Rothamsted Repository Download

A - Papers appearing in refereed journals

Agostini, F., Gregory, A. S. and Richter, G. M. 2015. Carbon sequestration by perennial energy crops: is the jury still out? *BioEnergy Research.* 8 (3), pp. 1057-1080.

The publisher's version can be accessed at:

- https://dx.doi.org/10.1007/s12155-014-9571-0

The output can be accessed at: https://repository.rothamsted.ac.uk/item/8v083/carbon-sequestration-by-perennial-energy-crops-is-the-jury-still-out.

09/12/2019 13:51   repository.rothamsted.ac.uk   library@rothamsted.ac.uk

Carbon Sequestration by Perennial Energy Crops: Is the Jury Still Out?

Francesco Agostini · Andrew S. Gregory ·
Goetz M. Richter

Abstract Soil organic carbon (SOC) changes associated with land conversion to energy crops are central to the debate on bioenergy and their potential carbon neutrality. Here, the experimental evidence on SOC under perennial energy crops (PECs) is synthesised to parameterise a whole systems model and to identify uncertainties and knowledge gaps determining PECs being a sink or source of greenhouse gas (GHG). For *Miscanthus* and willow (*Salix* spp.) and their analogues (switchgrass, poplar), we examine carbon (C) allocation to above- and belowground residue inputs, turnover rates and retention in the soil. A meta-analysis showed that studies on dry matter partitioning and C inputs to soils are plentiful, whilst data on turnover are rare and rely on few isotopic C tracer studies. Comprehensive studies on SOC dynamics and GHG emissions under PECs are limited and subsoil processes and C losses through leaching remain unknown. Data showed dynamic changes of gross C inputs and SOC stocks depending on stand age. C inputs and turnover can now be specifically parameterised in whole PEC system models, whilst dependencies on soil texture, moisture and temperature remain empirical. In conclusion, the annual net SOC storage change exceeds the minimum mitigation requirement (0.25 Mg C ha^{-1} $year^{-1}$) under herbaceous and woody perennials by far (1.14 to 1.88 and 0.63 to 0.72 Mg C ha^{-1} $year^{-1}$, respectively). However, long-term time series of field data are needed to verify sustainable SOC enrichment, as the physical and chemical stabilities of SOC pools remain uncertain, although they are essential in defining the sustainability of C sequestration (half-life >25 years).

F. Agostini · A. S. Gregory · G. M. Richter (✉)
Department of Sustainable Soils and Grassland Systems, Rothamsted Research, Harpenden, Hertfordshire AL5 2JQ, UK
e-mail: goetz.richter@rothamsted.ac.uk

Introduction

The term 'carbon (C) sequestration' describes processes by which atmospheric carbon dioxide (CO_2) is captured and stored in a long-term reservoir. This review focuses on the role of green plants as principal agents of biologically captured and stored soil organic C (SOC). In particular, we collated evidence for dedicated perennial energy crops (PECs) to sequester C and quantitative data to parameterise SOC turnover models.

PECs are fast-growing (perennial) species that can be grown on marginal soils with low inputs [1, 2] to generate energy either from direct combustion or via conversion to liquid fuels. The area of land devoted to PECs is likely to increase as countries seek to reduce their dependence on fossil fuels and greenhouse gas (GHG) emissions for climate change mitigation. This will require land conversion to energy crops, and questions over how much land is available and where PECs should be grown are debated at present amongst both scientists and policymakers [3]. Pertinent to these is a better understanding of SOC changes associated with different cropping systems, and the quantity and the quality of the residues returned to the soil [4].

PECs are considered to be a nearly C-neutral source of energy [4] as, on average, for each 0.6 kg of fossil fuel C used in cultivation, 1 kg of C is produced as biomass [5]. This estimate, however, can vary greatly according to the value chain and methodology employed, e.g. due to the selected boundary of the life cycle assessment (LCA), to differences in

resource use efficiencies or soil and climate characteristics. In addition, PECs mitigate C emissions by their ability to sequester C from litter, harvest residues and roots into SOC. Although principal differences between PECs in terms of residue returns are known, no conclusions on the impact of harvest on SOC were drawn [5, 6]. It has been proposed that C sequestration under PECs should be at least 0.25 Mg C ha^{-1} year^{-1} in order to make the crop C-neutral when converted to biofuel [7]. To date, estimates of C sequestered under PECs range between 0.6 and 3.0 Mg C ha^{-1} year^{-1} [1, 5]. However, little experimental evidence is available to describe C sequestration after land use change (LUC) to PECs, where in the soil and when SOC will reach its equilibrium and how long it will last if PECs are replaced by other crops.

The critical components that affect C sequestration rate are inputs and turnover rates, but data on these are very limited in PEC systems. The impact of LUC to PECs on SOC dynamics needs to be quantified in terms of changes immediately after crop establishment (resilience phase) and its equilibrium state [8]. Although there have been previous reviews on C sequestration under PECs [8–11], some of these were either unrelated to SOC [12, 13], or did not report any experimental evidence [1, 14, 15], or focused on specific geographical regions [16].

The overall aim of this review was to evaluate the currently available quantitative experimental evidence of C fluxes in low-input PEC systems specific for modelling the transition from arable land. As changes in soils require long-term evidence, we focus on the needs in order to develop and parameterise improved SOC models to simulate such transient

perennial systems. Specific focus for PEC systems is given to (1) defining the system components relevant to C sequestration; (2) collating and synthesising the experimental evidence of C input, turnover and sequestration rates; (3) outlining the modelling tools that simulate SOC dynamics; and (4) identifying the most important uncertainties for the quantification its SOC changes. In contrast to other analyses, we did not consider feedstock production with high nitrogen input, e.g. forage switchgrass [8], but focus on low-input ligno-cellulosic systems [17].

Emphasis has been given here to synthesise current knowledge of the whole system (Fig. 1) and to fill inevitable gaps of evidence for ligno-cellulosic feedstocks from herbaceous and woody crops to improve our understanding of the long-term effects on SOC and to highlight where further data are required.

Components of Perennial Energy Cropping (PEC) Systems

In the following sections, we first lay out the conceptual framework of the two main PEC systems we considered and then define the principal elements affecting their capability to sequester C into the soil (choice of crops, management, environmental control factors and yield) as these formed the basis of the evidence search outlined. Focus was given to *Miscanthus* and willow but, due to the scarcity of some direct experimental evidence in

Fig. 1 Conceptual model and components of carbon (*C*) dynamics under perennial energy crops (*PECs*): soil C pool/continuum; C source pools representing residues dispersed into the soil and their mean residence time (*MRT*), linked process-based models (*1*) soil–plant water dynamic, (*2*) soil biomass dynamic and (*3*) N limitation effect
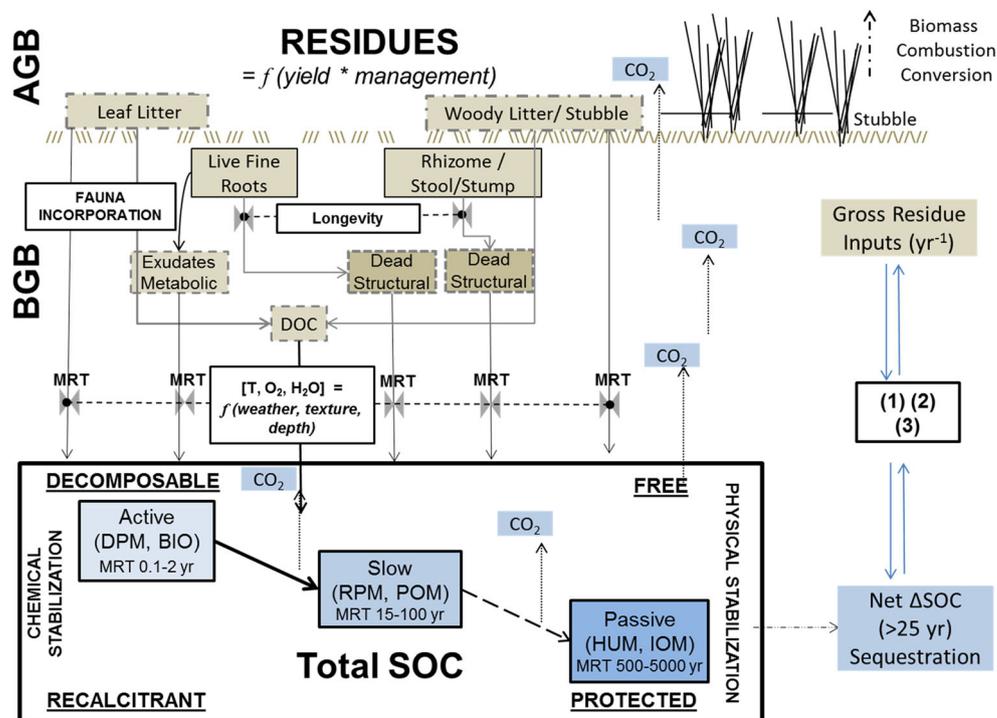
these crops, data from two proxy crops, switchgrass and poplar, have also been used.

PEC Systems

PEC systems consist of herbaceous crops and grasses (HCG) or short-rotation woody crops (SRWC), managed as short-rotation coppice (SRC) or short-rotation forestry (SRF). Worldwide, the main HCGs are C_4 grasses Miscanthus (Miscanthus spp. L.) and switchgrass (Panicum virgatum L.) and C_3 grasses such as reed canary grass (Phalaris arundinacea L.) and giant reed (Arundo donax L.). The main species planted as SRC are willow (Salix spp. L.) and poplar (Populus spp. L.), both being fast-growing. Depending on climate and soil conditions, different species are used for SRF [2, 18]. Miscanthus and switchgrass provide greater yields in warm and temperate regions than any other grasses [2, 18] in spite of Miscanthus showing higher sensitivity to drought. Reed canary grass and giant reed seem better suited to cooler northern Europe and Mediterranean regions, respectively.

Perennial systems are characterised by long-term occupation of land, continuous biomass production with variable harvest cycles (1–15-year duration), continual residue addition to the soil and little disturbance of soil and belowground biomass (BGB). The management of such systems greatly affects residue inputs as harvest practices vary the amount of aboveground biomass (AGB) removed depending on harvest date [19, 20] and harvest method [21, 22]. Eventually, the reversion of the perennials to arable will destroy the BGB in the plough layer, which could be removed or retained, decomposed and humified to become part of SOC (Fig. 1).

Miscanthus, switchgrass, willow and poplar may increase long-term SOC pools due to their extensive root systems [1]. LCA on data from the USA found poplar and switchgrass provided the largest overall GHG sinks [18], with poplar superior due to the fact that in grass systems, nearly all AGB is removed. Miscanthus seems to be the best choice in terms of C sequestration and input efficiency [23] due to its slow decay of residues and high BGB [13]. If Miscanthus is the best option for temperate Europe, switchgrass could be a better choice for dry areas and relatively poor soil quality [2]. The main input of C from switchgrass to SOC comes from its dense root system in the top 30 cm [24], although its residues have a fast C turnover [25].

SRWC PEC systems can increase the C stored belowground for a relatively long time, with contributions from dead wood [16]. Clearance at termination could leave variable amounts of BGB as coarse root and stumps [26]. Willow and poplar are favoured for SRC due to their high growth rates and broad genetic variability, which allows adaptation to different soil and climate conditions [27, 28]. It is likely that the amount of litter deposition is influenced by yield potential [8, 10, 29].

Components of the C Cycle Under PEC Systems

The C cycle of PECs is assumed to be divided into qualitatively different inputs from AGB and BGB (Fig. 1). Whilst harvested AGB will be converted to atmospheric CO_2 by combustion, litter and harvest residues undergo a phase of surface decomposition and incorporation. BGB components will enter the SOC pools dependent on their mortality/longevity [30–32]. Residue quantity, quality, decomposition rates and, hence, SOC dynamics differ between HCG and SRWC systems [33], and this review attempts to collate evidence for the annual gross residue inputs. Residues decompose according to a decay rate (k), dependent on composition, soil protective characteristics, microbial biomass and environmental conditions (temperature, precipitation, soil texture and water availability), which determine the system and compound specific mean residence time (MRT) [34, 35]. The amount of C transferred to soil is a function of several system components [10] such as (i) litter and harvest residues and their decomposition rates; (ii) BGB, its longevity, composition and decomposition rates; and (iii) depth and distribution of the root system. Therefore, each PEC system will affect SOC both directly through the composition and mass of plant residues and indirectly through its impact on the soil environment [36, 37].

Modelling SOC dynamics under PECs requires knowledge of the above-mentioned soil components, ideally integrated with simulated C capture and allocation to crop biomass components to the soil [38, 39]). In view of the main components defined here, our conceptual framework (Fig. 1) aims to align gross input components with SOC fractions of simulation models (see below; [34]). Environmental variables, such as soil water dynamics [40], temperature [38, 41], litter incorporation and dissolved organic C (DOC) transport [42] and nitrogen [14, 43], affect C cycling under perennial vegetation.

Review Method: Meta-Analysis and Data Synthesis

The search for published evidence on the components of the C cycle under PECs was carried out using the web-based search engines ISI Web of Knowledge[SM] (Thomson Reuter, New York, USA) and Google Scholar (Google, CA, USA). We initially used the keywords 'Biomass crop OR Energy crop OR Perennial crop combined with 'GHG', 'Soil C', 'Modelling', 'Residues', 'Litter', 'Root', 'Decomposition', 'Turn over' and 'DOC' covering the time period from 1994 to 2014. Due to the abundance of papers, we refined the search using key genera for bioenergy crops, omitting GHG balance and focusing on C sequestration. We further refined the search by searching for specific perennials, like Miscanthus, Salix/willow, Panicum/switchgrass and Populus/poplar as well as A. donax/giant reed. The literature was organised according to

its methodology and scope in four groups: global inventory, geographic information system (GIS) modelling and/or LCA; process-based modelling; and experimental evidence. Twelve topical groups were distinguished according to the information they could provide in terms of the overall objective (Fig. 2).

Initially, the largest number of papers dealt with assessing GHG emissions or offsets based on LCA or GIS modelling, which carry the main uncertainty that underlying C sequestration models were not specifically calibrated for PECs [44]. Another large subset of papers took a global perspective extrapolating field trials using models assuming a generalised management. Experimental evidence was reported in 85 papers, of which 44 were specific field trials for the PECs under consideration and a total of 57 studies had data useful for parameterising C sequestration (Tables 1, 2 and 3). Although some papers recorded specific fractions of SOC [11, 45, 46], little information was available on parameters affecting SOC accumulation in slowly decomposable or 'recalcitrant' forms [25, 35, 47]. A considerable number of papers using process models were published (47), but few studies modelled C specifically under *Miscanthus* (nine) and willow (six). Some parameter gaps could be filled using data in papers on switchgrass (four) and poplar (four), but it is questionable whether parameters from other woody systems (e.g. evergreen forest) were appropriate for this purpose. Experimental evidence for directly measured GHG emissions was found in 12 papers from eight different experiments, mainly on nitrous oxide (N_2O) from *Miscanthus* [48–51], switchgrass [52, 53], SRC willow and/or SRC poplar, mainly following arable [48, 49, 54] but some also after grassland [55, 56]. Some papers cover other GHGs, like CO_2 and/or methane [48, 49, 53–55, 57].

For the synthesis of experimental evidence, the data were extracted from the articles' tables and figures and scaled to a common unit (Mg C ha^{-1} year^{-1}). We did not account for differences in temperature or other experimental variables. C inputs were either those measured directly or were estimated from accumulated dry biomass data, dividing by length of experimental interval and assuming a C content of 45 % unless authors stated otherwise (Table 1). The MRT of biomass C inputs was calculated from the measured fraction of mass lost during a defined incubation period, expressed in years assuming a linear response (zero-order kinetics). The non-linear decomposition (first-order) rates observed in laboratory incubations were scaled for temperature assuming a Q_{10} of 2, as discussed in the respective paragraph (Table 2). It is important to note that MRT usually represents residue decomposition rather than longevity. When the MRT was calculated from the ratio of new or dead plant parts to standing biomass by direct observation (i.e. roots), MRT represents longevity or mortality [31, 58]. We list retention factors of C inputs retained as SOC where given (Table 3).

