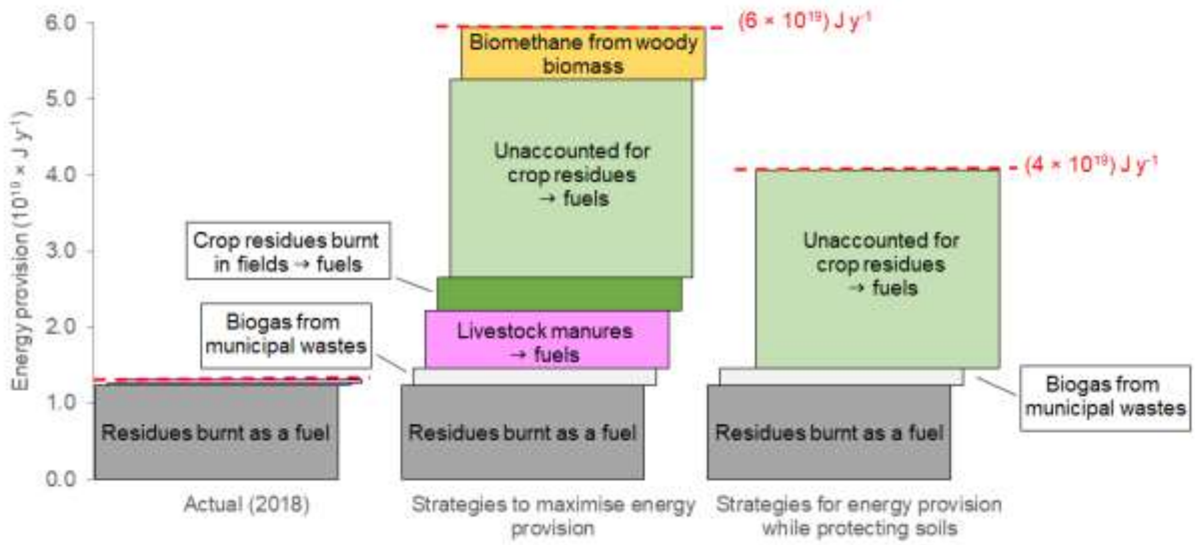

Figure 7 – Maximum potential impact of different strategies for using organic waste on energy provision in 2018. Note: red font indicates total energy provision.

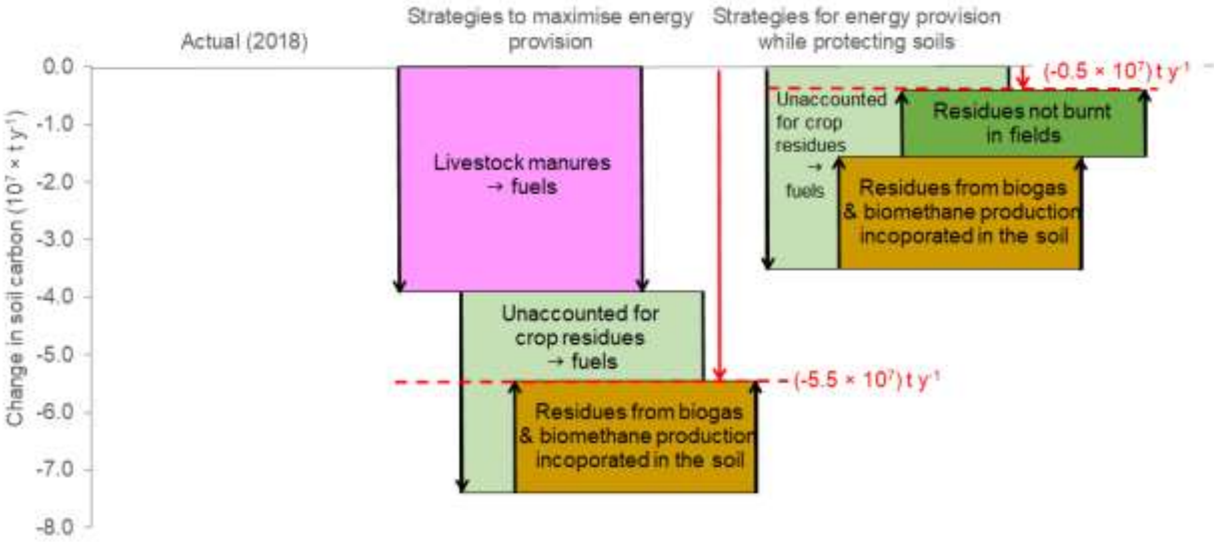

Figure 8 – Maximum potential impact of different strategies for using organic wastes in 2018 on soil carbon sequestration. Note – Black arrows indicate the magnitude and direction change in soil carbon associated with the different strategies; red arrow indicates net change in soil carbon; red font indicates net the magnitude of change in soil carbon.