Experimental Evidence for Model Parameters

Gross C Input Rates to Soil

Both HCG and SRWC can potentially deliver up to 40–50 % of the C they capture from the atmosphere to the soil as litter and BGB, which can vary from <1 to >4 Mg C ha^{-1} year^{-1} [59–61]. Several authors provide data on C inputs from PECs, generally discriminating between above- (litter) and

Fig. 2 Bar graph describing the meta-analysis of the papers on herbaceous crop/grass (*HCG*) and short-rotation woody crop (*SRWC*) perennial energy crop systems selected and reviewed for the present work. Each paper was assigned to one or more of 12 classes defining its main objective and then further allocated into one of four classes defining the main methodology used
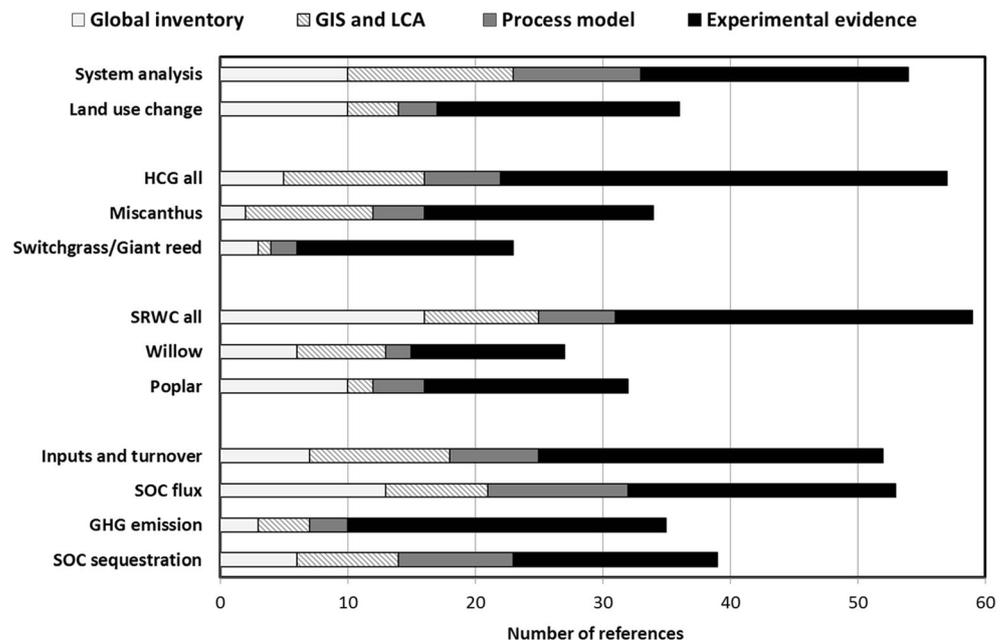
Table 1 Gross carbon input to soil under herbaceous crop/grass and short-rotation woody crop systems from different sources under different environmental conditions. The specific crop systems are *Miscanthus* (principally *M. × giganteus*), switchgrass and giant reed for HCG and willow and poplar for SRWC. Data not available or undisclosed is indicated (n/a)

System	Crop	Source	Age (year)	Depth (cm)	Soil texture	Conditions (MAT; MAP[a])	Method	Input (Mg C ha^{-1} year^{-1})	Reference
HCG	M	Leaf	2	Surface	Silt loam	10.7 °C; 630 mm	C analyser	1.39	Amougou [69]
		Leaf	3	Surface	Silt loam	11 °C; 713 mm	C analyser	1.50	Amougou [64]
		Litter	2–4	Surface	Silty clay loam	9.9 °C; 652 mm	Biomass (43 % C)	0.89	Christian [65]
		Litter	4–8	Surface	Silt clay loam	8.9 °C; 589 mm	TOC analyser	3.20	Beuch [62]
		Litter	14	Surface	Sandy	9.3 °C; 830 mm	Mass spectrometry	3.11	Dondini [135]
		Litter	3.5	Surface	Clayey silt	11.0 °C; 1042 mm	CHNSO analyser	2.17	Anderson-Teixeira [57]
		Rhizome	2–4	0–23	Silty clay loam	9.9 °C; 652 mm	Biomass (43 % C)	1.17	Christian [65]
		Rhizome[b]	3	0–30	Silt loam	11 °C; 713 mm	C analyser	2.66	Amougou [64]
		Rhizome	4	0–30	Sandy	9.3 °C; 715 mm	Biomass (43 % C)	1.70	Himken [85]
		Rhizome	6–8	0–40	Silt clay loam	8.9 °C; 589 mm	Biomass (43 % C)	1.16	Beuch [62]
		Rhizome	7	0–25	Fine silty	11.1 °C; 1023 mm	C analyser	1.40	Dohleman [66]
		Rhizome	14	0–35	Silty clay loam	9.3 °C; 704 mm	C analyser	1.01	Richter [80]
		Root	2–4	0–23	Silty clay loam	9.9 °C; 652 mm	Biomass (43 % C)	0.19	Christian [65]
		Root	3	0–30	Silt loam	11 °C; 713 mm	C analyser	0.25	Amougou [64]
		Root	5	0–120	Fine silty	10–40 % SWC[c]	Root scan	0.68	Monti [61]
		Root	5–6	0–180	Sandy loam	9.3 °C; 715 mm	Biomass (43 % C)	0.86	Neukirchen [84]
		Root	6–8	0–40	Silt clay loam	8.9 °C; 589 mm	Biomass (43 % C)	0.38	Beuch [62]
		Root	7	0–100	Fine silty	11.1 °C; 1023 mm	C analyser	0.55	Dohleman [66]
		Root	14	0–100	Silty clay loam	9.3 °C; 704 mm	Biomass (43 % C)	0.42	Richter [80]
		BGB[d]	14	0–60	Sandy	9.3 °C; 830 mm	Mass spectrometry	2.93	Dondini [135]
		BGB[d]	3.5	0–100	Clayey silt	11.0 °C; 1042 mm	CHNSO analyser	1.09	Anderson-Teixeira [57]
		All residue	9	0–100	Loamy sand	7.4 °C; 706 mm	Mass spectrometry	3.00	Hansen [106]
		All residue	16					3.94	
	SG	Litter	3	Surface	Sandy loam	n/a; 404 mm	C analyser	1.50	Frank [67]
		Litter	3.5	Surface	Clayey silt	11.0 °C; 1042 mm	CHNSO analyser	2.02	Anderson-Teixeira [57]
		Litter	4	Surface	Silt loam	16.0 °C; 1180 mm	CN analyser	1.04	Garten [71]
		Litter	5	Surface	Silty clay loam	n/a; 723 mm	Biomass (40 % C)	2.17	Wienhold [77]
		Litter	6	Surface	Loam	n/a; n/a	Biomass (40 % C)	2.57	Tufekcioglu [72]
		Crown	3	Surface	Sandy loam	n/a; 404 mm	C analyser	3.38	Frank [67]
		Rhizome	7	0–25	Fine silty	11.1 °C; 1023 mm	C analyser	0.47	Dohleman [66]
		Rhizome	4	0–30	Silt loam	16.0 °C; 1,180 mm	CN analyser	0.89	Garten [71]
		root	4	0–90	Silt loam	16.0 °C; 1,180 mm	CN analyser	1.32	Garten [71]
		Root	3	0–110	Sandy loam	n/a; 404 mm	C analyser	1.80	Frank [67]
		Root	5	0–120	Fine silty	n/a; 10–40 % SWC[c]	Root scan	0.77	Monti [61]
		Root	7	0–100	Fine silty	11.1 °C; 1,023 mm	C analyser	0.65	Dohleman [66]
		Root	4	75	Sandy loam	n/a (AL; USA)	n/a	0.81	Bransby, [9]
		Root	3	90	Fine sand	10.5 C; 178 mm	CNS-2000, IRMS	1.29	Collins [47]
		Fine root	6	0–35	Loamy	n/a; n/a	Biomass (40 % C)	0.65	Tufekcioglu [72]
		BGB[d]	3.5	0–30	Clayey silt	11.0 °C; 1,042 mm	CHNSO analyser	1.21	Anderson-Teixeira [57]
		BGB[d]	4	75	Sandy loam	n/a (AL; USA)	n/a	1.63	Bransby [9]
	GR	Root	5	0–120	Fine silty	10–40 % SWC[c]	Root scan	1.21	Monti [61]
SRWC	W	Leaf	4	Surface	Silt loam	9 °C; 981 mm	C analyser	1.16	Pacaldo [131]
			19					1.90	
		Litter	3	Surface	Clay	5.8 °C; 544 mm	Biomass (43 % C)	0.96	Rytter [58]
		Litter	3	Surface	Sand	5.8 °C; 544 mm	Biomass (43 % C)	0.63	Rytter [58]
		Coarse root	4	Surface	Silt loam	9 °C; 981 mm	C analyser	0.34	Pacaldo [131]
			19					0.14	
		Fine root[e]	4	Surface	Silt loam	9 °C; 981 mm	C analyser	0.63	Pacaldo [131]
			19					0.17	
		Fine root[e]	3	0–90	Clay	5.8 °C; 544 mm	Biomass (43 % C)	2.35	Rytter [58]

Table 1 (continued)

System	Crop	Source	Age (year)	Depth (cm)	Soil texture	Conditions (MAT; MAP[a])	Method	Input (Mg C ha^{-1} year^{-1})	Reference
		Fine root[e]	3	0–90	Sand	5.8 °C; 544 mm	Biomass (43 % C)	1.15	Rytter, [58]
		Fine root	2–3	0–70	Sand (washed)	5.6 °C; n/a	Biomass (43 % C)	2.19	Rytter [82]
	P	Leaf	5	Surface	n/a	15 °C; 676 mm	Biomass (43 % C)	2.48	Cotrufo [78]
		Leaf	4–10	Surface	Clay loam	14.3 °C; 964 mm	Dry combustion	1.98	Fang [79]
		Litter	2–4	Surface	Clay loam	3 °C; 463 mm	TOC analyser	0.61	Arevalo [60]
		Litter	3	Surface	Loam	16 °C; 735 mm	Isotopes	2.67	Gielen [99]
		Litter	9–11	Surface	Clay loam	3 °C; 463 mm	TOC analyser	4.68	Arevalo [60]
		Litter	6	Surface	Loam	n/a; n/a	Biomass (40 % C)	1.13	Tufekcioglu [72]
		Stump	3	Surface	Loam	16 °C; 735 mm	Isotopes	0.42	Gielen [99]
		Root	4–10	n/a	Clay loam	14.3 °C; 964 mm	Dry combustion	1.17	Fang [79]
		Coarse root	3	0–40	Loam	16 °C; 735 mm	Isotopes	0.66	Gielen [99]
		Coarse root	2–4	0–30	Clay loam	3 °C; 463 mm	TOC analyser	0.22	Arevalo [60]
		Coarse root	2–8	0–40	Sandy loam	10 °C; 417 mm	Biomass (43 % C)	0.32	Zhang [89]
		Coarse root	9–11	0–30	Clay loam	3 °C; 463 mm	TOC analyser	1.45	Arevalo [60]
		Coarse root	2–12	0–40	Sandy loam	8.4 °C; 204 mm	Biomass (43 % C)	0.16	Yan [93]
		Coarse root	3	0–70	Silt loam	n/a; n/a	Biomass (43 % C)	0.78	Calfapietra [90]
		Fine root[e]	2–4	0–30	Clay loam	3 °C; 463 mm	TOC analyser	0.46	Arevalo [60]
		Fine root[e]	3	0–40	Loam	16 °C; 735 mm	Isotopes	0.94	Gielen [99]
		Fine root[e]	2–8	0–40	Sandy loam	10 °C; 417 mm	Biomass (43 % C)	1.28	Zhang [89]
		Fine root[e]	9–11	0–30	Clay loam	3 °C; 463 mm	TOC analyser	0.83	Arevalo [60]
		Fine root	2	0–15 cm	Sandy	9.5 °C; 726 mm	Soil coring	0.17	Berhongaray [81]
		Fine root	3–15	0–40	Sandy loam	8.4 °C; 204 mm	Biomass (43 % C)	0.03	Yan [187]
		Fine root	7	0–5	Silt loam	n/a; n/a	Biomass (43 % C)	0.02	Abou-Jaoude [152]
		Fine root	6	0–35	Loamy	n/a; n/a	Biomass (43 % C)	0.46	Tufekcioglu [91]
		Fine root	19	0–150	Clay	10.4 °C; 630 mm	Dry combustion	0.06	Upson [92]

C carbon, *HCG* herbaceous crops/grasses, *SRWC* short-rotation woody crop, *M* Miscanthus, *SG* switchgrass, *GR* giant reed, *W* willow, *P* poplar, *n/a* not available, *MAT* mean annual temperature, *MAP* mean annual precipitation

[a] Mean annual temperature and mean annual precipitation, unless stated otherwise

[b] Two to 13 % necrotic

[c] Soil water content (%)

[d] Belowground biomass

[e] Fine roots turnover up to six times annually [32, 58, 83]; we present snapshot values

belowground sources (roots and/or rhizome/stool). Such inputs differ considerably in the literature (Table 1), even for the same kind of residue, mainly due to environment but also due to differences in sampling and analytical methods. The reliability of some data is reduced when C inputs are approximated from biomass yield and an average C content of the residues [30, 62]. In general, fresh and easily decomposable matter, like fine roots and leaves, have a slightly smaller biomass C fraction (43 %) than woody material (46 %) [60, 62].

Aboveground Inputs: Leaves, Litter and Harvest Residues

Aboveground residues from HCG consist of fallen leaves and shoots (stems with leaves) which are seasonally produced before or left after harvest. They may accumulate on the soil surface forming a litter layer. The input range appeared to be quite large as it is assumed to be proportional to yield [63] but also depends on harvest dates [64]. C inputs have been variously measured as leaf litter accumulation on the surface [45, 57, 62, 65], as the difference between peak and harvested yield [19], and harvest residues [66–68]. On average, the C input to soil from HCG litter ranges from <1 to 3 Mg C ha^{-1} year^{-1}. Detailed data on litter C inputs can be approximated from time series of dry matter measurements in an establishing [64, 65, 69] or mature crop [65].

Similar values exist for switchgrass [9, 17, 25, 43, 57, 67, 70, 71] which confirm the knowledge for *Miscanthus* and characterise differences for major land use alternatives. Litter inputs from leaves vary greatly under switchgrass due to management; they were highest in riparian buffer strips [72] and lowest under subsoil amelioration [73] and were affected by land quality [74–76] or water [47]. Litter additions were studied with regard to their effect on SOC fractions [77].