Onshore wind, hydropower, solar and geothermal schemes

Onshore wind, hydropower, solar and geothermal schemes have two major impacts on soils; 1. they remove land area that could otherwise be used for other purposes, and 2. they disturb the vegetation and hydrological regime of the soil, so impacting C emissions in the area around the power scheme infrastructure.

The land area required for such energy schemes depends on the land use and the size of power generation but, globally, is relatively small. In addition to the area required for foundations, the space occupied by any roads or other infrastructure required for operation of the different energy schemes should be accounted for. Roads are typically (4 to 10) m wide (up to 10 m during construction with (4 to 5) m permanent road), so will require (4 to 10) m^2 km^{-1} road (177). Although an onshore windfarm may occupy a large area of land, other land uses can continue to be implemented around turbines, resulting in only a small actual loss of productive area. Typically, the land footprint for a wind turbine on agricultural land can be up to 1 m^2 MWh^{-1} ((2.78×10^{-10}) m^2 J^{-1}) (176), although much lower values are possible in high capacity sites, for example in exposed sites in Scotland (177). Forestry installations require a larger area to reduce the effects of turbulence from the trees on the turbine performance; an additional area is felled and kept open during the wind farm lifetime, typically ~80 m from the turbine blade tip to forestry edge (178). This requires an additional (2×10^4) m^2 of forested area to be felled for each turbine (178). Assuming the typical size of a turbine is 3.1 MW (179) and the global average capacity factor of 34% (180), this amounts to just over 2 m^2 MWh^{-1} ((6.05×10^{-10}) m^2 J^{-1}). For hydropower, the land footprint is 5 to 10 m^2 MWh^{-1} ((1.39×10^{-9}) to ((2.78×10^{-9}) m^2 J^{-1}) (176). The global average land footprint for solar photovoltaic (PV) power is currently very low ((0.7 – 1.8) m^2 MWh^{-1} ((1.94 – 5.00) $\times 10^{-10}$) m^2 J^{-1}), due to solar energy being produced on rooftops and land that is unsuitable for cultivation or forest cover (181)(182). However, as market penetration is projected to increase, by 2050 the land footprint is likely to increase to (6 – 30) m^2 MWh^{-1} ((1.67×10^{-9}) to ((8.33×10^{-9}) m^2 J^{-1}) depending on irradiance and latitude (183). For geothermal schemes, the land footprint depends on the geothermal source, type of energy conversion used, power capacity, cooling system and location of wells, pipelines, substations and auxiliary buildings (184), but is estimated to be only ((0.03 - 0.40) m^2 MWh^{-1} ((((0.92×10^{-12}) to (1.29×10^{-10})) m^2 J^{-1}) (181)(182), so can be considered to be negligible at the present time.

Globally, the 2018 installed capacity for energy generation by on-shore wind was (5.42×10^5) MW (180), generating (1.20×10^9) MWh ((4.32×10^{18}) J y^{-1}) (185). In 2018 in Austria and Denmark, 86% of onshore windfarms were located on agricultural land, with only 7% on forested land (186). If a similar proportion is assumed worldwide, this would amount to a loss of only (1.0×10^3) km^2 of agricultural land and (1.8×10^2) km^2 forested land, giving a total global loss of land area in 2018 to onshore wind of only (1.2×10^3) km^2, which is equivalent to just 0.0008% of the global land area (68). Installed global hydropower capacity in 2018 generated (4.2×10^9) MWh y^{-1} ((1.51×10^{19})) J y^{-1} (187), which would equate to a slightly larger land area of ((2.1×10^4) to (4.2×10^4)) km^2 worldwide, equivalent to (0.01 to 0.03)% of the global land area (68). Global generation of power by solar PV in 2018 was (5.85×10^8) MWh y^{-1} ((2.11×10^{18}) J y^{-1}) in 2018 (188), which equates to an area of only ((4.1×10^2) – (1.1×10^3)) km^2 ((0.0003 – 0.0007)% of global land area), but this is projected to increase to (0.5 – 5)% of the total land area by 2050 (183). Depending on the soil, location, previous land use and management of vegetation under solar panels, this could result in soil C losses of up to (3.79×10^{-9}) g J^{-1} (183). Geothermal power generated in 2018 was only (9.0×10^7) MWh y^{-1} ((3.24×10^{17}) J y^{-1}), equating to an area of land less than ((4.2×10^1) km^2, which is only 0.00003% of global land area).