Table 2 Mean residence time of different sources of herbaceous crops/grasses and short-rotation woody crop systems under different environmental or controlled conditions. The specific crop systems are *Miscanthus* (principally *M.* × *giganteus*) and switchgrass for HCG and willow and poplar for SRWC. Data not available or undisclosed is indicated (n/a)

System	Crop	Source	Age (year)	Depth	Soil texture	Conditions[a]	Method	MRT (year)	Reference
HCG	M	Leaf	4–6	Surface	Loamy sand	25 °C; 12 % SWC	Incubation	0.90	Beuch [62]
		Leaf and stem	n/a	Surface	Silt loam	13 °C; 1,444 mm	Litter bag	1.28–1.39[b]	Kim [122]
		Leaf and stem	n/a	Surface	Sandy loam	3 °C; 45 % WFPS	Litter bag	3.19	Magid [108]
		Leaf and stem	n/a	Surface	Sandy loam	9 °C; 45 % WFPS	Litter bag	1.09	Magid [108]
		Stubble	4–6	Surface	Loamy sand	25 °C; 12 % SWC	Incubation	0.63	Beuch [62]
		Litter	2	Surface	Silt loam	15 °C; −80 kPa	Incubation	1.37	Amougou [64]
		Litter	1–3	Surface	Silt loam	10.7 °C; 630 mm	Litter bag	1.85	Amougou [69]
		Litter	n/a	Surface	Sandy loam	15 °C; 20 % SWC	Incubation	0.11–0.12[c]	Ernst [110]
		Litter	n/a	Surface	n/a	n/a	Litter bag	1.96	Yamane [123]
		Rhizome	2–3	Surface	Silt loam	15 °C; −80 kPa	Incubation	1.20–1.40	Amougou [64]
		Rhizome	4–6	Surface	Loamy sand	25 °C; 12 % SWC	Incubation	0.66	Beuch [62]
		Root	2–3	Surface	Silt loam	15 °C; −80 kPa	Incubation	2.40	Amougou, [64]
		Root	4–6	Surface	Loamy sand	25 °C; 12 % SWC	Incubation	1.18	Beuch [62]
	SG	Leaf	2	Surface	Fine loamy	25 °C; 60 % WFPS	Incubation	3.13[b]	Johnson [4]
		Litter	n/a	Surface	n/a	10–16 °C; n/a	Modelled	0.85[d]	Garten, [8]
		Stem	2	Surface	Fine loamy	25 °C; 60 % WFPS	Incubation	3.16[b]	Johnson [4]
		Root	2	Surface	Fine loamy	25 °C; 60 % WFPS	Incubation	3.31[b]	Johnson [4]
		Coarse root	n/a	0–30 cm	n/a	10–16 °C; n/a	Modelled	1.50[e]	Garten, [8]
		Fine root	n/a	0–30 cm	n/a	10–16 °C; n/a	Modelled	0.75[d]	Garten, [8]
SRWC	W	Leaf	1	Surface	Clay	5.5 °C; 660 mm	Litter bag	2.80[f]	Slapokas [111]
		Leaf	1	Surface	Clay	5.5 °C; 660 mm	Litter bag	1.20[g]	Slapokas [111]
		Fine root	2	0–10 cm	Sandy loam	10.7 °C; 800 mm	Litter bag	3.70–7.14[b,h]	Püttsepp [127]
		Fine root	1–5	0–50 cm	Clay	5.8 °C; 544 mm	Rhizotron	0.14–0.25	Rytter [83]
		Fine root	3	0–90 cm	Clay and sand	5.8 °C; 544 mm	Lysimeter	0.15–0.16	Rytter [58]
		Fine root	4	0–50 cm	Clay	5.8 °C; 544 mm	Soil coring	1.06–1.80[i]	Rytter [32]
		Fine root Root (1–2 mm)	2–3	0–70 cm	Sand (washed)	5.6 °C; 550 mm	Soil coring	0.22 0.43	Rytter [82]
	P	Leaf	5	Surface	Loam	15 °C; 676 mm	Litter bag	4.28–5.27	Cotrufo [78]
		Leaf	6	Surface	Silt loam	25 °C; 60–70 % RH	Isotopes	1.25	Rubino [109]
		Fine root	2	0–15 cm	Sandy	9.5 °C; 726 mm	Soil coring	0.42	Berhongaray [81]

MRT mean residence time, *HCG* herbaceous crops/grasses, *SRWC* short-rotation woody crop, *M Miscanthus*, *SG* switchgrass, *W* willow, *P* poplar, *n/a* not available, *SWC* soil water content, *RH* relative humidity, *WFPS* water-filled pore space

[a] Temperature and hydrological conditions, where these are either mean annual precipitation (in mm) in the field or soil water content, relative humidity, water-filled pore space and matric potential (*in kPa*) in the laboratory

[b] Calculated from dry matter loss

[c] Incubated with earthworms

[d] Model parameter

[e] Model parameter, based on Gill and Jackson [104]

[f] One-millimetre mesh size

[g] Four-millimetre mesh size

[h] The range covers different varieties

[i] Calculated by mortality/growth ratio

Under SRWC, litter production varies across species and genotypes, as leaf litter is proportional to LAI, and generally accumulates at the end of the season in a layer on the soil surface before being progressively incorporated into the top 10 to 15 cm of soil [78]. The range of C inputs from litter in SRWC (0.63 to 4.68 Mg C ha^{-1} year^{-1}) is only slightly larger

Table 3 Change in soil organic carbon and retention in soil under herbaceous crops/grasses and short-rotation woody crops systems from different sources under different environmental conditions. The specific crop systems are *Miscanthus* (principally *M. × giganteus*), switchgrass and giant reed for HCG and willow and poplar for SRWC. Data not available or undisclosed is indicated (n/a)

System	Crop	Source	Age (year)	Depth (cm)	Soil texture	Conditions (MAT; MAP[a])	Method	ΔSOC (Mg C ha^{-1} year^{-1})	Retention (%)	Reference
HCG	M	Leaf	3	Surface	Silt loam	15 °C; −80 kPa[c]	C analyser	0.40		Amougou [64]
		Rhizome	3	0–30	Silt loam	15 °C; −80 kPa[c]	C analyser	0.47		Amougou [64]
		Root	3	0–30	Silt loam	15 °C; −80 kPa[c]	C analyser	0.15		Amougou [64]
		All residue	3	0–30	Loamy sand	10.3 °C; 1,048 mm	Isotope ratio	0.60–0.72		Zimmermann [134]
		All residue	3.5	0–100	Clayey silt	11.0 °C; 1,042 mm	C balance	2.36	72	Anderson-Teixeira, [57]
		All residue	6	0–30	Sandy loam	13 °C; n/a	Isotope ratio	1.25–1.52		Zatta [86]
		All residue	9	0–100	Loamy sand	7.4 °C; 706 mm	MS	0.78	26	Hansen, [106]
		All residue	9	0–60	Silty clay loam	13.3 °C; 700 mm	CHNO; IRMS	2.08[d]		Cattaneo [140]
		All residue	14	0–30	Silty clay loam	10.3 °C; 704 mm	IRMS	0.41–0.46		Richter [80]
		All residue	14	0–60	Sand	9.3 °C; 830 mm	IRMS	3.20[e]	53	Dondini [45]
		All residue	15	0–30	Sandy loam	9.9 °C; 1,004 mm	MS	0.59	21.5	Clifton-Brown [19]
		All residue	16	0–100	Loamy sand	7.4 °C; 706 mm	MS	1.13	29	Hansen [106]
	SG	Litter	5	0–7.5	Silty clay loam	n/a; 723 mm;	CN analyser	0.89	41	Wienhold [77]
		BGB[b]	5	0–100	Sandy loam	n/a	n/a	1.10		Bransby [9]
		Root	3	0–90	Sand	10.5 °C; irrigated	IRMS	0.35	27 [h]	Collins [47]
		All residue	2	0–30	Silt loam	16.0 °C; 1,180 mm	CN analyser	0.40–0.85		Garten [71]
		All residue	3.5	0–100	Clayey silt	11.0 °C; 1,042 mm	C balance	2.28	70.6	Anderson-Teixeira, [57]
		All residue	4	0–90	Silty clay loam	6.3 °C; 602 mm	CNS analyser	2.40[d]		Lee [75]
		All residue	7	0–40	Silty clay loam	9.9–11.4 °C; 856–1,092 mm	CN analyser	1.00–2.57	50.1[i]	Bonin [137]
		All residue	9	0–30	Silty clay loam	n/a	IRMS	0.66[d]		Follett [138]
		All residue		0–150	Silty clay loam	n/a	IRMS	2.05		Follett [138]
		All residue	23	0–60	Loam subsoil	10.5 °C; 914 mm	Combustion	3.71[f]		Al Kaisi [73]
	GR	All residue	9	0–60	Silty clay loam	13.3 °C; 700 mm	CHNO; IRMS	3.62[d]		Cattaneo [140]
SRWC	W	Litter	22	Surface	Clay	5.8 °C; 544 mm	Extrapolated[j]	0.17	23	Rytter [26]
		Fine root	22	0–50	Clay	5.8 °C; 544 mm		0.24	12	
		Fine root	2–3	0–70	Sand (washed)	5.6 °C; n/a	Modelled	0.22		Rytter [82]
		Litter and root	7	0–40	Silty clay loam	9.9–10.9 °C; 856–1,057 mm	CN analyser	1.14, 3.57[g]		Bonin [137]
		All residue	12	0–25	Loamy sand	9.3 °C; 595 mm	C analyser	0.22		Hellebrand [54]
		All residue	12	0–25	Loamy sand	9.3 °C; 595 mm	C analyser	0.34		
		All residue	5–19	0–45	Silt loam	9 °C; 981 mm	C analyser	−0.06		Pacaldo [143]
	P	Litter	22	Surface	Clay	5.8 °C; 544 mm	Extrapolated[j]	0.22	23	Rytter, [26]
		Fine root	22	0–50	Clay	5.8 °C; 544 mm		0.30	12	
		All residue	4–11	0–50	Clay loam	3 °C; 463 mm	C analyser	2.29		Arevalo [60]
		All residue	12	0–25	Loamy sand	9.3 °C; 595 mm	C analyser	0.23		Hellebrand [54]
		All residue	12	0–25	Loamy sand	9.3 °C; 595 mm	C analyser	0.53[d]		
		All residue	15	0–40	Sandy loam	10 °C; 417 mm	Wet oxidation	0.13[g]		Zhang [89]
		All	19	0–150	Clay	10.4 °C; 630 mm	Dry combustion	0.47[g]		Upson [92]

ΔSOC change in soil organic carbon, *HCG* herbaceous crops/grasses, *SRWC* short-rotation woody crops, *M Miscanthus*, *SG* switchgrass, *GR* giant reed, *W* willow, *P* poplar, *(n/a)* not available, *MAT* mean annual temperature, *MAP* mean annual precipitation

[a] Mean annual temperature and mean annual precipitation, unless stated otherwise

[b] Belowground biomass

[c] Matric potential (in kPa)

[d] Soil fertilised with N

[e] Including 0.81 Mg C ha^{-1} year^{-1} in soil micro-aggregates

[f] Re-cultivated subsoil

[g] Soils under short-rotation forestry

[h] Retention based on root input only

[i] Retention based on yield

[j] From Rytter [58]

than those from HCG. Consistent across species and different locations, observed inputs seem to increase with stand age [60, 78], which is not reflected in long-term extrapolations [26]. There is also dieback of smaller stems especially during the first year [79] which can be a few percent of the biomass. Harvest method can have a profound impact on losses, but woody harvest residues <5 % seem achievable [22].

Belowground Inputs: Root and Rhizome

Data for C inputs from BGB are difficult to interpret, especially for roots because they are based on many single observations [45, 61, 62, 64, 80] rather than time series [26, 32, 81–83]. Due to differences in stand age, depth and sampling frequency inputs appear very variable [65, 66, 84]. On average, BGB inputs appear similar to that from litter under *Miscanthus*, switchgrass and willow but much smaller under poplar. There is some indication that inputs increase over time [60]. Inputs from roots may depend on soil texture and other environmental factors, but the dataset is too small to derive a rule; however, longevity of roots and their turnover depends on root diameter [30, 31].

For HCG, authors differentiate the C contribution to soil between those from the rhizome, a storage organ often present in grasses used as biomass crops, and from fine, fibrous roots. The latter contributes less than 25 % of the apparent annual BGB input under *Miscanthus* (Table 1 and Fig. 4). BGB is concentrated in the upper part of the profile [61]; in spite of large amounts of roots under *M. giganteus* observed down to 1 to 1.8 m [57, 61, 66, 80, 84], more than 50 % were in the top 30 cm. The C inputs from *Miscanthus* rhizomes (1.01 to 2.66 Mg C ha^{-1} year^{-1}) were generally greater [57, 61, 66, 80, 85] than those from roots (0.19–0.86 Mg C ha^{-1} year^{-1}, Table 1). For switchgrass, more C was allocated to the roots than to the rhizomes [25, 66, 71]. Some studies did not separate rhizomes from roots [9, 45, 57, 86] or ignore them completely [47, 61, 67, 72].

Fewer data were available for SRWC systems due to the practical difficulties of excavating and studying tree roots. For different species in SRWC systems, 25–35 % of total tree biomass was found in the roots [87], with 10 % specified as fine roots, confirmed for a 10-year-old poplar plantation [5]. Several authors investigated fine roots from willow and poplar grown as SRC [5, 30, 81, 82, 88, 89] and SRF [72, 90–93], values for the latter being lower (Table 1). Quantification of fine roots is challenging, and indirect methods were proposed although roots concentrate in the upper 5–10 cm of soil [94]. Variable fractions of willow biomass can be found belowground dependent on stand age and soil texture [58]; 48–58 % of total biomass was allocated in the first year and 40 % allocated in succeeding years [26]. Observed values of C input from fine roots of SRWC poplar range between 0.46 and 1.28 Mg C ha^{-1} year^{-1}, and between 0.16 and 1.45 Mg C

ha^{-1} year^{-1} can be found in coarse roots (Table 1). It was suggested that about 20 % of SRC poplar biomass was allocated to structural roots [95] which may contribute significantly to biomass C, whilst fine roots contribute to SOC due to their faster turnover compared to structural components [26, 82]. Average fine root inputs are higher under willow than poplar (Fig. 3b).

Root Exudates

Little is known about the effect of root exudates on SOC. Only two papers have been published with root exudate data from *Miscanthus* [96, 97]. Kaňova et al. [97] have quantified and qualified exudates from roots of 15-year-old plants. Based on their hourly flux measurements, C exudate inputs from *Miscanthus* could be up to 0.5 g g^{-1} living root produced annually. Based on root biomass data (Table 1), C exudates and 2-year MRT (Table 2) and assuming 210-day season with an average 12-h day length roots could contribute between 0.4 and 1.7 Mg C ha^{-1} year^{-1} to the C balance. Hromadko's data [96] were an order of magnitude larger than those presented by Kaňova [97] which seems unrealistically large. Further investigations showed that these root exudates contributed to soil respiration and may affect SOC pools in multiple ways.

C measured in the rhizosphere of 2-year-old willows, grown in the glasshouse using stable isotopes (^{14}C) pointed to high rhizodeposition of C in older plants which was not proportional to the root production [98]. This C turned over very fast and only 2 % was found in the soil microbial biomass. However, 11 % of rhizodeposits were retained in the soil organic matter. For a 3-year-old poplar stand, it was suggested that C from fine roots (including exudates) could stimulate the microbial biomass and induce turnover of already stored SOC [99]. Rhizodeposition and root exudates can cause a priming effect on the turnover of C stored in the soil [100], which could increase soil C emission [101, 102] and lower the sequestration rate. However, living roots can decrease (50 %) or increase (380 %) SOC mineralisation rates causing losses equivalent to the root C input to soil, the range depending on plant species and age [103]. The loss of existing SOC increased with root biomass in a poplar plantation subjected to free air C enrichment (FACE) [99] but did not appear to impair SOC increase.

In summary, gross inputs from leaf and stem litter and harvest residues are more tangible than from BGB. The production of fine roots and root exudates could vary greatly between species, genotypes, management and the age of the perennial and is tedious to assess. There is little quantitative knowledge about the site-specific interaction between the size of the residue inputs and SOC and how they were affected by management (e.g. fertiliser, soil fertility, harvest). Inputs from perennials are continuous, vary over their life time and their

layers. Coarse and dense roots of old undisturbed grassland turn over much more slowly (0.92–1.32 years) than fresh inputs [125].

In SRWC systems, similar discrepancies due to experimental procedures occur: Only 15–18 % of the original mass was lost from litter bags over 8 months [78], whilst a loss of 80 % of the same litter was estimated during an 11-month period using ^{13}C-labelled material [109]. These losses translated into MRTs about four times longer (4.3–5.3 years) than those determined from isotope ratios (1.15 years) and about twice those calculated when accounting for the 10 °C difference in incubation temperatures. In situ decay rates of leaves were much higher when using large instead of small mesh litter bags [111]. Leaf material accessible to earthworms almost completely disappeared from the large mesh litter bags within 1 year (95 %; MRT ~1 year), whilst material accessible only to fungal decomposition only lost between 40 and 80 % (MRT 2.5 to 1.25 years).

It is disputed whether fine roots contribute most to SOC in woody systems due to their relatively rapid decomposition [5, 99, 126]. However, fine willow [58] and poplar roots [81] were characterised by a high turnover rate (Table 2). However, the average production of fine roots under poplar was smaller compared to willow (Fig. 3b; 0.56 vs. 0.99 Mg C ha^{-1} year^{-1}). Turnover times for willow roots [127] determined from a mass loss (15–25 %) in litter bags were much longer than those determined using other methods (Table 2).

The longevity of fine willow roots can range between 55 and 350 days [32, 58, 83]. Similar values were found for fine poplar roots [30]. Coarse roots (CR) represent a variable fraction of the root biomass (10 and 40 %; [58, 99]), but much smaller values emerged recently [82]. Very coarse, anchoring roots are likely to have a long MRT (e.g. lifetime of the stand), but there are no reports on decomposition of those [81, 82, 99].

In summary, a good number of contrasting results regarding residue turnover was compiled from the literature, which enables initial parameterisation of plant–soil interaction and C flux models (see below). However, there is relatively little known about the in situ turnover of live rhizomes and coarse roots as longevity becomes an additional level of complexity. The large range of MRT reflects the experimental conditions, long MRT being an artefact of litter bag usage. However, a small number of studies using isotopes [109], model systems [110, 111] and field sampling [32, 81, 82] allow their scaling to in situ rates.

Carbon Retention and Storage in Soil

The factors affecting the soil C balance are numerous and vary depending on climate, soil conditions and crop management, e.g. harvest date and residue removal [62, 128]. Integrated studies on C enrichment, e.g. POPFACE, have shown that increased atmospheric CO_2 can induce a decrease of SOC [78] attributed to a priming effect [99, 102, 129, 130]. However, physical protection and chemical adsorption, both controlled by soil texture (e.g. clay content and cation exchange capacity), may play an important role for long-term C retention [4, 34, 35, 101]. C retention is the net result of annual gross inputs/production, mortality and turnover of the components.

Figure 3 synthesises the average gross inputs (Table 1) and their MRT (Table 2) for each residue type, distinguishing litter and roots. It is important to realise that inputs were often based on snapshots, whilst detailed time series were the exception, especially with regard to root production and turnover [32, 81, 82]. Grasses and poplar have similar C inputs from AGB litter, larger than willow SRC (~2 vs. 1 Mg C ha^{-1} year^{-1}) with a wide range of MRT (<1 to >4 years), some of which may be overestimated. Within the SRC data, there is more quantitative evidence from poplar [60, 72, 78, 79, 99, 109] than for willow [58, 111, 131].

For the BGB, inputs from HCG roots and rhizomes needed to be separated, which was not always done [45, 57, 86, 106]. Annual C inputs from *Miscanthus* roots are about half that of switchgrass with similar turnover time. Rhizome inputs from switchgrass were much smaller than from *Miscanthus* which explains the evidence that switchgrass sequestered twice the
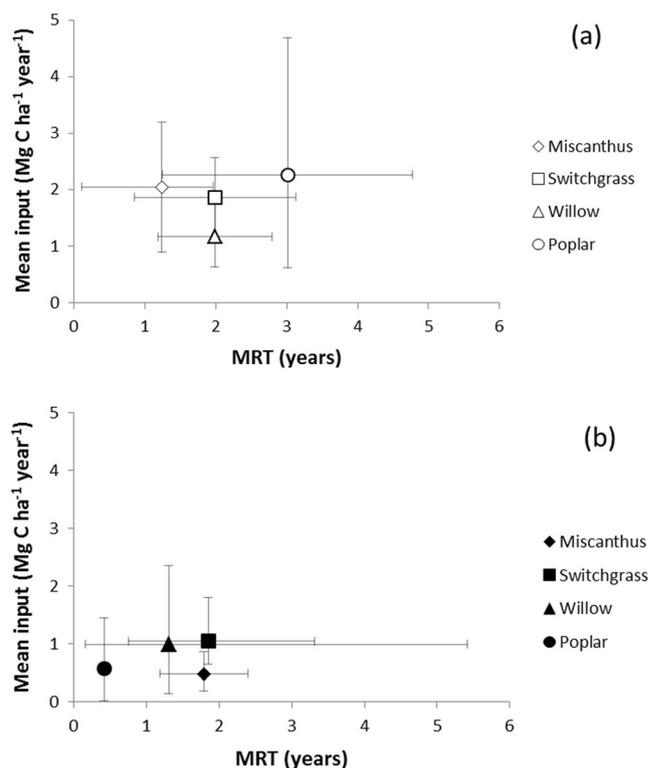
Fig. 3 Mean C input and mean residence time (*MRT*) from **a** leaf litter and **b** roots of perennial energy crops as derived from the data in Tables 1 and 2, respectively. The *bars* show the maximum and minimum values in the range

amount of SOC (Fig. 4). In spite of methodological uncertainty regarding turnover of fine roots, the MRT of HCG was about four to five times longer than that of SRWC (Fig. 3b). The annual average C input to the soil as fine root under SRWC is between 0.4 and 1 Mg C ha^{-1} year^{-1}, but their fast turnover is diminishing the contribution to SOC storage and retention to about 10 % [82]. However, what really makes the difference between the two systems in addition to the difference of SRC fine roots turnover is the dynamic of the rhizome.

Although annual rhizome inputs are similar to those from litter, its MRT is large due to its longevity [1, 68] which delays their decomposition and incorporation into SOC. The analogue in the SRWC system is to be found in the coarse roots (1–2 mm) with longer MRT [82] or woody roots, the stump or stool [4] for which no evidence of turnover and retention exists.

Evidence for Long-Term C Storage Change

Potential, default C sequestration rates for arable crops, grassland and forests have been estimated to be 0.3, 0.5–0.7 and 0.1–0.2 Mg C ha^{-1} year^{-1}, respectively [132]; others found 0.33 Mg C ha^{-1} year^{-1} for both grassland and forest systems [133]. The net SOC storage change under PECs observed in most recent studies exceeds these values (Table 3). The C effectively sequestered under *Miscanthus* (% retention) ranges between 21 % [19] and ~70 % [57] of the gross C inputs. Across different environments and time periods, quite similar net storage changes could be found (3 to 15 years) [19, 134]. However, a comparison across similar environments indicates that with increasing age (9- to 16-year-old *Miscanthus* stands), the amount of annual SOC increment rises by more than 40 %
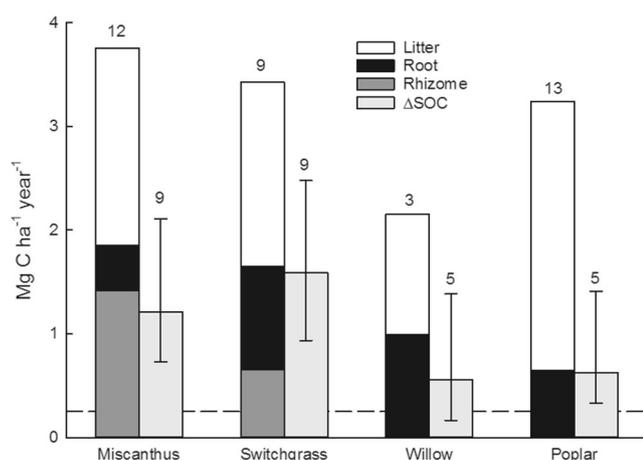
Fig. 4 The mean input of carbon (*C*) (*stacked column*) and the mean change in soil organic C (*ΔSOC*) (*single column*) from perennial energy crops as derived from the data in Tables 1 and 3, respectively. The *bars on the ΔSOC* column show the maximum and minimum values in the range, and the *numbers above all the columns* give the numbers of references considered

[106]. High rates of SOC were sequestered in a soil very low in C [135] compared to those established on grassland sites (e.g. [19, 86]. Assuming steady-state C inputs [135] could lead to an underestimation of MRT [136]. Yields and litter inputs increased during the establishment phase [65] and the postulated delay for rhizome decomposition [106] contradict such assumptions.

Under switchgrass, a large number of studies [8, 9, 47, 57, 75, 77, 137, 138] showed sequestration rates, which were similar (Table 3) but higher on average (Fig. 4). Retention rates were also higher on average (50 %), probably due to large fraction of root inputs.

In-depth analysis supports the concept of an initial resilience phase of SOC after LUC to PEC in which *Miscanthus*-derived SOC did not fully replace the continuing turnover of resident C during early establishment. Especially when planted into grassland soils [86, 139], the initial residue input and turnover rates from PEC are too low to compensate for the loss. Zimmermann et al. [141] also showed that BGB has a greater effect on SOC than litter inputs, possibly due to delayed incorporation of the latter. Early estimates of long-term SOC stock changes based on a single observation [19, 135] need to be revised using a time series with early phase evidence [86, 134, 139] and accounting for spatial heterogeneity [46, 57, 80, 86]. Sequestration rates are not predictive when based on single early or late observations [19, 135, 140], average C inputs and turnover rates.

C inputs originate from and feed into pools of different stability [45, 141], and initially, C is stored as particulate organic matter (POM) [139], which is a relatively slow SOC pool under switchgrass [25, 47]. This could explain the lack of real enrichment of recalcitrant or mineral-associated SOC during the first 4 or even 6 years under *Miscanthus*, especially when planted into former grassland [86]. These results are in line with findings for switchgrass [25]. Losses of SOC under *Miscanthus* genotypes with higher BGB were suggested to indicate higher soil respiration rate and C turnover induced by root exudates, causing a priming effect [86]. Considering the enrichment in the labile fraction [86, 135, 139], it was suggested that at least 20 years was needed to evaluate the impact of *Miscanthus* in terms of sustainable SOC enrichment, which contradicted earlier estimates [19, 45, 106, 135]. With regard to verifying the simulation of sequestration dynamics, it seems necessary to have observations for inputs and changes of both total and specific SOC fractions along the whole growth period. For switchgrass, similar average inputs appear to cause an overall 25 % higher average stock change (ΔSOC, Fig. 4) than *Miscanthus* (1.59 vs 1.21 Mg C ha^{-1} year^{-1}), likely due to the lower root inputs from *Miscanthus* (Fig. 3b).

Based on long-term field trials for SRWC and accounting for all residues, overall lower C sequestration rates were calculated for willow and poplar (Table 3; average 0.56 and 0.63 Mg C ha^{-1} year^{-1}, respectively) than for HCG systems.

An increased annual SOC storage change was observed in response to fertilisation, particularly in poplar stands [54]. These low-storage changes were supported by findings for SRF using labelled C [142], which also showed that new C accumulated in SOC pools was characterised by high turnover rates. However, ΔSOC under SRF/SRC changed within 11 years from an initial loss of 0.74 during the first 2 years to an increase of 5.82 Mg ha^{-1} year^{-1}, during the last 2 years [60]. For Table 3, we interpolated after the initial loss an average annual SOC gain of 2.29 Mg ha^{-1} year^{-1} between years 4 and 11, which over the whole period would have been an increase of only 1.87 Mg ha^{-1} year^{-1}. These numbers are similar to the average gains reported for SRC poplar earlier (1.63 [68] to 2.43 C Mg ha^{-1} year^{-1} [10]). These sequential long-term measurements are essential to extrapolate results from short-term controlled experiments to long-term field rotations (e.g. Rytter [26, 58]; Table 3). New data show high sequestration rates for switchgrass on arable soils [137] but no long-term change for SRC-willow on former grassland soils [143].

In conclusion, the variability of ΔSOC under PECs reflects the uncertainty due to the limited periods of measurements, a series of simplifying assumptions (linearity, steady state) and environmental variability (see above). Based on Table 3, the C retention is higher in herbaceous (*Miscanthus*) than woody PEC systems (21–72 vs. 12–23 % retained of original input), which is confirmed by comparing independent averages (Fig. 4). Discrimination between inputs from above- and belowground residues (Tables 1 and 2) suggests input- and plant-specific C retention factors (Fig. 3). The view that roots contribute more to SOC than aboveground sources [100] seems to be reflected in Figs. 3b and 4, and recent data showed a tight relationship between root and *Miscanthus*-derived SOC [80]. The average sequestration rates (Fig. 4) are largely based on all residues and require further analysis in terms of component-specific contributions.

Decomposition and Stabilisation of SOC Pools

The stabilisation of SOC pools depends essentially on the relation between soil temperature [41], chemical composition [144] and physical protection. The latter defines their kinetic properties and temperature sensitivity [145, 146]. The least decomposable residues seem to be the most sensitive to temperature [120]. Particulate (POM) and mineral-associated organic matter (MOM) are formed in the soil, which stabilise SOC, e.g. limiting the access of soil microbes [25]. Hansen [106] found 65 % of *Miscanthus*-derived C stabilised in POM. Occluded POM is particularly stable, and root C has been found to accumulate in this pool even under arable crops [147]. The amount of C occluded, which is considered as a main index of long-term sequestered C in soils under forest [148] as well under *Miscanthus* [11], remains to be quantified.

The intra-aggregate fraction of POM [45, 139] is small but could be a close approximation.

Lignin as a biomass component together with polyphenols reduces decomposition rates [149]. However, lignin decomposition rates are highly variable depending on soils and land use and are not necessarily linked to the stability of the derived soil C pool [146, 149]. Recalcitrant SOC components, like lignin, increase with depth and clay content due to chemical binding [35, 149]. The bio-physicochemical complexity of SOC could explain its turnover variability under PECs. *Miscanthus*, whose litter is higher in lignin than other grasses [69], seems to reduce the turnover rate of existing SOC by increasing the insoluble C fraction resistant to microbial attack [116, 150]. On the other hand, a priming effect was suggested for *Miscanthus* root exudates [86], and a two- to fourfold increase in the mineralisation rate was reported after establishment of switchgrass [43]. For forest systems [142], it has been shown that new C inputs from plant residues can enter relatively fast decomposable SOC pools, particularly if the soil had a low clay content [148]. Under SRWC systems, the predominance of decomposable fine roots could explain the initial loss of SOC [60] and higher losses under FACE [99].

CO$_2$ Emission and Other Losses

In general terms, C losses from soil come from root respiration (ca. 20–40 % of plant-fixed C; [151]), microbial respiration (40 % of SOC in decomposable pools [151]) and leaching of DOC. Pieces of evidence for losses from SOC pools using measurements of CO$_2$ emissions are considered essential to explain the complex dynamics of C fluxes [66] and were included in several studies [48, 49, 52, 53, 55, 57]. Earlier estimates of C emissions from PECs are usually derived from models, and typically very little CO$_2$ is emitted from *Miscanthus* stands of different ages [59]. N$_2$O emission was 4 to 6 Mg CO$_2$-equivalent higher after grassland conversion to willow and poplar, but CO$_2$ fluxes were 30 to 40 % less than on the grass reference [55]. Conversion from arable to *Miscanthus* or switchgrass confirms higher CO$_2$ emissions under grassland (e.g. prairie) [57] reflecting its higher biological activity which eventually increased SOC stocks. N fertilisation in switchgrass actually reduced the GHG emission per unit harvested biomass [53]. Under SRWC systems, coppicing itself can result in a 50 % increase in CO$_2$ emission, mainly due to live root respiration [152]. Root respiration has also been found to increase in SRWC systems under enhanced atmospheric CO$_2$ concentrations [130], which seems to be generally valid for FACE experiments under older (12–16 %) and recently established (22–46 %) systems [153].

DOC typically comprises compounds of low molecular weight. Exudates (the main source of DOC) can be adsorbed by the soil, abiotically decomposed, volatised or leached. DOC could, in spite of its small size, be of relevance under

PECs due to the extensive root system and large rhizome biomass. To date, there have been no specific studies published to quantify DOC in PEC systems. De Neergaard et al. [98] found that willows allocate between 1 and 10 % of plant C to root turnover and exudation, similar to those found for other plants [154]. DOC persistence or loss by leaching was mentioned for *Miscanthus* only qualitatively [155]. However, under grassland, 20–30 % of DOC was rapidly decomposed by microbial respiration, <5 % was retained in the soil, whilst the rest was assimilated by the soil microbial biomass [156]. Similar conclusions were reached for DOC in forest soils [157], where the soil microbial biomass assimilated 70 % of the DOC and respired the remaining 30 %.

In summary, under PECs, the main difficulty in assessing C turnover and retention is the uncertainty regarding the physical and chemical stability of SOC pools. It is the stability of these pools which defines the sustainability of soil C sequestration and eventually the effective C neutrality of the system [125, 142]. The half-life of >25 years considered for a C pool to be 'inert' [121], and similar times as a criterion for SOC sequestration [5, 35, 151, 158] seems rather short. They are in contrast to much longer times assumed for the 'inert' and stabilised humus fractions in SOC turnover models [159, 160]. From these considerations, the postulate arises that sequestration can only be assessed if a time series of long-term measurements of these pools are available (e.g. maintaining monitoring and demonstration trials for PECs).

Modelling Approaches Adopted for SOC Under PEC Systems

When assessing the effects of land use change on SOC turnover, model complexity is dependent on data availability and process understanding at different scales. For PEC systems, the evidence base is still emerging and is too sparse to model the process dynamics, let alone to predict a state of equilibrium [8]. For large-scale applications, process-based models were simplified, based on a series of assumptions (e.g. steady state) and controlled by empirical inputs (e.g. yield maps [63]). In the following, we will review process-based models of varying degree of complexity/empiricism and their actual application to PEC systems. We will use an expanded semi-empirical process model to rank the importance of the key inputs and parameters presented above.