While onshore wind, hydroelectric, solar and geothermal schemes in 2018 together provided (2.19×10^{19}) J y^{-1}, ~3.66% of the 2018 global energy supply (44), they occupied an area of land which is equivalent to less than (4.4×10^4) km^2, 0.03% of the total land area and 0.3% of the global harvested area (68). However, for onshore wind, the impacts on hydrological regime have more potential to be globally significant than the area of land directly occupied. To avoid loss of productivity, non-productive land is often used to site energy schemes, and for windfarms in the UK, Ireland and Spain, this is often on deep peats which are generally in windy areas with high capacity for energy generation, but also hold large amounts of C that are vulnerable to loss with land use change. Large windfarm developments in the Xistral Mountains in Galicia are examples of windfarms on areas dominated by blanket bog in Spain (190). In Scotland in 2014, 62% of windfarms were located on peats (191). These developments result in drainage of the peats with associated gaseous, dissolved and erosion losses of C (177)(192)(193). The amount of C lost is highly dependent on the condition of the peat (%C, bulk density and water table depth), the extent of drainage around any infrastructure (e.g. roads, cable trenches), and the extent of infrastructure required (177). In order to minimise C losses from peatlands, infrastructure should be located and designed to minimise drainage of highly organic soils, for instance by constructing and maintaining floating roads to ensure they do not sink and avoiding areas of deep peat (177). However, following decarbonization of the electricity grid, the time required for a windfarm sited on a peatland to pay back these losses (through reduced use of fossil fuels) will usually become longer than the lifetime of the windfarm even with careful planning of the location of infrastructure (194). Therefore, to reduce damage to these valuable areas, which provide important habitats and large stores of soil C, construction of windfarms on undegraded peats should be avoided (195).

Conclusions

Renewable energy provision is an important component of our global drive to reduce greenhouse gas emissions and to limit climate change. However, to avoid damaging our important soil resources, implementation of renewable energy schemes needs to be done with care.

Peats are sometimes referred to as a renewable energy source, but the combustion of peat releases, in a short period of time, C that accumulated over thousands of years, and any C sequestration possible due to subsequent plant inputs provides negligible compensation over the short term. Therefore, peat is not a renewable resource and peat extraction should be phased out. Burning of peats provides only ((1.41×10^{17}) J y^{-1} energy, ~0.02% of the 2018 global energy supply, yet burning alone emits (7.80 to 8.67) $\times 10^6$) t y^{-1} C, which is ~(0.7 to 0.8)% of the C losses from LUCF. In addition to the direct losses by burning, peat extraction is likely to cause peats to drain which is a significant source of C emissions globally; despite covering only 3 – 4% of the global land area, in 2009 net greenhouse gas emissions from all drained peats (not only from peat extractions) were equivalent to ~(50 to 75)% of C emissions from all LUCF.

Bioenergy crops in 2018 provided (1.67×10^{18}) J y^{-1} energy, ~0.3% of the global energy supply. However, if growing bioenergy crops requires permanent land use to be disturbed, it can result in large losses of soil C and a downward trend in soil productivity. With implementation of policies to avoid these damaging changes in land use, the bioenergy crops grown in 2018 could have sequestered an additional ((2.10 to 6.79) × 10^6) t y^{-1} C (~0.4% of the C losses from LUCF) through incorporation of the residues from energy conversion processes into the soil. Therefore, bioenergy provision can have a positive impact on both soils and energy supply, but policies are needed to ensure soils are protected and food production is maintained.

Organic wastes available in 2018 could have provided up to (5.94 × 10^{19}) J y^{-1} energy (~10% of the global energy supply). This is a huge potential, but it comes at a C cost of (5.5 × 10^7) t y^{-1} lost from the soil (~5% of the C losses from LUCF). This is due to energy provision reducing the C available for soil amendment. However, the concentration of nutrients in some crop residues is too low for direct incorporation into the soil; in this case some form of pre-treatment is needed. Methods that provide energy while also retaining nutrients in the residues, such as anaerobic digestion, have high potential to benefit both soils and energy provision. Livestock manures have a low biogas yield compared to other organic wastes, so residues such as food, green and industry wastes would be better feedstocks. Avoiding using manures as a source of energy would reduce energy provision to (5.19 × 10^{19}) J y^{-1} (~9% of global energy supply) but would also avoid soil degradation. However, at the household scale, if manure would otherwise be burnt as a fuel, anaerobic digestion can increase inputs to soils by retaining some C and nutrients that would otherwise be lost.

Onshore wind, hydropower, solar and geothermal schemes in 2018 provided (2.19 × 10^{19}) J y^{-1} (~3.66% of global energy supply), with a low cost in land area; less that (4.4 × 10^4) km^2 globally (~0.03% of the total land area and ~0.3% of the harvested area). However, if sited on peatlands, windfarms could result in large losses of soil C that are in excess of any fossil fuel C emissions that would be replaced by renewable energy. Therefore, policies are needed to avoid siting energy infrastructure on deep peats.

In order to ensure renewable energy provision does not damage our soils and future capability to produce food, comprehensive policies and management guidelines are needed. These should follow three guiding principles; 1. avoid peats, 2. avoid converting permanent land use to rotational crops, and 3. return all suitable residues remaining from energy conversion processes to the soil.

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Competing Interests

We have no competing interests.