Process-Based Models

There are several papers which review models applied to the C balance [14, 15] or GHG emissions [12, 16, 30] under PECs [161]. The most recent review on models relevant to PEC systems screens complex agro-ecosystem models with a focus on N_2O emission [15]. CENTURY [160] and RothC [159] were identified as especially important. The principles of C fluxes under land use change from arable to grassland and forestry systems certainly apply to the transition to PEC as shown recently using a hybrid C and N model including routines of CENTURY [162].

A total of 25 papers modelling C in PECs systems were published between 2000 and 2013: 16 used RothC and derived models, five used CENTURY or DayCENT and two used both models (Table 4). Two papers dealt with a generic description of the C balance under poplar [10] and switchgrass [8]. The popularity of RothC may lie in the minimal data requirement, its adaptability, which allows its integration, and the easier calibration of residue pools compared to CENTURY [163]. Authors have modified and coupled RothC by adding and calibrating new specific soil and residue C pools, accounting for tree [164] and crop [165] residues. RothC has been coupled with specific crop growth and litter degradation models to simulate SOC dynamics under *Miscanthus*, willow and poplar SRC [166]. Dondini et al. [45] applied a soil fractionation technique [141] to improve the calibration of SOC pools and to account for the recalcitrant nature of plant residues. Within FullCAM [166, 167], RothC was used in woodland systems by applying debris partitioning and decomposition rates measured in agricultural soils. However, previous work [168] showed that the C cycle in forest systems needed a chemical model for litter decomposition to integrate RothC with plant residues. They also suggested locally calibrated decomposition rates to account for soil temperature and water content gradients [166]. However, the question is whether a litter layer is mandatory in a SRWC system when managed as SRC, as surface litter in spite of its chemical composition seems to be incorporated within a season [111].

C Dynamics as a Function of Soil Depth

The release of C from litter decomposition is a function of different factors, mainly controlled by its incorporation into the soil and by soil depth, which induces temperature and soil moisture gradients. CENTURY and APSIM decrease the decomposition rate of litter accumulating on the soil surface. Soil water dynamics are considered a key parameter in all SOC models applied to PECs [14, 15]. The models APSIM, CENTURY and STICS include a component for litter decomposition using a rate-modifying function for soil depth, which account for changes of temperature and moisture along the soil profile. Jenkinson and Coleman [40] introduced a similar empirical coefficient to reduce SOM decomposition in the subsoil. In addition, decomposition of old SOM can be affected by root priming [103], which varies with root density along the soil profile and crop C balance [99]. In summary, depth-dependent decomposition has been implemented by

Table 4 Use of RothC and CENTURY models and derived routines in simulating C dynamic under herbaceous crops/grasses (HCG) and short-rotation woody crops (SRWC) grown as PECs and similar systems

Scale	System	Species	Soil	Location	Model	Objective	Duration	Output kind	Output unit	Paper
Laboratory	HCG	*Miscanthus* ryegrass	Sandy	DK	CENTURY	Rate calibration	n/a	ΔSOC	% C loss	Foereid [116]
		Mix grass	Quartz sand	n/a	RothC	Rate calibration	n/a	Decomposition rate	Year^{-1}	Hoffmann [113]
		Alpine grass	Leptosol	CH	RothC	Rate calibration	Equilibrium	ΔSOC	Mg C ha^{-1}	Leifeld [174, 175]
		Alpine grass	48 soils	CH	RothC	Pool calibration	n/a	ΔSOC	Mg C ha^{-1} year^{-1}	Zimmermann [141]
Field	HCG	*Miscanthus*	Sandy loam	IRL	RothC	C sequestration	14	ΔSOC	Mg C ha^{-1} year^{-1}	Dondini [135]
		Miscanthus	n/a	USA	DAYCENT	Mod. comparison	9	ΔSOC	C g m^{-2}	Davis [59]
		Miscanthus	Clay loam	PL	CENTURY	C sequestration	2	ΔSOC	Mg C ha^{-1} year^{-1}	Poeplau [11]
		Miscanthus	Different soils	NL; DK; CH; D	RothC	C sequestration	>10	ΔSOC	Mg C ha^{-1} year^{-1}	Poeplau
		Miscanthus	Sandy loam	UK	RothC	C sequestration	7	ΔSOC	Mg C ha^{-1} year^{-1}	Zatta [86]
		Alpine grass	Silty loam	CH	RothC	C sequestration	6	ΔSOC	C g m^{-2}	Niklaus [129]
		Grass	Different soils	IRL	RothC	C sequestration	40	ΔSOC	Mg C ha^{-1} year^{-1}	Xianli [163]
		Switchgrass	n/a	USA	DAYCENT	Mod. comparison	9	ΔSOC	C g m^{-2}	Davis [59]
	SRWC	Olive	Clay loam	E	RothC	Management	30	ΔSOC	Mg C ha^{-1} year^{-1}	Nieto [164]
		Pinus	Chromosol	AUS	RothC	Management	20	ΔSOC	Mg C ha^{-1}	Paul [167]
		Pinus	Sand to clay loam	AU	RothC	System analysis	1–2	Litter loss	% loss	Paul [166]
		Pinus		AU	RothC	System analysis	40	ΔSOC	Mg C ha^{-1}	Paul [168, 172]
		Pinus	Sandy soils	E	Both Mod	Mod. comparison	35	ΔSOC	C g m^{-2} years^{-1}	Romanya [188]
		Willow	Clay loam	PL	CENTURY	C sequestration	2	ΔSOC	Mg C ha^{-1} year^{-1}	Borzecka [186]
		Willow	Clay sandy loam	UK	CENTURY	C sequestration	3 and 24	ΔSOC	Mg C ha^{-1} year^{-1}	Grogan [189]
Regional GIS	HCG	*Miscanthus*	Soil map	England and Wales	RothC	C sequestration	3	TOC	Mg C ha^{-1} year^{-1}	Hillier [63]
	SRWC	Willow, poplar	Soil map		RothC	C sequestration	3	TOC	Mg C ha^{-1} year^{-1}	[63]
		Willow, Poplar	Soil map	Scotland	DAYCENT	GHG emission	6–30	ΔSOC	Mg C ha^{-1} year^{-1}	Shibu [162]
		Forest	Soil map	H	Both Mod	Mod. comparison	Eq	ΔSOC	Mg C ha^{-1}	Fallon [119]

DK Denmark; *CH* Switzerland; *IRL* Ireland; *USA* United States of America; *PL* Poland; *NL* The Netherlands; *D* Germany; *UK* United Kingdom; *E* Spain; *AUS* Australia; *H* Hungary

introducing (i) water and heat flux models [14, 169] or (ii) empirical coefficients [40] or (iii) including C transport, i.e. diffusion and adsorption of DOC [42].

C Pools and Their Decomposition Rate

Most process-based SOC models assume that degradation of residues from biomass can be represented by a continuum or two or more pools of plant or soil OM of similar chemical composition and degradability [161]. Each pool decomposes following first-order kinetics integrated with temperature and moisture functions. The use of more pools implies more calibrating parameters which improve long-term prediction [8] but

increase the uncertainty of the model. Multi-pool or compartment models present challenges when applied to ligno-cellulosic (LC) plants. First, chemical or physical identification of specific pools with defined decomposition rates is very difficult [141, 163]. Second, it has never been proven experimentally that a multi-pool chain model can represent the decreasing decomposition of residues due to chemical and physical protection [118, 148, 170]. Finally, non-linear interactions between decomposition rate and temperature can exist for different pools, with the effect being greatest at lower temperatures and for 'slowly decomposable' compounds but least for 'old' SOC [14]. Further, C turnover derived from laboratory incubation of soil and residues overestimate those in undisturbed, natural systems [168, 171, 172]).

Simulations were greatly improved when decomposition rates were calibrated using labelled C measurements [173–175] and applied to the experimental evidence presented for poplar litter [78, 109]. Decomposition rates of LC residues can vary with temperature, or through interaction between temperature and 'recalcitrance', which could depend on their chemical composition [4, 109]. Recalcitrance can also be modelled using a delay time of decomposition in dependence of accessible soil pore space, which simulate the physical protective effect of soil from microbial activity [176]. Other research differentiated inter- and intra-aggregate and particulate OM to derive pools fitting the different model concepts [45]. In perennials, however, biological protection (longevity) of coarse roots, stool and rhizomes (see above) is a major conceptual challenge that has not been met.

Modelling Using Disaggregated C Inputs

So far, modelling SOC under biofuel crops has mainly relied on C inputs approximated from crop yields [19, 45, 86, 139]. A generalised link between yields and residues is plausible [63]. However, the assumption of another 10 % of roots to be accounted for as residue input [135] is not supported by any evidence. It is clear from the experimental evidence synthesised above (Tables 1 and 3 and Fig. 4) that the assumption of a single input source of residues may not be adequate to describe the C turnover and retention in the PEC soil system. Considering a single input source does not account for the surface accumulation of the litter and the longevity of the BGB. Therefore, we expanded RothC [159] in order to receive disaggregated C input fractions (litter, rhizomes, roots and root exudates) to test the importance of this new assumption and to explore the effect of different residue turnover rates (Table 2) on the change of SOC.

Based on model calibration using a 14-year-old *Miscanthus* experiment at Rothamsted (UK), we analysed the sensitivity of the modelled SOC to the input and turnover rates of segregated residues (Fig. 5). The optimised parameter set assumed yearly C gross inputs for litter, root and rhizome of 1.8, 2.2 and 2.3 Mg ha^{-1}, respectively, and an MRT for litter/root and rhizomes of 2.3 and 8 years, respectively. The sensitivity analysis showed that for each 1 % change of gross residue C input, the simulated SOC changed by 0.13, 0.11 and 0.1 % for root, litter and rhizome, respectively (Fig. 5a). For a 1 % change of respective residue MRT, the simulated SOC changed by 0.02, 0.01 and 0.05 % for root, litter and rhizome, respectively (Fig. 5b).

Figure 5a clearly implies that not only yield but harvest management (e.g. removal of residues in *Miscanthus*) would affect SOC through the input rate of surface litter which is an important input of ecosystem models [151, 168] and LCA for energy crops [177]. There are very few data for *Miscanthus* [62, 65, 150], switchgrass [178, 179], willow [143, 180] or poplar [22] available. None of these would allow deriving a dose–response function.

Using reviewed parameters for residue deposition and decomposition under SRC willow (Tables 1 and 2), we simulated a very similar ΔSOC as under *Miscanthus*. However, the simulated cumulative CO_2 emission was 20 % higher than under *Miscanthus*, because the turnover of fine willow roots is 17 times faster than *Miscanthus* roots.

Further Development Needs

What are the key points that can be derived from the review of the evidence base and its current status for modelling C sequestration (ΔSOC) under PECs?

There Is No Steady State!

The evidence presented above clearly shows that there is no justification for the assumption of a steady-state sequestration rate as stated by others [8, 136]. The opposite is evident: initial sequestration rates are low due to small inputs during the establishing phase [25, 43, 46, 60, 86, 106, 134], which could depend on the productivity and fertility of the soil, its former land use and planting material/success. Although not significant, initial SOC gains are greater on former arable than on former grassland when planted with *Miscanthus* in spite of greater—probably yield-dependent—sequestration rates on grassland [134]. A long resilience phase for SOC was seen under *Miscanthus* established on former grassland [86], and no change of SOC was measured under 19-year-old willow SRC [143]. A key to this phenomenon could be the high losses of SOC under grass which are likely to be related to higher microbial biomass in the soil of former grassland [181]. Our scenario analysis of a compartmentalised residue input and turnover model shows that differentiation in turnover rates can impact on the dynamics. The modelling of SOC fractionation seems essential for the persistence and sustainability of C sequestration, showing that initially new SOC is part of easily decomposable POM [139] and later part of the inter-aggregate fraction [25, 45]. This would need a series of new experiments to see the decrease of each fraction after returning to arable.

Interaction Between Crop Growth and Soil

Root dry matter in the surface horizon is an order of magnitude larger than in the subsoil [31, 32, 80, 84]. This affects the water extraction dynamics and the root exudates. To simulate hydrological effects on the SOC decomposition phenomena, numerous modelling approaches exist. The effect of soil moisture is complex and non-linear; a single moisture threshold for decreased microbial activity and soil respiration does not exist [14]. Usually, process-based SOC models [41, 182] have decomposition reduction functions based on soil moisture, derived from the soil water potential or soil water content. Bauer et al. [41] compared various decomposition routines in a unique modified version of RothC and showed 2 % sensitivity of CO_2 emissions due to soil
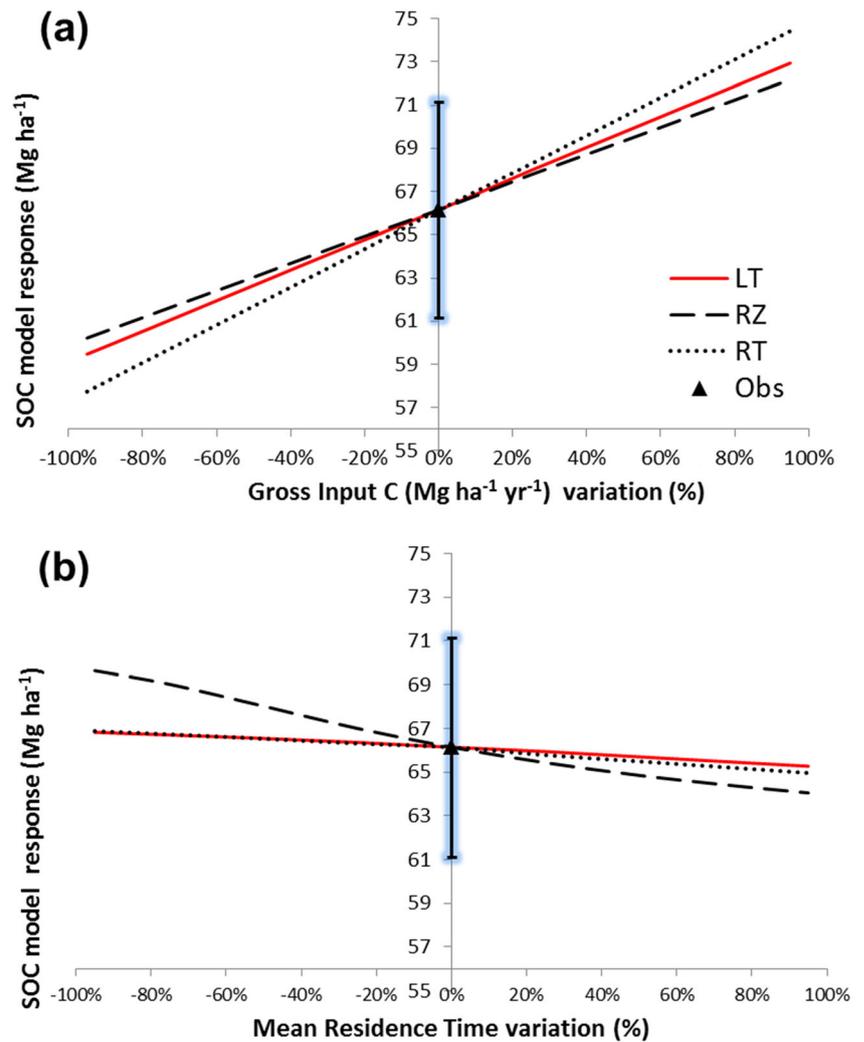
moisture variation. The uncertainty of soil moisture derived from pedotransfer functions can affect C flux simulations [183]. Bulk density in particular is a dynamic property when changing from arable to perennial agricultural systems [24], and its variation increases the uncertainty of stock changes [80, 164, 184] and needs to be accounted for to capture the effect of spatial heterogeneity on SOC dynamics and sequestration.

Scaling Up—Is There Enough Knowledge?

Empirical coefficients, such as those used in the IPCC Tier 1 method [39], are insufficient to account for SOC sequestration under PECs. Spatial variation accounted for by GIS-based simulation provides yield-dependent potential C inputs and sequestration in soils [63]. Their assumptions that the soil-crop system was at potential productivity, and equilibrium could be wrong where net SOC formation is delayed [86], cover is patchy [46, 139] or residues vary in their decomposability [108]. There is no doubt that, technically, various plant ecosystems and a SOC decomposition model of desired

complexity can be integrated with some empirical method for assessing site/soil-specific SOC stock changes, as shown for the GEFSOC system [39]. Currently, the experimental evidence for the effect of soil texture and baseline SOC on sequestration potential [63] is so small that a general relationship for its spatial extrapolation is not credible. Data for ΔSOC from marginal soils with a low baseline SOC, such as the site in Ireland [135], are target areas but are not available to validate a generalised model. Indeed, topography and soil distribution maps can be included as inputs to process-based models to simulate the impacts on growth and SOC for PECs.

An alternative up-scaling technique is to run C process models (i.e. RothC) at small scales to initialise larger spatial regression models [185]. However, the limitations of process-based models become more apparent when simulating at larger scales with regard to their limits to consider the effect of soil depth [168], adequate temporal resolution [171] and surface litter decomposition [173, 174]. It is difficult to obtain sufficiently detailed and widespread data to generalise and verify a generic process description.

Concluding Remarks

Is the jury still out? What are the certainties and uncertainties with regard to the assessment of carbon sequestration under PECs? We draw attention to four major points when considering this:

First, our review shows annual ΔSOC of HCG and SRWC exceeding the minimum sequestration rate, and more certainty applies to C4-HCG than C3-SRWC due to isotope (^{13}C) use. Based on the C inputs retained, ΔSOC (Fig. 4), one can distinguish retentive HCG (30–55 %) from transformative SRWC (19–33 %) systems.

Second, underpinning evidence for the C balance components is more certain for AGB than BGB inputs: (a) litter fall and harvest residues are more easily measured than roots and rhizomes/stool and (b) gross inputs of BGB components profoundly depend on their MRT. Turnover is derived from time series estimating productivity/mortality over standing biomass, but time series are the exception over single 'snapshot' observations.

Third, severe restrictions apply to the validity of current *in situ* residue turnover rates due to technical limitations (litter bags) and steady-state assumptions. Laborious alternative methods have filled knowledge gaps but described dynamics in young plantations, which may not be extrapolated to mature stands. C sequestration implies a long-term view and analysis on SOC fractions that represent a turnover time in excess of 25 to 50 years.

Fourth, effects from the environment, fertility and management, e.g. harvest intensity of the systems, are not covered in dose–response experiments, and evidence for target ecosystems, like marginal land, is underrepresented.

In terms of modelling, reliable projections can only be achieved if simulations capture short- and long-term dynamics, synthesising data from chrono-sequences, disaggregating inputs and understanding C fluxes within SOC fractions. Experimental evidence for the SOC dynamics, size and persistence of the SOC pools prior to establishment, during the lifetime and after termination of the PECs must underpin modelling.

Glossary

AGB	Aboveground biomass
BGB	Belowground biomass
Component	Pool or residue fraction of A/BGB within a systems model
Decomposition/ turnover rate (k)	Portion of [stored or added] C lost from a given pool (time, e.g. year^{-1})
DOC	Dissolved organic C
FACE	Free air C enrichment
GHG	Greenhouse gas
MRT	Mean residence time, average storage time [years] for a given residue or C pool; $1/k$ assuming zero order
Half-life	Time in which 50% of the C component is degraded; assuming first-order decay: $t_{1/2}=\ln2/k$
Soil C retention or storage	C retained without specifying a defined time span
Soil C sequestration	C 'sequestered' when stored in soil for at least 25 years
HCG	Herbaceous crops and grasses
PEC	Perennial energy crops
SRC	Short-rotation coppice
SRF	Short-rotation forestry
SRWC	Short-rotation woody crops (SRC and SRF)
LCA	Life cycle assessment
(i)LUC	(Indirect) land use change

References

1. Lemus R, Lal R (2005) Bioenergy crops and carbon sequestration. Crit Rev Plant Sci 24(1):1–21. doi:10.1080/07352680590910393
2. Lewandowski I, Scurlock JMO, Lindvall E, Christou M (2003) The development and current status of perennial rhizomatous grasses as energy crops in the US and Europe. Biomass Bioenergy 25(4):335–361
3. Lovett AA, Sünnenberg GM, Richter GM, Dailey AG, Riche AB, Karp A (2009) Biomass production and land use trade-offs revealed by GIS constraint and yield mapping of Miscanthus in England. Bio Energy Res 2(1):17–29. doi:10.1007/s12155-008-9030-x
4. Johnson JMF, Barbour NW, Weyers SL (2007) Chemical composition of crop biomass impacts its decomposition. Soil Sci Soc Am J 71(1):155–162
5. Sartori F, Lal R, Ebinger MH, Eaton JA (2007) Changes in soil carbon and nutrient pools along a chronosequence of poplar plantations in the Columbia Plateau, Oregon, USA. Agric Ecosyst Environ 122(3):325–339. doi:10.1016/j.agee.2007.01.026
6. Lewandowski I, Heinz A (2003) Delayed harvest of Miscanthus—influences on biomass quantity and quality and environmental impacts of energy production. Eur J Agron 19(1):45–63
7. Volk TA, Verwijst T, Tharakan PJ, Abrahamson LP, White EH (2004) Growing fuel a sustainability assessment of willow biomass crops. Front Ecol Environ 2(8):411–418

8. Garten CT (2012) Review and model-based analysis of factors influencing soil carbon sequestration beneath switchgrass (Panicum virgatum). Bioenergy Res 5(1):124–138. doi:10.1007/s12155-011-9154-2

9. Bransby DI, McLaughlin SB, Parrish DJ (1998) A review of carbon and nitrogen balances in switchgrass grown for energy. Biomass Bioenergy 14(4):379–384

10. Garten CT, Wullschleger SD, Classen AT (2011) Review and model-based analysis of factors influencing soil carbon sequestration under hybrid poplar. Biomass Bioenergy 35(1):214–226. doi:10.1016/j.biombioe.2010.08.013

11. Poeplau C, Don A (2014) Soil carbon changes under Miscanthus driven by C4 accumulation and C3 decomposition—toward a default sequestration function. GCB Bioenergy 6(4):327–338. doi:10.1111/gcbb.12043

12. Don A, Osborne B, Hastings A (2011) Land-use change to bioenergy production in Europe: implications for the greenhouse gas balance and soil carbon. Glob Chang Biol Bionergy 4:372–391

13. Heaton E, Dohleman FG, Miguez AF (2010) Miscanthus: a promising biomass crop. Adv Bot Res 56:75–137

14. Battle-Aguilar J, Brovelli A, Porporato A, Barry DA (2010) Modelling soil carbon and nitrogen cycles during land use change. A review. Agron Sustain Dev 31(2):251–274. doi:10.1051/agro/2010007

15. Thomas ARC, Bond AJ, Hiscock KM (2013) A multi-criteria based review of models that predict environmental impacts of land use-change for perennial energy crops on water, carbon and nitrogen cycling. GCB Bioenergy 5(3):227–242. doi:10.1111/j.1757-1707.2012.01198.x

16. Sartori F, Lal R, Ebinger MH, Parrish DJ (2006) Potential soil carbon sequestration and CO2 offset by dedicated energy crops in the USA. Crit Rev Plant Sci 25(25):441–472

17. Liebig MA, Schmer MR, Vogel KP, Mitchell RB (2008) Soil carbon storage by switchgrass grown for bioenergy. Bioenergy Res 1(3–4):215–222. doi:10.1007/s12155-008-9019-5

18. Adler PR, Del Grosso SJ, Parton WJ (2007) Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems. Ecol Appl 17(3):675–691

19. Clifton-Brown JC, Breuer J, Jones MB (2007) Carbon mitigation by the energy crop, Miscanthus. Glob Chang Biol 13(11):2296–2307

20. Larsen SU, Jorgensen U, Kjeldsen JB, Laerke PE (2014) Long-term Miscanthus yields influenced by location, genotype, row distance, fertilization and harvest season. Bioenergy Res 7(2):620–635. doi:10.1007/s12155-013-9389-1

21. Meehan PG, McDonnell KP, Finnan JM (2013) An assessment of the effect of harvest time and harvest method on biomass loss for Miscanthus × giganteus. Glob Chang Biol Bioenergy 5(4):400–407. doi:10.1111/j.1757-1707.2012.01205.x

22. Berhongaray G, El Kasmioui O, Ceulemans R (2013) Comparative analysis of harvesting machines on an operational high-density short rotation woody crop (SRWC) culture: one-process versus two-process harvest operation. Biomass Bioenergy 58:333–342. doi:10.1016/j.biombioe.2013.07.003

23. Lewandowski I, Schmidt U (2006) Nitrogen, energy and land use efficiencies of Miscanthus, reed canary grass and triticale as determined by the boundary line approach. Agric Ecosyst Environ 112(4):335–346

24. Zan CS, Fyles JW, Girouard P, Samson RA (2001) Carbon sequestration in perennial bioenergy, annual corn and uncultivated systems in southern Quebec. Agric Ecosyst Environ 86(2):135–144

25. Garten CT, Wullschleger SD (2000) Soil carbon dynamics beneath switchgrass as indicated by stable isotope analysis. J Environ Qual 29(2):645–653. doi:10.2134/jeq2000.00472425002900020036x

26. Rytter RM (2012) The potential of willow and poplar plantations as carbon sinks in Sweden. Biomass Bioenergy 36:86–95

27. Kuzovkina YA, Volk TA (2009) The characterization of willow (Salix L.) varieties for use in ecological engineering applications: co-ordination of structure, function and autecology. Ecol Eng 35(8):1178–1189. doi:10.1016/j.ecoleng.2009.03.010

28. Lasch P, Kollas C, Rock J, Suckow F (2009) Potentials and impacts of short-rotation coppice plantation with aspen in Eastern Germany under conditions of climate change. Reg Environ Chang 10(2):83–94. doi:10.1007/s10113-009-0095-7

29. Kollas CP, Lasch JR, Suckow F (2009) Bioenergy potential in Germany—assessing spatial patterns of biomass production. Int Agrophysics 23:343–352

30. Block RMA, Rees KCJ, Knight JD (2006) A review of fine root dynamics in Populus plantations. Agrofor Syst 67(1):73–84. doi:10.1007/s10457-005-2002-7

31. Gill RA, Burke IC, Lauenroth WK, Milchunas DG (2002) Longevity and turnover of roots in the shortgrass steppe: influence of diameter and depth. Plant Ecol 159(2):241–251. doi:10.1023/A:1015529507670

32. Rytter RM (1999) Fine-root production and turnover in a willow plantation estimated by different calculation methods. Scand J For Res 14(6):526–537

33. Lettens S, Muys B, Ceulemans R, Moons E, Garcia J, Coppin P (2003) Energy budget and greenhouse gas balance evaluation of sustainable coppice systems for electricity production. Biomass and Bioenergy 24 (3):179–197. doi:Pii S0961-9534(02)00104-6

34. Dungait J, Hopkins D, Gregory A, Whitmore A (2012) Soil organic matter turnover is governed by accessibility not recalcitrance. Glob Chang Biol 18:1781–1796

35. Marschner B, Brodowski S, Dreves A, Gleixner G, Gude A, Grootes P (2008) How relevant is recalcitrance for the stabilization of organic matter in soils? J Plant Nutr Soil Sci 171:91–110

36. Lal R (2003) Offsetting global CO2 emissions by restoration of degraded soils and intensification of world agriculture and forestry. Land Degrad Dev 14(3):309–322

37. Lal R (2008) Carbon sequestration. Philos Trans R Soc Lond Ser B Biol Sci 363(1492):815–830. doi:10.1098/rstb.2007.2185

38. Bauer J, Kirschbaum MUF, Weihermuller L, Huisman JA, Herbst M, Vereecken H (2008) Temperature response of wheat decomposition is more complex than the common approaches of most multi-pool models. Soil Biol Biochem 40(11):2780–2786. doi:10.1016/j.soilbio.2008.07.024

39. Easter M, Paustian K, Killian K, Williams S, Feng T, Al-Adamat R, Batjes NH, Bernoux M, Bhattacharyya T, Cerri CC, Cerri CEP, Coleman K, Falloon P, Feller C, Gicheru P, Kamoni P, Milne E, Pal DK, Powlson DS, Rawajfih Z, Sessay M, Wokabi S (2007) The GEFSOC soil carbon modelling system: a tool for conducting regional-scale soil carbon inventories and assessing the impacts of land use change on soil carbon. Agric Ecosyst Environ 122(1):13–25. doi:10.1016/j.agee.2007.01.004

40. Jenkinson DS, Coleman K (2008) The turnover of organic carbon in subsoils. Part 2. Modelling carbon turnover. Eur J Soil Sci 59(2):400–413. doi:10.1111/j.1365-2389.2008.01026.x

41. Bauer J, Herbst M, Huisman JA, Weihermuller L, Vereecken H (2008) Sensitivity of simulated soil heterotrophic respiration to temperature and moisture reduction functions. Geoderma 145(1–2):17–27. doi:10.1016/j.geoderma.2008.01.026

42. Braakhekke MC, Christian Beer M, Hoosbeek MR, Reichstein M, Kruijt B, Schrumpf M, Kabat P (2011) SOMPROF: a vertically explicit soil organic matter model. Ecol Model 222:1712–1730

43. Ma Z, Wood CW, Bransby DI (2000) Carbon dynamics subsequent to establishment of switchgrass. Biomass Bioenergy 18(2):93–104

44. Andrén O, Kätterer T (2001) Basic principles for soil carbon sequestration and calculating dynamic country-level balances including future scenarios. Assessment methods for soil carbon. Edited by R. Lal, J.M. Kimble, R.F. Follett, and B. A. Stewart, Lewis Publishers:495–511

45. Dondini M, Van Groenigen KJ, Del Galdo I, Jones MB (2009) Carbon sequestration under Miscanthus: a study of 13C distribution in soil aggregates. Glob Chang Biol Bioenergy 1(5):321–330. doi:10.1111/j.1757-1707.2009.01025.x

46. Zimmermann J, Styles D, Hastings A, Dauber J, Jones MB (2014) Assessing the impact of within crop heterogeneity ('patchiness') in young Miscanthus × giganteus fields on economic feasibility and soil carbon sequestration. GCB Bioenergy 6(5):566–576. doi:10.1111/gcbb.12084

47. Collins HP, Smith JL, Fransen S, Alva AK, Kruger CE, Granatstein DM (2010) Carbon sequestration under irrigated switchgrass (Panicum virgatum L.) production. Soil Sci Soc Am J 74(6): 2049–2058. doi:10.2136/sssaj2010.0020

48. Drewer J, Finch J, Lloyd R, Baggs E, Skiba U (2011) How do soil emission of N20, CH4 and CO2 from perennial bioenergy crops differ from arable annual crops? GCB Bioenergy. doi:10.1111/j.1757-1707.2011.01136

49. Gauder M, Butterbach-Bahl K, Graeff-Hoenninger S, Claupein W, Wiegel R (2012) Soil-derived trace gas fluxes from different energy crops-results from a field experiment in Southwest Germany. Glob Chang Biol Bioenergy 4(3):289–301. doi:10.1111/j.1757-1707.2011.01135.x

50. Jorgensen RN, Jorgensen BJ, Nielsen NE, Maag M, Lind AM (1997) N2O emission from energy crop fields of Miscanthus "Giganteus" and winter rye. Atmos Environ 31(18):2899–2904

51. Toma Y, Fernandez FG, Sato S, Izumi M, Hatano R, Yamada T, Nishiwaki A, Bollero G, Stewart JR (2011) Carbon budget and methane and nitrous oxide emissions over the growing season in a Miscanthus sinensis grassland in Tomakomai, Hokkaido, Japan. Glob Chang Biol Bioenergy 3(2):116–134. doi:10.1111/j.1757-1707.2010.01070.x

52. Nikiema P, Rothstein DE, Min D-H, Kapp CJ (2011) Nitrogen fertilization of switchgrass increases biomass yield and improves net greenhouse gas balance in northern Michigan, USA. Biomass Bioenergy 35(10):4356–4367. doi:10.1016/j.biombioe.2011.08.006

53. Schmer MR, Liebig MA, Hendrickson JR, Tanaka DL, Phillips RL (2012) Growing season greenhouse gas flux from switchgrass in the northern Great Plains. Biomass Bioenergy 45:315–319. doi:10.1016/j.biombioe.2012.05.026

54. Hellebrand HJ, Straehle M, Scholz V, Kern J (2010) Soil carbon, soil nitrate, and soil emissions of nitrous oxide during cultivation of energy crops. Nutr Cycl Agroecosyst 87(2):175–186. doi:10.1007/s10705-009-9326-z

55. Nikiema P, Rothstein DE, Miller RO (2012) Initial greenhouse gas emissions and nitrogen leaching losses associated with converting pastureland to short-rotation woody bioenergy crops in northern Michigan, USA. Biomass Bioenergy 39:413–426. doi:10.1016/j.biombioe.2012.01.037

56. Palmer MM, Forrester JA, Rothstein DE, Mladenoff DJ (2014) Establishment phase greenhouse gas emissions in short rotation woody biomass plantations in the Northern Lake States, USA. Biomass Bioenergy 62:26–36. doi:10.1016/j.biombioe.2014.01.021

57. Anderson-Teixeira KJ, Masters MD, Black CK, Zeri M, Hussain MZ, Bernacchi CJ, DeLucia EH (2013) Altered belowground carbon cycling following land-use change to perennial bioenergy crops. Ecosystems 16(3):508–520. doi:10.1007/s10021-012-9628-x

58. Rytter RM (2001) Biomass production and allocation, including fine-root turnover, and annual N uptake in lysimeter-grown basket willows. For Ecol Manag 140:177–192

59. Davis SC, Parton WJ, Dohleman FG, Smith CM, Grosso SD, Kent AD, DeLucia EH (2009) Comparative biogeochemical cycles of bioenergy crops reveal nitrogen-fixation and low greenhouse gas emissions in a Miscanthus × giganteus agro-ecosystem. Ecosystems 13(1):144–156. doi:10.1007/s10021-009-9306-9

60. Arevalo CBM, Bhatti JS, Chang SX, Sidders D (2011) Land use change effects on ecosystem carbon balance: from agricultural to hybrid poplar plantation. Agric Ecosyst Environ 141(3–4):342–349. doi:10.1016/j.agee.2011.03.013

61. Monti A, Zatta A (2009) Root distribution and soil moisture retrieval in perennial and annual energy crops in Northern Italy. Agric Ecosyst Environ 132(3–4):252–259. doi:10.1016/j.agee.2009.04.007

62. Beuch S, Boelcke B, Belau L (2000) Effect of the organic residues of Miscanthus x giganteus on the soil organic matter level of arable soils. J Agron Crop Sci-Z-Acker Pflanzenbau 184(2):111–119

63. Hillier J, Dailey G, Aylott M, Whittaker C, Richter G, Riche A, Murphy R, Taylor G, Smith P (2009) Greenhouse gas emissions from four bioenergy crops in England and Wales: integrating spatial estimates of yield and soil carbon balance in life cycle analyses. Glob Chang Biol Bioenergy 1(4):267–281. doi:10.1111/j.1757-1707.2009.01021.x

64. Amougou N, Bertrand I, Machet JM, Recous S (2011) Quality and decomposition in soil of rhizome, root and senescent leaf from Miscanthus x giganteus, as affected by harvest date and N fertilization. Plant Soil 338(1–2):83–97. doi:10.1007/s11104-010-0443-x

65. Christian DG, Poulton PR, Riche AB, Yates NE, Todd AD (2006) The recovery over several seasons of N-15-labelled fertilizer applied to Miscanthus x giganteus ranging from 1 to 3 years old. Biomass Bioenergy 30(2):125–133

66. Dohleman FG, Heaton EA, Arundale RA, Long SP (2012) Seasonal dynamics of above- and below-ground biomass and nitrogen partitioning in Miscanthus x giganteus and Panicum virgatum across three growing seasons. Glob Chang Biol Bionergy 4(5): 534–544

67. Frank AB, Berdahl JD, Hanson JD, Liebig MA, Johnson HA (2004) Biomass and carbon partitioning in switchgrass. Crop Sci 44(4): 1391–1396

68. Hansen EM (1993) Soil carbon sequestration beneath hybrid poplar plantations in the North Central United States. Biomass Bioenergy 5(6):431–436

69. Amougou N, Bertrand I, Cadoux S, Recous S (2012) Miscanthus × giganteus leaf senescence, decomposition and C and N inputs to soil. Glob Chang Biol Bionergy 4(6):698–707. doi:10.1111/j.1757-1707.2012.01192.x

70. Liebig MA, Johnson HA, Hanson JD, Frank AB (2005) Soil carbon under switchgrass stands and cultivated cropland. Biomass Bioenergy 28:347–354

71. Garten CT Jr, Smith JL, Tyler DD, Amonette JE, Bailey VL, Brice DJ, Castro HF, Graham RL, Gunderson CA, Izaurralde RC, Jardine PM, Jastrow JD, Kerley MK, Matamala R, Mayes MA, Metting FB, Miller RM, Moran KK, Post WM III, Sands RD, Schadt CW, Phillips JR, Thomson AM, Vugteveen T, West TO, Wullschleger SD (2010) Intra-annual changes in biomass, carbon, and nitrogen dynamics at 4-year old switchgrass field trials in west Tennessee, USA. Agric Ecosyst Environ 136(1/2):177–184. doi:10.1016/j.agee.2009.12.019

72. Tufekcioglu A, Raich JW, Isenhart TM, Schultz RC (2003) Biomass, carbon and nitrogen dynamics of multi-species riparian buffers within an agricultural watershed in Iowa, USA. Agrofor Syst 57(3):187–198. doi:10.1023/a:1024898615284

73. Al-Kaisi MM, Grote JB (2007) Cropping systems effects on improving soil carbon stocks of exposed subsoil. Soil Sci Soc Am J 71(4):1381–1388. doi:10.2136/sssaj2006.0200

74. Hartman JC, Nippert JB, Orozco RA, Springer CJ (2011) Potential ecological impacts of switchgrass (Panicum virgatum L.) biofuel

cultivation in the Central Great Plains, USA. Biomass Bioenergy 35(8):3415–3421. doi:10.1016/j.biombioe.2011.04.055

75. Lee DK, Owens VN, Doolittle JJ (2007) Switchgrass and soil carbon sequestration response to ammonium nitrate, manure, and harvest frequency on conservation reserve program land. Agron J 99(2):462–468. doi:10.2134/agronj2006.0152

76. Zenone T, Gelfand I, Chen J, Hamilton SK, Robertson GP (2013) From set-aside grassland to annual and perennial cellulosic biofuel crops: effects of land use change on carbon balance. Agric For Meteorol 182:1–12. doi:10.1016/j.agrformet.2013.07.015

77. Wienhold BJ, Varvel GE, Johnson JMF, Wilhelm WW (2013) Carbon source quality and placement effects on soil organic carbon status. Bioenergy Res 6(2):786–796. doi:10.1007/s12155-013-9301-z

78. Cotrufo MF, De Angelis P, Polle A (2005) Leaf litter production and decomposition in a poplar short-rotation coppice exposed to free air CO2 enrichment (POPFACE). Glob Chang Biol 11:971–982

79. Fang S, Xue J, Tang L (2007) Biomass production and carbon sequestration potential in poplar plantations with different management patterns. J Environ Manag 85(3):672–679. doi:10.1016/j.jenvman.2006.09.014

80. Richter GM, Agostini F, Redmile-Gordon M, White R, Goulding KWT (2015) Sequestration of C in soils under Miscanthus can be marginal and is affected by genotype-specific root distribution. Agric Ecosyst Environ 200(1):169–177. doi:10.1016/j.agee.2014.11.011

81. Berhongaray G, Janssens IA, King JS, Ceulemans R (2013) Fine root biomass and turnover of two fast-growing poplar genotypes in a short-rotation coppice culture. Plant Soil 373(1–2):269–283. doi:10.1007/s11104-013-1778-x

82. Rytter RM (2013) The effect of limited availability of N or water on C allocation to fine roots and annual fine root turnover in Alnus incana and Salix viminalis. Tree Physiol 33(9):924–939. doi:10.1093/treephys/tpt060

83. Rytter RM, Rytter L (1998) Growth, decay, and turnover rates of fine roots of basket willows. Can J For Res-Revue Can De Rech Forestiere 28(6):893–902

84. Neukirchen D, Himken M, Lammel J, Czypionka-Krause U, Olfs H-W (1999) Spatial and temporal distribution of the root system and root nutrient content of an established Miscanthus crop. Eur J Agron 11:301–309

85. Himken M, Lammel J, Neukirchen D, Czypionka-Krause U, Olfs H-W (1997) Cultivation of Miscanthus under West European conditions: seasonal. Plant and Soil 189 (117–126)

86. Zatta A, Clifton-Brown J, Robson P, Hastings A, Monti A (2014) Land use change from C3 grassland to C4 Miscanthus: effects on soil carbon content and estimated mitigation benefit after six years. GCB Bioenergy 6(4):360–370. doi:10.1111/gcbb.12054

87. Whittaker RH, Marks PL (1975) Methods of assessing terrestrial productivity. In: Primary productivity of the biosphere, ed H Lieth and R Whittaker:55–118

88. Rytter R, Hansson A (1996) Seasonal amount, growth and depth distribution of fine roots in an irrigated and fertilized Salix viminalis L. plantation. Biomass Bioenergy 11:129–137

89. Zhang J, Shangguan T, Meng Z (2011) Changes in soil carbon flux and carbon stock over a rotation of poplar plantations in northwest China. Ecol Res 26(1):153–161. doi:10.1007/s11284-010-0772-5

90. Calfapietra C, Gielen B, Galema ANJ, Lukac M, De Angelis P, Moscatelli MC, Ceulemans R, Scarascia-Mugnozza G (2003) Free-air CO2 enrichment (FACE) enhances biomass production in a short-rotation poplar plantation. Tree Physiol 23(12):805–814

91. Tufekcioglu A, Raich JW, Isenhart TM, Schultz RC (1998) Fine root dynamics, coarse root biomass, root distribution, and soil respiration in a multispecies riparian buffer in Central Iowa, USA. Agrofor Syst 44(2–3):163–174. doi:10.1023/a:1006221921806

92. Upson MA, Burgess PJ (2013) Soil organic carbon and root distribution in a temperate arable agroforestry system. Plant Soil 373(1–2):43–58. doi:10.1007/s11104-013-1733-x

93. Yan M, Zhang X, Zhou G, Gong J, You X (2011) Temporal and spatial variation in soil respiration of poplar plantations at different developmental stages in Xinjiang, China. J Arid Environ 75(1):51–57. doi:10.1016/j.jaridenv.2010.09.005

94. Al Afas N, Marron N, Zavalloni C, Ceulemans R (2008) Growth and production of a short-rotation coppice culture of poplar-IV: fine root characteristics of five poplar clones. Biomass Bioenergy 32(6):494–502. doi:10.1016/j.biombioe.2007.11.007

95. Grigal DF, Berguson WE (1998) Soil carbon changes associated with short-rotation systems. Biomass Bioenergy 14(4):371–377

96. Hromadko L, Vranova V, Techer D, Laval-Gilly P (2010) Composition of root exudates of Miscanthus × giganteus Greef Et Deu. Acta Universitatis Agriculturae Et Silviculturae Mendelianae Brunensis LVIII 1:71–76

97. Kaňova H, Carre J, Vranova V, Rejšek K (2010) Organic compounds in root exudates of Miscanthus × giganteus Greef Et Deu and limitation of microorganisms in its rhizosphere by nutrients. Acta Univ Agric Et Silviculturae Mendelianae Brunensis LVIII 5:203–210

98. De Neergaard A, Porter JR, Gorissen A (2002) Distribution of assimilated carbon in plants and rhizosphere soil of basket willow (Salix viminalis L.). Plant Soil 245(2):307–314

99. Gielen B, Calfapietra C, Lukac M, Wittig VE, De Angelis P, Janssens IA, Moscatelli MC, Grego S, Cotrufo MF, Godbold DL, Hoosbeek MR, Long SP, Miglietta F, Polle A, Bernacchi CJ, Davey PA, Ceulemans R, Scarascia-Mugnozza GE (2005) Net carbon storage in a poplar plantation (POPFACE) after three years of free-air CO2 enrichment. Tree Physiol 25(11):1399–1408

100. Rasse DP, Rumpel C, Digna MF (2005) Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. Plant Soil 269:341–356

101. Kell DB (2011) Breeding crop plants with deep roots: their role in sustainable carbon, nutrient and water sequestration. Ann Bot 108:407–418

102. Kuzyakov Y (2002) Review: factors affecting rhizosphere priming effects. J Plant Nutr Soil Sci 165:382–396

103. Cheng W (2009) Rhizosphere priming effect: its functional relationships with microbial turnover, evapotranspiration, and C-N budgets. Soil Biol Biochem 41:1795–1801

104. Gill RA, Jackson RB (2000) Global patterns of root turnover for terrestrial ecosystems. New Phytol 147:13–31

105. Masuzawa T, Hogetsu K (1977) Seasonal changes in amount of carbohydrate and crude protein in rhizome of Miscanthus sacchariflorus. Bot Mag-Tokyo 90(1019):181–191

106. Hansen EM, Christensen BT, Jensen LS, Kristensen K (2004) Carbon sequestration in soil beneath long-term Miscanthus plantations as determined by 13C abundance. Biomass Bioenergy 26(2):97–105. doi:10.1016/s0961-9534(03)00102-8

107. Moorhead DL, Currie WS, Rastetter BW, Parton J, Harmon ME (1999) Climate and litter quality controls on decomposition: an analysis of modeling approaches. Glob Biogechemical Cycles 13(2):575–589

108. Magid J, Luxhoi J, Lyshede OB (2004) Decomposition of plant residues at low temperatures separates turnover of nitrogen and energy rich tissue components in time. Plant Soil 258(1):351–365

109. Rubino M, Dungait JAJ, Evershed RP, Bertolini T, De Angelis P, D'Onofrio A, Lagomarsino A, Lubritto C, Merola A, Terrasi F (2010) Carbon input belowground is the major C flux contributing to leaf litter mass loss: evidences from a 13C labelled-leaf litter experiment. Soil Biol Biochem 42(7):1009–1016. doi:10.1016/j.soilbio.2010.02.018

110. Ernst G, Henseler I, Felten D, Emmerling C (2009) Decomposition and mineralization of energy crop residues governed by

earthworms. Soil Biol Biochem 41(7):1548–1554. doi:10.1016/j.soilbio.2009.04.015

111. Slapokas T, Granhall U (1991) Decomposition of willow-leaf litter in a short-rotation forest in relation to fungal colonization and palatability for earthworms. Biol Fertil Soils 10:241–248

112. Oren R, Ellsworth DS, Johnsen KH (2001) Soil fertility limits carbon sequestration by forest ecosystems in a CO_2-enriched atmosphere. Nature 411:469–471

113. Hoffmann B, Muller T, Joergensen RG (2010) Carbon dioxide production and oxygen consumption during the early decomposition of different litter types over a range of temperatures in soil-inoculated quartz sand. J Plant Nutr Soil Sci 173(2):217–223. doi:10.1002/jpln.200800187

114. Luxhoi J, Magid J, Tscherko D, Kandeler E (2002) Dynamics of invertase, xylanase and coupled quality indices of decomposing green and brown plant residues. Soil Biol Biochem 34(4):501–508

115. Eiland F, Klamer M, Lind AM, Leth M, Baath E (2001) Influence of initial C/N ratio on chemical and microbial composition during long term composting of straw. Microb Ecol 41(3):272–280

116. Foereid B, de Neergaard A, Høgh-Jensen H (2004) Turnover of organic matter in a Miscanthus field: effect of time in Miscanthus cultivation and inorganic nitrogen supply. Soil Biol Biochem 36(7):1075–1085. doi:10.1016/j.soilbio.2004.03.002

117. Frey SD, Six J, Elliott ET (2003) Reciprocal transfer of carbon and nitrogen by decomposer fungi at the soil litter interface. Soil Biol Biochem 35(1001):1004

118. Rovira P, Vallejo VR (2007) Labile, recalcitrant, and inert organic matter in Mediterranean forest soils. Soil Biol Biochem 39(1):202–215

119. Falloon P, Smith P, Szabo J, Pasztor L (2002) Comparison of approaches for estimating carbon sequestration at the regional scale. Soil Use Manag 18(3):164–174. doi:10.1079/sum2002118

120. Felten D, Emmerling C (2011) Effects of bioenergy crop cultivation on earthworm communities—a comparative study of perennial (Miscanthus) and annual crops with consideration of graded land-use intensity. Appl Soil Ecol 49:167–177. doi:10.1016/j.apsoil.2011.06.001

121. von Lützow M, Kögel-Knabner I, Ekschmitt K, Flessa H, Guggenberger G, Matzner E, Marschner B (2007) SOM fractionation methods: relevance to functional pools and to stabilization mechanisms. Soil Biol Biochem 39(9):2183–2207. doi:10.1016/j.soilbio.2007.03.007

122. Kim S, Kim JG (2009) Humulus japonicus accelerates the decomposition of Miscanthus sacchariflorus and Phragmites australis in a floodplain. J Plant Biol 52(5):466–474. doi:10.1007/s12374-009-9060-8

123. Yamane I (1974) Decomposition of litter of Miscanthus sinesins during five years under semi-natural conditions. Rep Inst Agr Res 25:25–30

124. de Graaff M-A, Six J, Jastrow JD, Schadt CW, Wullschleger SD (2013) Variation in root architecture among switchgrass cultivars impacts root decomposition rates. Soil Biol Biochem 58:198–206. doi:10.1016/j.soilbio.2012.11.015

125. Klumpp K, Soussana JF, Falcimagne R (2007) Long-term steady state C-13 labelling to investigate soil carbon turnover in grasslands. Biogeosciences 4(3):385–394

126. Nadelhoffeer KJ, Aber JD, Melillo JM (1985) Fine roots, net primary production, and soil nitrogen availability: a new hypothesis. Ecology 66:1377–1390. doi:10.2307/1939190

127. Püttsepp Ü, Lõhmus K, Koppel A (2007) Decomposition of fine roots and α-cellulose in a short rotation willow (Salix spp.) plantation on abandoned agricultural land. Silva Fennica 41(2):247–258

128. Saffih-Hdadi K, Mary B (2008) Modeling consequences of straw residues export on soil organic carbon. Soil Biol Biochem 40(3):594–607. doi:10.1016/j.soilbio.2007.08.022

129. Niklaus PA, Falloon P (2006) Estimating soil carbon sequestration under elevated CO_2 by combining carbon isotope labelling with soil carbon cycle modelling. Glob Chang Biol 12(10):1909–1921. doi:10.1111/j.1365-2486.2006.01215.x

130. Hoosbeek MR, Vos JM, Meinders MBJ, Velthorst EJ, Scarascia-Mugnozza GE (2007) Free atmospheric CO_2 enrichment (FACE) increased respiration and humification in the mineral soil of a poplar plantation. Geoderma 138(3–4):204–212. doi:10.1016/j.geoderma.2006.11.008

131. Pacaldo RS, Volk TA, Briggs RD (2013) Greenhouse gas potentials of shrub willow biomass crops based on below- and aboveground biomass inventory along a 19-year chronosequence. Bioenergy Res 6(1):252–262. doi:10.1007/s12155-012-9250-y

132. Soussana JF, Loiseau P, Vuichard N, Ceschia E, Balesdent J, Chevallier T, Arrouays D (2004) Carbon cycling and sequestration opportunities in temperate grasslands. Soil Use Manag 20:219–230

133. Post WM, Kwon KC (2000) Soil carbon sequestration and land-use change: processes and potential. Glob Chang Biol 6:317–328

134. Zimmermann J, Dauber J, Jones MB (2012) Soil carbon sequestration during the establishment phase of Miscanthus × giganteus: a regional-scale study on commercial farms using 13C natural abundance. GCB Bioenergy 4(4):453–461. doi:10.1111/j.1757-1707.2011.01117.x

135. Dondini M, Hastings A, Saiz G, Jones MB, Smith P (2009) The potential of Miscanthus to sequester carbon in soils: comparing field measurements in Carlow, Ireland to model predictions. Glob Chang Biol Bioenergy 1(6):413–425. doi:10.1111/j.1757-1707.2010.01033.x

136. Derrien D, Amelung W (2011) Computing the mean residence time of soil carbon fractions using stable isotopes: impacts of the model framework. Eur J Soil Sci 62(2):237–252. doi:10.1111/j.1365-2389.2010.01333.x

137. Bonin CL, Lal R (2014) Aboveground productivity and soil carbon storage of biofuel crops in Ohio. Glob Chang Biol Bioenergy 6(1):67–75. doi:10.1111/gcbb.12041

138. Follett RF, Vogel KP, Varvel GE, Mitchell RB, Kimble J (2012) Soil carbon sequestration by switchgrass and no-till maize grown for bioenergy. Bioenergy Res 5(4):866–875. doi:10.1007/s12155-012-9198-y

139. Zimmermann J, Dondini M, Jones MB (2013) Assessing the impacts of the establishment of Miscanthus on soil organic carbon on two contrasting land-use types in Ireland. Eur J Soil Sci 64(6):747–756. doi:10.1111/ejss.12087

140. Cattaneo F, Barbanti L, Gioacchini P, Ciavatta C, Marzadori C (2014) C-13 abundance shows effective soil carbon sequestration in Miscanthus and giant reed compared to arable crops under Mediterranean climate. Biol Fertil Soils 50(7):1121–1128. doi:10.1007/s00374-014-0931-x

141. Zimmermann M, Leifeld J, Schmidt MWI, Smith P, Fuhrer J (2007) Measured soil organic matter fractions can be related to pools in the RothC model. Eur J Soil Sci 58(3):658–667. doi:10.1111/j.1365-2389.2006.00855.x

142. Lichter J, Barron SH, Bevacqua CE, Finzi AC, Irving KE, Stemmler EA, Schlesinger WH (2005) Soil carbon sequestration and turnover in a pine forest. Ecology 86 (7):1835–1847. doi:10.1890/04-1205

143. Pacaldo RS, Volk TA, Briggs RD (2013) No significant differences in soil organic carbon contents along a chronosequence of shrub willow biomass crop fields. Biomass Bioenergy 58:136–142. doi:10.1016/j.biombioe.2013.10.018

144. Conant RT, Ryan MG, Ran IG, Gren A, Birge HE, Davidson EA, Eliasson PE (2011) Temperature and soil organic matter decomposition rates—synthesis of current knowledge and a way forward. Glob Chang Biol 17:3392–3404

145. Aber J, Melillo J, Mcclaugher C (1990) Predicting long-term patterns of mass loss, nitrogen dynamics, and soil organic matter. Can J Bot 68:2201–2208

146. Davidson EA, Janssens IA (2006) Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature 440(7081):165–173. doi:10.1038/nature04514

147. Wander M, Xueming Y (2000) Influence of tillage on the dynamics of loose- and occluded-particulate and humified organic matter fractions. Soil Biol Biochem 32:1151–1160

148. Mueller L, Carsten W, Koegel-Knabner I (2008) Soil organic carbon stocks, distribution, and composition affected by historic land use changes on adjacent sites. Biol Fertil Soils 45(4):347–359. doi:10.1007/s00374-008-0336-9

149. Thevenot M, Dignac MF, Rumpel C (2010) Fate of lignins in soils: a review. Soil Biol Biochem 42(8):1200–1211. doi:10.1016/j.soilbio.2010.03.017

150. Kahle P, Beuch S, Boelcke B, Leinweber P, Schulten HR (2001) Cropping of Miscanthus in Central Europe: biomass production and influence on nutrients and soil organic matter. Eur J Agron 15(3):171–184

151. Smith PL, Kutsch G, Werner L et al (2010) Measurements necessary for assessing the net ecosystem carbon budget of croplands. Agric Ecosyst Environ 139(3):302–315. doi:10.1016/j.agee.2010.04.004

152. Abou Jaoude R, Lagomarsino A, De Angelis P (2011) Impacts of nitrogen fertilisation and coppicing on total and heterotrophic soil CO_2 efflux in a short rotation poplar plantation. Plant Soil 339(1–2):219–230. doi:10.1007/s11104-010-0570-4

153. Lukac M, Lagomarsino A, Moscatelli MC, De Angelis P, Cotrufo MF, Godbold DL (2009) Forest soil carbon cycle under elevated CO_2—a case of increased throughput? Forestry 82(1):75–86. doi:10.1093/forestry/cpn041

154. Warembourg FR, Roumet C, Lafont F (2003) Differences in rhizosphere carbon-partitioning among plant species of different families. Plant Soil 256:347–357

155. Kim SY, Kang H (2008) Effects of elevated CO_2 on below-ground processes in temperate marsh microcosms. Hydrobiologia 605:123–130. doi:10.1007/s10750-008-9325-0

156. Boddy E, Hill PW, Farrar J, Jones DL (2007) Fast turnover of low molecular weight components of the dissolved organic carbon pool of temperate grassland field soils. Soil Biol Biochem 39(4):827–835

157. van Hees PAW, Jones DL, Finlay R, Godbold DL, Lundström US (2005) The carbon we do not see—the impact of low molecular weight compounds on carbon dynamics and respiration in forest soils: a review. Soil Biol Biochem 37(1):1–13. doi:10.1016/j.soilbio.2004.06.010

158. McLaughlin SB, Kszos LA (2005) Development of switchgrass (Panicum virgatum) as a bioenergy feedstock in the United States. Biomass Bioenergy 28(6):515–535

159. Coleman K, Jenkinson DS, Crocker GJ, Grace PR, Klir J, Korschens M, Poulton PR, Richter DD (1997) Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. Geoderma 81(1–2):29–44

160. Parton WJ, Hartman M, Ojima D, Schimel D (1998) DAYCENT and its land surface submodel: description and testing. Glob Planet Chang 19(1–4):35–48. doi:10.1016/S0921-8181(98)00040-X

161. Manzoni S, Porporato A (2009) Soil carbon and nitrogen mineralization: theory and models across scales. Soil Biol Biochem 41:1355–1379

162. Shibu ME, Matthews RB, Bakam I, Moffat AJ, Baggaley NJ (2012) Estimating greenhouse gas abatement potential of biomass crops in Scotland under various management options. Biomass Bioenergy 47:211–227. doi:10.1016/j.biombioe.2012.09.039

163. Xianli X, Liu W, Kiely G (2011) Modeling the change in soil organic carbon of grassland in response to climate change: effects of measured versus modelled carbon pools for initializing the Rothamsted Carbon model. Agric Ecosyst Environ 140(3–4):372–381. doi:10.1016/j.agee.2010.12.018

164. Nieto OM, Castro J, Fernandez E, Smith P (2010) Simulation of soil organic carbon stocks in a Mediterranean olive grove under different soil-management systems using the RothC model. Soil Use Manag 26(2):118–125. doi:10.1111/j.1475-2743.2010.00265.x

165. Powlson DS, Glendining MJ, Coleman K, Whitmore AP (2011) Implications for soil properties of removing cereal straw: results from long-term studies. Agron J 103(1):279–287. doi:10.2134/agronj2010.0146s

166. Paul KI, Polglase PJ (2004) Prediction of decomposition of litter under eucalypts and pines using the FullCAM model. For Ecol Manag 191(1–3):73–92. doi:10.1016/j.foreco.2003.11.007

167. Paul KI, Polglase PJ (2004) Calibration of the RothC model to turnover of soil carbon under eucalypts and pines. Aust J Soil Res 42(8):883–895. doi:10.1071/sr04025

168. Paul KI, Polglase PJ, Richards GP (2003) Predicted change in soil carbon following afforestation or reforestation, and analysis of controlling factors by linking a C accounting model (CAMFor) to models of forest growth (3PG), litter decomposition (GENDEC) and soil C turnover (RothC). For Ecol Manag 177(1–3):485–501

169. Herbst M, Hellebrand HJ, Bauer J, Huisman JA, Simunek J, Weihermuller L, Graf A, Vanderborght J, Vereecken H (2008) Multiyear heterotrophic soil respiration: evaluation of a coupled CO_2 transport and carbon turnover model. Ecol Model 214(2–4):271–283. doi:10.1016/j.ecolmodel.2008.02.007

170. Rethemeyer J, Grootes PM, Brodowski S, Ludwig B (2007) Evaluation of soil C-14 data for estimating inert organic matter in the RothC model. Radiocarbon 49(2):1079–1091

171. Hashimoto S, Wattenbach M, Smith P (2011) Litter carbon inputs to the mineral soil of Japanese Brown forest soils: comparing estimates from the RothC model with estimates from MODIS. J For Res 16(1):16–25. doi:10.1007/s10310-010-0209-6

172. Paul KI, Polglase PJ, Richards GP (2003) Sensitivity analysis of predicted change in soil carbon following afforestation. Ecol Model 164(2–3):137–152. doi:10.1016/s0304-3800(03)00027-9

173. Leifeld J (2008) Biased C-14-derived organic carbon turnover estimates following black carbon input to soil: an exploration with RothC. Biogeochemistry 88(3):205–211. doi:10.1007/s10533-008-9209-4

174. Leifeld J, Zimmermann M, Fuhrer J (2008) Simulating decomposition of labile soil organic carbon: effects of pH. Soil Biol Biochem 40(12):2948–2951. doi:10.1016/j.soilbio.2008.08.019

175. Leifeld J, Zimmermann M, Fuhrer J, Conen F (2009) Storage and turnover of carbon in grassland soils along an elevation gradient in the Swiss Alps. Glob Chang Biol 15(3):668–679. doi:10.1111/j.1365-2486.2008.01782.x

176. Ludwig B, Kuka K, Franko U, von Luetzow M (2008) Comparison of two quantitative soil organic carbon models with a conceptual model using data from an agricultural long-term experiment. J Plant Nutr Soil Sci 171(1):83–90. doi:10.1002/jpln.200700053

177. Hamelin L, Jorgensen U, Petersen BM, Olesen JE, Wenzel H (2012) Modelling the carbon and nitrogen balances of direct land use changes from energy crops in Denmark: a consequential life cycle inventory. Glob Chang Biol Bioenergy 4(6):889–907. doi:10.1111/j.1757-1707.2012.01174.x

178. Anderson-Teixeira KJ, Davis SC, Masters MD, Delucia EH (2009) Changes in soil organic carbon under biofuel crops. GCB Bioenergy 1(1):75–96. doi:10.1111/j.1757-1707.2008.01001.x

179. Varvel GE, Vogel KP, Mitchell RB, Follett RF, Kimble JM (2008) Comparison of corn and switchgrass on marginal soils for bioenergy. Biomass Bioenergy 32(1):18–21. doi:10.1016/j.biombioe.2007.07.003

180. Pacaldo RS, Volk TA, Briggs RD (2014) Carbon sequestration in fine roots and foliage biomass offsets soil CO_2 effluxes along a 19-year chronosequence of shrub willow (Salix x dasyclados) biomass crops. Bioenergy Res 7(3):769–776. doi:10.1007/s12155-014-9416-x

181. Richter GM, Tuck G, Smith J (2001) Modelling variation in C and N loss and effect on N mineralisation after grassland ploughing over

a catchment. In: Rees, RM, Ball, BC, Campbell, CD, Watson, CA (Eds) Sustainable management of soil organic matter. CAB International 67–73

182. Findeling A, Garnier P, Coppens F, Lafolie F, Recous S (2007) Modelling water, carbon and nitrogen dynamics in soil covered with decomposing mulch. Eur J Soil Sci 58:196–206

183. Guber AK, Pachepsky YA, van Genuchten MT, Rawls WJ, Simunek J, Jacques D, Nicholson TJ, Cady RE (2006) Field-scale water flow simulations using ensembles of pedotransfer functions for soil water retention. Vadose Zone J 5(1):234–247

184. Schmer MR, Liebig MA, Vogel KP, Mitchell RB (2011) Field-scale soil property changes under switchgrass managed for bioenergy. Glob Chang Biol Bioenergy 3(6):439–448. doi:10.1111/j.1757-1707.2011.01099.x

185. Guo L, Falloon P, Coleman K, Zhou B, Li Y, Lin E, Zhang F (2007) Application of the RothC model to the results of long-term experiments on typical upland soils in northern China. Soil Use Manag 23(1):63–70. doi:10.1111/j.1475-2743.2006.00056.x

186. Borzêcka-Walker M, Faber A, Borek R (2008) Evaluation of carbon sequestration in energetic crops (Miscanthus and coppice willow). Int Agrophysics 22:185–190

187. Yan M, Zhang X, Jiang Y, Zhou G (2011) Effects of irrigation and plowing on soil carbon dioxide efflux in a poplar plantation chronosequence in northwest China. Soil Sci Plant Nutr 57(3): 466–474. doi:10.1080/00380768.2011.590943

188. Romanya J, Cortina J, Falloon P, Coleman K, Smith P (2000) Modelling changes in soil organic matter after planting fast-growing Pinus radiata on Mediterranean agricultural soils. Eur J Soil Sci 51(4):627–641

189. Grogan P, Matthews R (2002) A modelling analysis of the potential for soil carbon sequestration under short rotation coppice willow bioenergy plantations. Soil Use Manag 18(3):175–183. doi:10.1079/sum2002119